QUANTIFICATION OF SOIL ECOSYSTEM SERVICES FROM ORGANIC FERTILIZED RICE PRODUCTION IN BEAUMONT, TEXAS

A Thesis

by

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ABSTRACT

The socio-economic implications of replacing synthetic fertilizer with an organic fertilizer or soil amendment have been evaluated for rice farmers and their community. We approach this knowledge gap by linking alterations in the soil biophysical properties with meaningful benefit-relevant soil ecosystem services produced in rice paddies. The Denitrification and Decomposition (DNDC) model simulated greenhouse pot experiment from Beaumont, Texas was validated with grain yield, methane and dissolved soil organic C data from a three-factorial randomized experiment with source (Nature Safe (13%) and urea) and rates of N fertilizers (untreated control, 50, 100, 150, 200, and 250 kg N ha^{-1}) as independent variables. Transitioning to organic fertilizer resulted in lower grain yields, higher methane and soil organic C storage; however, no statistical significance was found between treatments. Next, DNDC simulations were calibrated with previously reported field rice yields and methane emissions from conventional and organic amended rice farming near Beaumont, TX. The model was used to simulate single and 22-year simulations. Significant treatment effect was observed for methane emissions, grain yield and soil C and N cycling. A causal chain approach was used to identify relevant beneficial indicators from soil ecosystem services quantified using DNDC simulations. Therefore, changes in soil C cycles were linked to the social cost of C, rice grain yields, potential property losses to flooding events, suspended solids visible in local waterways and aquatic biodiversity. Similarly, changes in soil organic N cycling were linked to reduction in N input for the next crop

cycle, social cost of C, capacity of soil to sustain a double crop, and evasion of eutrophication. Adopting organic amendments improved soil health and ecosystem services but also increased methane and nitrous oxide emissions and the overall global warming potential. Transitioning to organic amendments should be made with the understanding that the tradeoff is increased emissions per hectare comparable to driving a 2015 Ford F150 model for 15,000 miles a year. Finally, the DNDC model has several limitations and needs to include holistic measurements of soil biological, physical and chemical changes to be useful for soil ecosystem services quantification.

DEDICATION

To my parents, AQ, LT and DP.

For their love, support and patience.

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CHAPTER I

INTRODUCTION TO DENITRIFICATION AND DECOMPOSITION (DNDC)

MODEL SIMULATIONS OF RICE PRODUCTION

Introduction

Denitrification and Decomposition (DNDC) is a biogeochemical model with applications extending from analysis of crop production, to changes in soil functions. DNDC can be used to evaluate the effects of farming practices on soil as it tracks changes in the goods and services generated from soil. In addition to upland crops, DNDC has been extensively used to simulate rice production under anaerobic conditions (Li et al., 2002; Yu et al., 2011; Fumoto et al., 2010). The model simulates a saturated soil ecosystem by applying the ‘anaerobic balloon’ concept to measure the shrinking or swelling of the soil anaerobic fraction based on the soil redox potential (E_h) and reduction reaction rates (Li et al., 2004). Similarly, it simulates crop production by tracking photosynthesis, respiration, carbon (C) allocation, tillering, and release of organic C and O_2 from the plant roots (Fumoto et al., 2008). The model links a crop growth sub-model to a soil decomposition sub-model using the rate of root exudation of organic C and other primary substrates consumed in soil reduction reactions.

DNDC model simulations have been tested against site-scale observations for various crops in the United States, China, Thailand, and India (Li, 2000; Cai et al., 2003; Babu et al., 2006). Previous studies have used the model to simulate rice-rotated cropping systems in China to evaluate the effects of water and fertilizer management on ecosystem services (Chen et al., 2016). Similarly, model simulations have also been used to make soil organic C flux measurements from croplands (Tang, 2006). Net C flux was determined using DNDC simulations across mainland China using crop residue and manure as input sources and carbon dioxide flux, methane (CH_4) flux and organic matter leaching as output sources (Tang, 2006). Results from the analysis showed that the soil had shifted more towards the role as a source of C than a sink (Tang, 2006). Similarly, the model was used in Japan to study the changes in methane emission from rice paddies as a function of residue management and synthetic fertilizer (Fumoto et al., 2008). Reduction in soil methane emission were observed from initiating mid-season drainage and intermittent drainage water regiments in rice (Fumoto et al., 2010). The model has been extensively used in mapping the implications of adopting various management alternatives in crop production and has the potential to be used as an effective assessment tool for soil health, and ecosystem services.

The DNDC model was first described in 1992 as a process-based model for simulating the evolution of nitrous oxide as driven by rainfall events, and since then has evolved to simulate many agricultural production systems (Li et al., 1992). Review papers that describe the evolution of the model, its components and its ability to model greenhouse gas from agricultural soils have been published (Gilhespy et al., 2014; Giltrap et al., 2010). Model evolution and application to simulating changes under anaerobic conditions from varying practices in rice production have not been collectively documented. To continue novel research in the field and find practical use for the model to understand not only the biophysical but also the socio-economic aspects of rice farming, a literature review of research studies building up to this approach must be recorded.

Understanding the impact of agricultural management decisions on soil C and N biogeochemical cycles in rice farms can be provided by model simulations. Therefore, the overall objective of this study was to describe the main features and evolutions of DNDC 9.0 and its application to simulate rice production under various management regiments across different regions. The main purpose of this chapter is to review the current state of the DNDC model in anaerobic rice simulations by, (1) describing the main features of the model and its evolution to accurately simulate the flooded rice ecosystem, (2) assessing use of the model to simulate the effects of rice farm management on soil C and N cycles across various regions, (3) exploring other opportunities to extend the use and to improve simulations of rice production.

Current studies using models are focused not only on academic research but are directed towards influencing policy. Models predicting soil organic matter have been historically used as scientific tools to determine best land use types and soil management practices to support policy in agriculture (Oades, 1984; Eilliot and Coleman, 1988; Paul and Robertson, 1989). Crop models can help policy makers by predicting soil erosion, leaching of nutrients, effects of climate change and crop yields (Boote, Jones and Pickering, 1996). In the case of simulating rice production, many models focus only on crop growth dynamics, for example, ORYZA2000 (Bouman and Laar, 2015) and INTERCOM (Bastiaans, et al., 1997) or on rice paddy methane emissions, for example, MERES (Matthews, Wassmann and Arah, 2000). DNDC is one of the few models that takes a holistic approach to rice growth simulations by providing validated information on crop yields as well as soil temperature, hydraulics and trace gas emissions. Therefore, this paper reviews the evolution of the DNDC model to simulate saturated conditions under rice production, and its applications to assess the effects of management alterations on paddy soil C and N cycles.

CHAPTER II

EVOLUTION OF THE DNDC MODEL ANAEROBIC SIMULATIONS: APPLICATION TO SUSTAINABLE RICE PRODUCTION

Introduction

The need for sustainability in the investment of natural resources for anthropogenic benefits can be recognized by accepting the fact that natural resources are limited (Daly and Farley 2010). Approximately one billion hectares of natural ecosystems are predicted to be converted to agriculture by 2050 (Tilman et al., 2001). An estimated 480 million metric tons of milled rice is produced every year (Muthayya et al., 2014), and the Food and Agriculture Organization has predicted a 14% expansion in harvested land for rice with an 88% increase in yield by 2030. Despite its low nutrient content, rice continues to be a primary food source for more than half of the world's population (Sullivan, 2003). The major global effect of rapid expansion of rice farms is increased atmospheric methane emission. According to the EPA greenhouse gas (GHG) emissions report in 2016, 1.6 Gt of methane was emitted from rice cultivation. Therefore, it is important to shift our paradigm from expanding production area to increasing the efficiency of the current resources, since the cropland soil ecosystem functions such as carbon (C) cycle and greenhouse gas regulatory ecosystem services are closely linked to management decisions.

Traditional practices, which include continuous flooding and planting of young seedlings (Datta, 1981) used in combination with nitrogen-based fertilizers, continue to dominate rice farming. Conservationists have challenged this conventional system with an alternate organic system of rice production. Changes have been proposed for application of fertilizers, pesticides, and growth regulators with alternatives such as crop rotation, residue incorporation, animal manure, and biological pest control (Sullivan, 2003). Implementations of organic production techniques have shown an increase in rice yields (Jeyabal and Kuppuswamy, 2001) along with both positive and negative environmental impacts (Hokazono and Hayashi, 2012). For example, application of organic management alternatives in Japan led to reduction of nitrous oxide emissions but at the same time increased methane production. Studying the changes in soil dynamics from incorporating organic farming techniques is thus a prerequisite for evaluating its potential for source reduction and mitigation of atmospheric greenhouse gas emission.

On the other hand, land that is currently being used for agricultural production continues to play an important role in C exchange with the atmosphere. Agricultural management practices regulate soil C cycles and can change the role of farmlands from being a source to a sink of atmospheric C. It is estimated that the world's agricultural and degraded soils have capacity to absorb 50 to 66% of the total historic C added into the atmosphere, and increasing C sequestration in soils has further shown to have a positive effect on crop yields (Lal, 2004). Changes in management practices have previously shown to increase soil C sequestration in agricultural and restored ecosystems from 0 to 150 kg C ha^{-1} $year^{-1}$ in cracking clay soils in the Central Highlands of Queensland, Australia (Armstrong et al., 2003), and 100 to 1000 kg C ha^{-1} $year^{-1}$ in other locations including south Australia, west Canada, and across the United States (Grace et al., 1995; Campbell et al., 2000; West and Post, 2002). Therefore, a final C footprint of rice production from all sinks and sources can be delineated by studying biogeochemical cycle of C in cropland ecosystems.

Most of the available literature is focused more on the influence of water management on methane emission from flooded rice fields (Li et al., 2002; Jiao et al., 2006; Fitzgerald et al., 2000). However, there have been some studies that have shown the influence of changing fertilizer management on the soil organic C cycle as well (Choudhury and Kennedy, 2004). Selective studies that have researched alternative N treatments in rice production and their influence on field methane flux were listed in Table 2.1.

Table 2.1. Compilation of methane emissions recorded for varying N treatment sources and concentrations

Location	**N treatment (rate in kg N ha^{-1})**	**Methane (kg ha^{-1})**	**Reference**
Japan	Ammonium Sulphate (100)	55	Cai et al., 1997
	Ammonium Sulphate (300)	39	
	Urea (100)	88	
	Urea (300)	82	
India	Ammonium Sulphate (120)	33	Ghosh et al., 2003
	Urea (120)	37	
	Potassium Nitrate (120)	28	
	Compost (30) + Urea (90)	187	Das and Adhya, 2014
	Urea (120)	150	
China	Urea + Ammonium Phosphate (150)	136	Zou et al., 2005
	Urea + Ammonium Phosphate (250)	112	
	Urea (100)	105	Qin et al., 2010
	Crop residue manure (100)	127	
Bangladesh	Ammonium Sulphate (400)	96	Ali et al., 2012
USA	Ammonia (aq) (80)	116	Pittelkow et al., 2013
	Ammonia (aq) (140)	118	
	Ammonia (aq) (200)	121	
	Ammonia (aq) (260)	104	
The Philippines	Green manure (Sesbania rostrata) (115)	206	Gon and Neue, 1995
	Green manure (Sesbania rostrata) (160)	443	
	Green manure (Sesbania rostrata) (155)	389	

Comparison of synthetically fertilized with organically fertilized rice systems in India and China has shown that organic amendment significantly increased methane emissions (Das and Adhya, 2014; Qin, et al. 2010). However, another study in the Philippines showed that the amount of methane increase is a function of the quality and the quantity of organic materials added (Gon and Neue, 1995). The green manure produced from harvested nitrogen fixing plant, Sesbania, grown between rice crops that was incorporated without prior composting had the highest methane emission (Gon and Neue, 1995). Literature demonstrated that application of fresh organic amendment such as green manure increased methane emission much more significantly compared to application of composted material with higher humification (Neue, 1993). Similarly, mixing of dehydrated manure produced from crop residue and farmyard manure in China showed a much smaller change in methane emission when compared to urea (Qin et al., 2010). Field-based studies showed that application of ammonium sulfate had the most reduced methane emissions.

Clearly, the effects of alternative managements such as switching to organic fertilizers and changing water regiments during rice growth are not easy to predict. Since field trials for trace gas emissions are limited due to cost, time, and reliability, model simulations are ideal solutions to studying long-term effects on soil security and economic benefits. Simulating the saturated conditions under which rice is grown to check for management effects is complex and requires major modifications. DNDC is a viable option with its easy access, user friendly interface and integrated model algorithm to simulate soil trace gas emissions.

Model Description and Modifications

The Denitrification and Decomposition (DNDC) model is a biogeochemical model created in 1992 to simulate nitrous oxide emissions from upland crops (Li et al., 1992a; Li et al., 1994). The model allows parameterization of major agricultural managements including tillage, fertilization, manure amendment, flooding, and crop rotations. It has been modified extensively to optimize its use in determining greenhouse gas emissions from rice paddies. The model was developed to track the effects of changing farming practices on both crop development and soil environmental factors including temperature, moisture, Eh, pH, and substrate concentration gradient. These factors collectively determine the rates of various competing oxidation-reduction reactions in a saturated soil system which ultimately determines emissions of trace gases such as methane, carbon dioxide and nitrous oxide from the paddy ecosystem.

The soil biogeochemistry sub-model: decomposition

The DNDC model is described as an amalgamation of three interlinked sub-models, soil climate, crop growth, and soil biogeochemistry. Decomposition is a significant part of the soil biogeochemistry sub-model that mediates the rate and conditions under which greenhouse gases are emitted from a saturated rice ecosystem.

Initially, the DNDC model followed the Molina's et al. (1983) approach that divided the soil profile into uniform horizontal layers (2 cm in thickness), where all decomposition occurred layer by layer in the labile and resistant components of three active C pools: decomposable residues (8%), microbial biomass (2%) and humads (10%) (Li et al., 1992a). Decomposition in the initial model caused accumulation of organic C, soluble C, ammonium and nitrates, but did not look at methane emissions until 2000 (Li et al., 1992a; Li et al., 2000). Specific decomposition rates (SDR), before the methane emission feature was added, were based on laboratory studies of decomposition of residues with varying C and N ratios. The laboratory determined specific decomposition rates were adjusted by a fixed reduction factor (0.025) to simulate the lower rates generally observed under field conditions (Li et al., 1992a). When this reduction factor was used with the methane features, it overestimated the pools of decomposable soil organic C and had to be corrected to get reasonable predictions of emissions. In which case, a much lower soil organic C composition of the active C pools had to be used (Fumoto et al., 2008; Li et al., 2000; Cai et al., 2003; Babu et al., 2006). This indicated that the modeled methane simulation showed sensitivity to both initial soil organic C composition and concentration.

Evolution of soil redox potential was added to the model later to better simulate fermentation processes (Li et al., 2004). Wetland soils are characterized by the presence of saturated zones with depleted oxygen levels, increased denitrification of nitrates, and redox reactions with electron donors and acceptors such as manganese, iron, sulfate and nitrates (Achtnich et al., 1995; Li et al., 2004; Fumoto et al., 2008). These reactions are further mediated by the combined effects from soil microbial populations, soil pH, and soil redox potential (E_h) as they determine which redox reaction will consume the available free electrons (Achtnich et al., 1995; Li et al., 2004; Fumoto et al., 2008). The model fixes the initial concentration of each oxidant based on soil data and tracks changes in the soil redox potential using the Nerst equation (equation (1)),

$$E_h = E_0 + RT/nF \times \ln([\text{oxidant}]/[\text{reductant}]), \qquad (1)$$

where E_h is redox potential (mV) of the oxidation-reduction reaction, E_0 is the standard electromotive force (mV), R is the gas constant (8.3 J mol^{-1} k^{-1}), T is the absolute temperature (in kelvin), n is the transferred electron number, F is the Faraday constant (96485 C mol^{-1}), [oxidant] is the concentration of the dominant oxidant and [reductant] is the concentration of the dominant reductant in the system in mols.

This equation determines E_h by using the concentrations of the dominant oxidants and reductants in the soil liquid phase. The change in oxidant concentrations in the soil solution based on microbial consumption is determined using the Michaelis and Menten equation (equation (2)). Therefore, the model determines the rate of each reduction reaction based on the concentration of oxidants and available electron donors.

$$F_{[\text{oxidant}]} = a[DOC/(b + DOC)] \times [\text{oxidant}/(c + \text{oxidant})], \qquad (2)$$

where $F_{[\text{oxidant}]}$ is the fraction of the oxidant reduced during a time step, DOC is the available C concentration, and a, b and c are coefficients of the reaction. Furthermore, by combining the Nernst equation with the Michaelis-Menten equation using the concentration of oxidants as a common factor, the model defines the anaerobic volume fraction of the soil (Li et al., 2004).

Soil E_h is used to divide the soil into aerobic and anaerobic microsites present outside and inside the determined anaerobic volume fraction. The size of a microsite is used to determine the allocation of various substrates within each site. The model also defines that reduction reactions can occur only within an anaerobic fraction and oxidations reactions in the aerobic fraction. As the rate of the oxidation-reduction reactions changes, the anaerobic volume fraction, which is coined as an “anaerobic balloon”, tends to shrink or swell (Li et al., 2004).

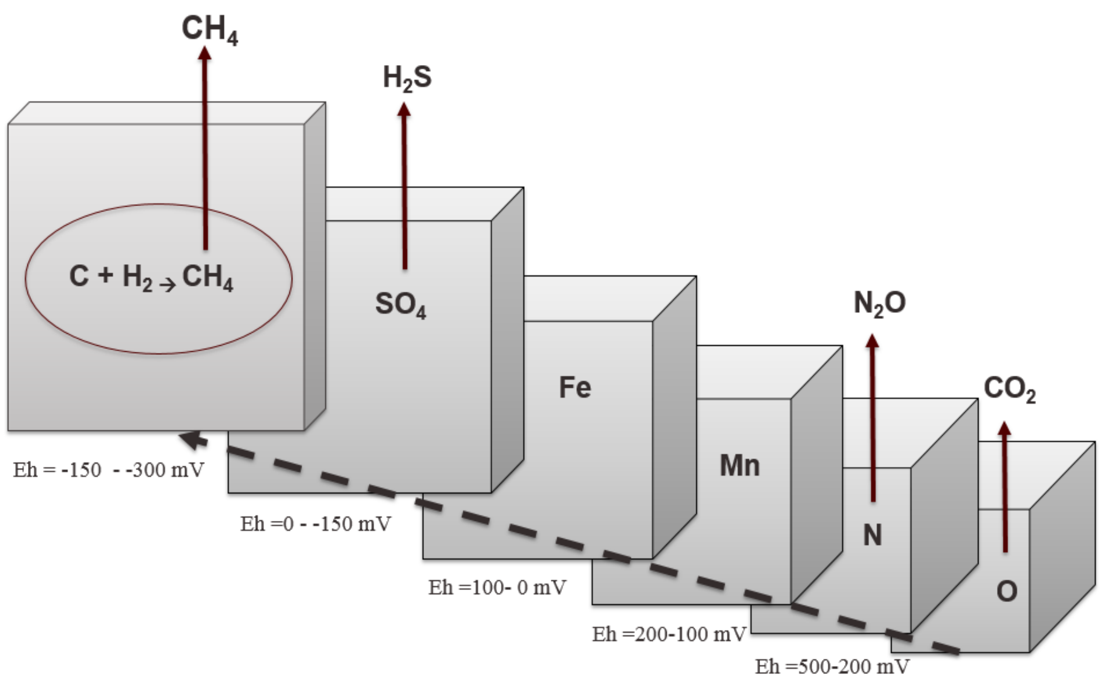


Figure 2.1. Microbe mediated reductive reaction sequence and soil redox potentials for each anaerobic balloon formation and transition with their respective trace gas emissions.

The model follows the principles of the redox ladder (Fig. 2.1) under which it proceeds from one reduction reaction to another only once all the oxidants for the succeeding reaction are depleted. Therefore, as the soil Eh declines, all oxygen is consumed that bursts the first anaerobic balloon allowing the system to move to the next reduction reaction in the ladder. The series is driven by sequential depletions of oxygen, NO_3^-, Mn^{4+}, Fe^{3+} and SO_4^{2-} and finally proceeds to methanogenesis which occurs at a much higher Gibbs free energy. Thus, by tracking concentration changes in oxidants and other ions that cause the formation and depletion of a series of anaerobic balloons, the model estimates soil Eh along with trace gases such as carbon dioxide, nitrous oxide and methane (Li et al., 2004).

The rice growth sub-model

The initial C and N dynamic model was supplemented with an empirical plant growth sub-model and cropping practice routines (fertilization, irrigation, tillage, crop rotation and manure amendments) (Li, et al., 1992). The model used a daily crop growth curve to calculate N extracted from the soil NO_3^- and NH_4^+ pools. Crop growth was dependent on soil water content and N concentrations. This growth model was validated against short-term decomposition experiments, seasonal soil carbon dioxide respiration patterns and a long-term soil organic C sequestration values (Li et al., 1994). Li et al., (1994) also demonstrated that the model could help isolate agricultural practices such as manure additions, which enhance sequestration of C in arable land (Li et al., 1994).

To improve this initial crop growth model, an alternative Crop-DNDC that integrated detailed crop growth algorithm was developed to simulate C, N and water cycles (Zhang et al., 2002). In addition to physiological processes, water stress and nitrogen stress were also calculated and related to soil biogeochemical and hydraulic processes. Crop residues incorporated at the end of growing season was also quantified.

This was a good alternative to use for the empirical model when greater detailed crop growth information was available (Giltrap et al., 2010).

To optimize DNDC to simulate rice development and growth, MACROS, an established model of crop carbon metabolism was added to the crop growth sub-model (Li et al., 2004). This modification allowed the model to relate methane production with plant C metabolism by linking methanogenesis as a function of available electron donors, C and O_2, that are released from plant roots and as a function of tiller conductance. Root growth rate and total biomass were modeled, and the root density was assumed to be uniform over the top 20 cm of the horizon. The model also allowed simulated rice to be rotated with other crops, allowing predictions of effects of crop rotation on the greenhouse gas emissions from rice fields.

In 2008 the model was further revised, and methane production was directly linked to plant C metabolism. Additional variables that account for concentration of H_2 and dissolved organic C, which are both immediate electron donors in soil redox reactions (Achtnich et al., 1995), were added to the soil biogeochemistry sub-model. Thus, the effect of alternative electron acceptors was included in the model methane predictions. Following these revisions, the DNDC model has been applied for various studies on rice farms as described in the following section.

DNDC Application in Rice Production

A compilation of the most compelling papers demonstrating the model application to rice ecosystems for various countries are described in Tables 2.2 through 2.4. Experimental designs, water managements between continuous flooding (CF), midseason drainage (MD), intermittent drainage (ID), and shallow flooding (SF), and fertilizer concentrations along with results for each study have been listed.

China

With 18.3% of the global paddy fields, China is the largest rice producing country in the world and is accountable for 27.3% of global rice production (FAO, 2013). However, as the largest rice producing country, China also faces a wide range of environmental degradations from rice farming and approximately emits 7.41 Tg methane per year from rice production (Yen, et al. 2009). As a result, many studies have been conducted to explore possible management alternatives that mitigate greenhouse gas emissions and promote sustainable farming. Therefore, availability of data and high impact from the region has led some of the earliest studies of DNDC model rice simulations to be conducted in China.

The focus of application of the DNDC model in simulating rice production systems in China has been to estimate methane emissions from paddy fields Table 2.2. The model was applied to study transitional effects from continuous flooding to mid-season drainage (MD) on methane emissions (Li et al., 2002). They made methane emission predictions at a regional scale using historical data from 1980-2000, when midseason paddy drainage was implemented in China (Li et al., 2002). Transitioning to MD for all paddy fields in China showed a reduction in the overall methane flux (Li, et al. 2002). Further studies on the effects of water management transition showed other effects including an increase in nitrous oxide (Li et al., 2005). Similarly, this approach was combined with fertilizer effect that showed shallow flooding and application of ammonium sulfate together showed the greatest reduction of methane emissions (Li et al., 2006). A later study in 2016 also showed that shallow flooding with optimal N application rates can also conserve N in the soil while increasing yields at the same time (Chen et al., 2016). However, it must be noted that although the model predictions so far have been sufficient to get trends, daily emissions were not always accurate for changing soil properties.

Later studies were focused more the applications of the model at upscaling from field to regional scales and integration with geographic information system (GIS), grid data, and remote sensing to strengthen the input data features (Zhao et al., 2015; Yu et al., 2011; Zhang et al. 2011). Incorporation of a polygon-based data set showed that fewer variations of DNDC simulated soil area, methane emission and rates were observed for grid data sets with cell size less than 2 km in comparison to cell size larger than 8 km (Yu et al., 2011). Similarly, a GIS database that included rice maps derived from the Landsat TM images was constructed to hold spatially differentiated input for regional simulations of a 10.93 million ha domain using DNDC (Zhang et al., 2011). More recently, soil data sets with high spatial variability were prepared in raster format with a pixel size of 100 by100space m and additional scripting was constructed to use the raster data set for DNDC regional simulation (Zhao et al., 2015). By coding to use spatial raster files, DNDC site-specific mode was used to simulate rice production at a regional scale with increased spatial resolution of the input and, resulted in improved model representation of spatial variability of the modeled rice yield (Zhao et al., 2015).

Application of the model to evaluate other benefits besides mitigation of methane emission has also been reported in China. A 2006 study determined emission and accumulation patterns for all croplands, a majority of which grew rice or double cropped with rice and wheat, from mainland China (Tang et al., 2006). A C net flux analysis at a regional scale of rice fields from all of mainland China showed an annual soil organic C loss at a rate of 78.89 Tg C yr^{-1} (Tang et al., 2006). Another 2016 study was conducted for delineation of soil ecosystem services in terms of total grain yield, N leaching reduction and greenhouse gas (methane, nitrous oxide and carbon dioxide) emission reduction (Chen et al., 2016). This is the only paper from this region that identifies soil ecosystem services, but it still lacks the steps required to link the services to human welfare, which is important in establishing the true implications of altering management practices on the farmers and their community.

Table 2.2. Compilation of DNDC model application in rice production in China.

Reference	Experimental Design with DNDC	Water Manage-ment	Fertilizer	Results
Li et al., 2002	Change in methane emission from water management changes during 1980-2000	CF MD	Varied	CF to MD reduced methane fluxes by approximately 5 Tg CH_4 yr^{-1}.
Li et al., 2004	Model sensitivity and upscaling	CF MD	Varied	Most Sensitive Factor developed for DNDC. Reduced uncertainty related to upscaling.
Li et al., 2005	Model impacts of water management on greenhouse gas emission in 1990	CF MD	Varied	CF to MD reduced methane fluxes by 40%, increased nitrous oxide fluxes by 50%, and reduced carbon dioxide fluxes by 0.65 Tg CO_2-C yr^{-1}.
Li et al., 2006	Model effects of alternative water management, fertilizer and rice straw on greenhouse gas emission from 2000-2020	CF MD SF	Off-season straw Sulphate fertilizer Slow-release fertilizer	Order of net greenhouse gas emission reduction: upland rice> SF> ammonium sulphate> MD> off-season straw> slow-release fertilizer> CF.

Table 2.2. Continued

Reference	Experimental Design with DNDC	Water Manage-ment	Fertilizer	Results
Tang et al., 2006	Organic C storage estimation from varied croplands in 1998	Varied	Urea (40%) Ammonium bicarbonate (40%) Ammonium dihydrogen phosphate (20%)	soil organic C is lost at a rate of 78.89 Tg C yr^{-1}
Zhang et al., 2009	Simulation of global warming potential from rice fields 1982-2000	Varied	Varied	Rice-wheat rotation had cumulative GWP of 565 Tg CO_2 equivalent. Gleyed paddy soils had the highest GWP. Submergenic paddy soils had the lowest GWP.
Yu et al., 2011	Model the effects of soil spatial resolution on quantifying methane and nitrous oxide emissions from rice fields	Varied	Varied	For accurate model simulation, a grid data set with cell size of 8 km was optimal

Table 2.2 Continued

Reference	Experimental Design with DNDC	Water Manage-ment	Fertilizer	Results
Zhang et al., 2011	Integrate remote sensing mapping with the model to quantify methane emissions from the Sanjiang Plain of Northeastern China in 2006	CF	Synthetic fertilizer (60 kg N ha^{-1}) Urea (90 kg N ha^{-1}) Urea + Synthetic (60 kg N ha^{-1})	2.44 million ha of rice fields in the plain emitted 048 -0.58 Tg CH_4-C Fields with high soil organic C, long crop season and high rice biomass enhanced CH_4 production.
Chen et al., 2016	Model impacts of water and fertilizer management on ecosystem services of rice-rotated systems in 2005	CF MD SF	Optimized based on water management	SF with optimal N application can enhance national scale ecosystem services: 34.3% reduction in greenhouse gas emission, 3.8% reduction of overall N loss, and 1.7% increase of rice yields.
Zhao et al., 2015	Regional application of the model for rice in NE-China in 2009	Varied	Varied	Specific model input data was defined for each raster cell with an area of 100m × 100m. External scripts were developed to assimilate the raster input data automatically for efficient use of model in regional analysis.

Continuous flooding (CF); Midseason drainage (MD); Shallow flooding (SF)

India

A compilation of studies that applied the DNDC model in simulating rice production systems in India have been listed in Table 2.3. India is also a major contributor to the global rice market and produces about 104.32 Mt of rice in an area of 4.3 million ha. In recognition to an increasing demand for the crop and its consequent greenhouse gas inputs, application of the revised DNDC v 8.5 (Li et al., 2004) was immediately seen in India for upscaling and sensitivity analysis of simulated greenhouse gas emission and soil N cycle from rice fields (Pathak et al., 2005; Pathak et al., 2006; Babu et al., 2006).

The concept of water management effects on greenhouse gas emissions was applied to rice fields in India and showed consistent results with the Chinese counterpart (Li et al. 2000; Li et al. 2002; Li et al. 2005). Midseason drainage reduced methane, but increased nitrous oxide emissions in the study (Pathak et al., 2005). A more comprehensive analysis of the N cycle was also conducted using DNDC that lead to demarcations of areas with highest N depletion rates among rice paddies in India (Pathak et al., 2006). Model simulations validated against field trials showed high correlation between observed and simulated results for methane emissions, but higher discrepancies were noted for correlation for nitrous oxide emissions (Babu et al., 2006). Improvements on regional simulations were made by combining DNDC framework with the PALSAR-derived rice maps which further supported the effects brought by change in water management on greenhouse gas emissions from rice fields (Salas et al., 2007).

Table 2.3. Compilation of DNDC model application in rice production in India.

Reference	Experimental Design with DNDC	Water Manage-ment	Fertilizer	Results
Pathak et al., 2005	Model calibration and upscaling to predict greenhouse gas emission	CF MD	Urea (60-300 kg N ha^{-1})	Estimation of agronomic management influence on greenhouse gas emissions from rice fields in India. MD reduced methane emission by 0.97 Tg of CH_4-C, but increased nitrous oxide emission by 0.02 Tg N_2O-N.
Pathak et al., 2006	Simulate N balance in Rice-Wheat System	Varied	Annual input from varied sources (7-98 kg N ha^{-1})	Model estimated outputs from uptake, volatilization, leaching, and denitrification range (4-175 kg N ha^{-1}). Areas of largest depletion were delineated from upscaling.
Babu et al., 2006	Field validation of the model for methane and nitrous oxide emissions from rice	CF	Urea 0 – 100 kg N ha^{-1}	Less than 20% discrepancies between observed and simulated methane seasonal flux. Relative deviation for nitrous oxide seasonal emission ranged from -237.8 to 28.6%.

Table 2.3 Continued

Reference	Experimental Design with DNDC	Water Manage-ment	Fertilizer	Results
Babu et al., 2006	Modelling methane emission from rice-based production with field validation and sensitivity analysis in 1996	CF MD	Urea (60 kg N ha^{-1}) + green maure or compost or *Azolla* (20 kg N ha^{-1})	Agreement between observed and simulated grain yield, total biomass, N uptake and seasonal methane emission. Increasing mid-season aeration reduced methane rate
Salas et al., 2007	Combined model of greenhouse gas emissions from rice paddies with satellite radar observations	CF MD SF	Urea (140 kg N ha^{-1})	Combination of PALSAR-derived rice maps with DNDC framework enabled detailed regional analysis of impacts of water management and cropping systems on greenhouse gas emissions.

Continuous flooding (CF); Midseason drainage (MD); Shallow flooding (SF)

Japan

A compilation of studies that applied the DNDC model in simulating rice production systems in Japan have been listed in Table 2.4. Accuracy of the DNDC for simulating long-term soil organic C dynamics in Japanese paddy soils was validated against the results from long-term changes analysed over the five sites (Shirato, 2005). Model simulations effective in simulating long-term soil organic C changes, but discrepancies were seen for plots with low soil organic C. The study concluded the DNDC anaerobic balloon concept to be a useful tool in simulating rice greenhouse gas emissions (Shirato, 2005). A new adaptation, DNDC-Rice was formulated when the modelled methane emission was revised by the Japanese research group in 2008 (Fumoto et al., 2008). The revised model simulates crop growth by looking at photosynthesis, respiration, C allocation, tillering, and root release of organic C and O_2 (Fumoto et al., 2008). An additional model variable was added to account for concentration of H_2 and dissolved organic C (DOC) in soil, since they are the main electron donors in other reductive reactions in an anaerobic system that are known to affect the rate of methanation (Fumoto et al. 2008).

Application of the DNDC-Rice using a GIS database at a 1 × 1 km mesh scale was conducted for Japan's Hokkaido region (Fumoto et al., 2010). Model predictions were well correlated with observed data for both seasonal and daily methane flux, and the alternative water regimes (AWR), which included MD and intermittent drainage (ID), showed a high potential of mitigating methane emissions compared to conventional CF (Fumoto et al., 2010). However, DNDC-Rice simulation of nitrous oxide had discrepancies that were caused by model failure to accurately calculate N release rates from the fertilizers (Fumoto et al., 2010). Further research with DNDC-Rice have been conducted in Japan, but the focus of this paper is limited to DNDC v 9.5.

Table 2.4. Compilation of DNDC model application in rice production in Japan.

Reference	Experimental Design with DNDC	Water Manage-ment	Fertilizer	Results
Shirato, Y. 2005	Model suitability to simulate long-term soil organic C dynamics in paddy soils	CF	NPK NPK + Straw	Simulated overall soil organic C variation over 16 to 22-year period agreed with the 5 validation sites. Discrepancies were observed for simulations of plots with low soil organic C content soils. Effective in simulating long-term soil organic C dynamics.
Fumoto et al., 2008	Model revision to simulate methane flux from rice paddy fields with varying residue management and fertilizer regimes	CF MD	Straw application: (1.6 & 3.1ton C ha^{-1}) Urea (0, 100, & 300 kg N ha^{-1}) Ammonium Sulphate (0, 100, & 300 kg N ha^{-1})	Model simulation of rice growth, anaerobic soil processes, and tiller effect revised. Negative effect of ammonium sulphate application on methane emission was simulated. Revised model simulations were accurate for seasonal but not for daily methane flux. Soil heterogeneity and cultivar-specific effects seen on the revised model.

Table 2.4 Continued

Reference	Experimental Design with DNDC	Water Manage-ment	Fertilizer	Results
Fumoto et al., 2010	Assess methane mitigation potentials of alternative water management in rice fields using DNDC-Rice	CF Alternative water regimes (AWR) (MD & ID)	Compound mineral fertilizer (90 kg N ha^{-1}) Coated urea (40 kg N ha^{-1}), ammonium sulphate (20 kg N ha^{-1})	AWR can reduce methane emission by up to 41% compared to CF. Net potential to reduce greenhouse gas from Japan's total rice fields was 4.3 Tg CO_2 Eq. yr^{-1}. National scale analysis requires more detailed construction of national database.

Continuous flooding (CF); Midseason drainage (MD); Shallow flooding (SF)

Discussion

DNDC is in the process of evolving into an accurate and detailed model that can simulate upland and flooded rice growth, soil C and N cycles, and soil water dynamics. The model has the potential to be used for ecosystem evaluation if it provides more details on the effects of changing management on grain yield, soil gas flux (methane, nitrous oxide and carbon dioxide), and soil organic C and N pool composition and concentration. DNDC C and N cycle simulations were demonstrated to be sensitive to soil organic C, clay content (Li et al., 1992), fertilization rate, tillage, and the time of the study (Li et al., 1994). Application of the model for soil organic C dynamics, however, was known to vary greatly between short-term and long-term analysis (Li et al., 1994). This phenomenon most likely occurs because the model does not simulate detailed soil physical and biological changes and interlink them with soil chemical cycles.

DNDC simulations fall short to provide a holistic soil biophysical and chemical impact from alternative managements but is accurate at simulating soil chemical processes, denitrification and decomposition, from saturated soils. Previous studies have simulated rice production under varying rotations, tillage types, fertilizers, manure amendments, and irrigation regiments and the subsequent grain yields, soil greenhouse gas emissions, and nutrient cycling. Model simulated seasonal methane emissions, grain yields, total biomass, crop N uptake and seasonal nitrous oxide emissions from rice production were well correlated with field observed data (Babu et al., 2006; Babu et al., 2006; Pathak et al., 2005; Fumoto et al., 2008). However, model generally overestimated daily methane emissions and had to be re-calibrated by adjusting the microbial activity index, which is an index for indicating the impact of soil environment on its microbial activity. Similar problems were observed for the validation of model simulated nitrous oxide emissions with field data collected in both India and Japan (Babu et al., 2006; Fumoto et al., 2008). Model sensitivity to the microbial activity index indicates that DNDC simulation of methane can be improved by including details about soil microbial population as well as composition of methanogens in the system.

The impacts of DNDC studies on alternative rice management techniques have been to mitigate the global methane emission from rice fields. These studies have concluded that the greenhouse gas flux from fields are sensitive to the amount and frequency of fertilizer input, multiple rotation crop systems, incorporation of crop residue into the field (Chen et al., 2016), variations in water management techniques (Li et al., 2006), and concentration of reducible Fe^{3+} in soil (Fumoto et al., 2008). For example, applying ammonium sulfate fertilizer and mid-season drainage significantly reduced methane emissions (Fumoto et al., 2008; Banger et al., 2012). Adding organic amendment helped increase soil organic C but also increased greenhouse gas emissions. However, net C impact of implementing organic fertilizers with varying C to N ratio in rice production has not been determined.

DNDC, if modified to provide holistic biophysical and chemical quantification of ecosystem services from rice soil ecosystems, can also be used to determine socio-economic impacts of management decisions in rice farming. The flexibility of the model to simulate rice growth under varying management practices will allow for the delineation of the long-term benefits to soil quality and sustainability. Furthermore, DNDC is one of the few models that can simulate crop growth and soil biochemical changes under anaerobic conditions and has the room for upscaling its simulations of soil dynamics to include physical and biological impacts of management changes.

With the capacity to simulate crop growth systems at both local and regional scale, models such as DNDC are applicable to inform policy and evaluation of alternative farming practices. It has been applied to quantify the effects of management choices on agro-ecosystem services generated from rice-rotated cropping system at a macro-scale across mainland China (Chen et al., 2016). Model results concluded that shallow flooding and optimal N applications rates showed highest enhancement of ecosystem services. The model was able to identify production mechanisms that are predicted to reduce greenhouse gas emissions, overall N loss and simultaneously increase rice yields, in comparison to conventional systems (Chen et al., 2016). Socio-economic valuation of soil ecosystem services can be further conducted by combining DNDC model simulations with an integrated social, ecological, and economic concept framework. Therefore, pushing the application of the model from physical measurements towards socio economic inferences on rice farmers and the community.

Many additions and improvements over the years have evolved the DNDC model into a reliable platform for studying the complex chemistry within anaerobic rice production systems. However, model simulations lack soil physical and biological effects on methane emissions and so continue to show deviation from observed measurements for methane simulations. The most recent revisions made by Dr. Fumoto and his team from Japan was in 2008 and culminated in the development of a new version of the model named DNDC-Rice (Fumoto, et al. 2008). However, validation of the revised model and DNDC-Rice continues to show inconsistencies suggesting that the model can be used quantitatively to estimate methane flux but is not sophisticated enough to account for the effects of heterogeneity from soil parameters and rice cultivar species.

Conclusion

This chapter summarizes the evolution of DNDC model and its capacity to accurately simulate soil C cycles under anaerobic conditions that are typical in rice paddies. The model uses the concept of 'anaerobic balloons' that shrink or swell based on the soil redox potential and concentration of dissolved organic C and H_2 as electron donors. Similarly, by adding MACROS crop growth sub-model, rice growth simulations were improved and linked to soil methane production through plant root and soil interaction.

The revised model was extensively used in Asia to simulate both local and regional scale rice production with varying management techniques. The data collected from these simulations were used to identify greenhouse gas mitigating and soil organic C assimilating practices mainly in China, India and Japan. However, field validations of model predictions were not without discrepancies. Therefore, at its current capacity, DNDC would give accurate predictions of more immediate soil chemical changes. It would not give a holistic picture of the long-term effects of changing irrigation, fertilization and other rice farming techniques on soil condition, capability, capital, connectivity and codification of paddy soils.

DNDC was developed with the aim of model applications beyond biophysical measurements and to provide an easy interface and detailed output that to assist development of farming policies. The model simulations would give a true picture if modified to include detailed soil physical and biological dynamics and further combined with various geographical information systems software that reduce error and improve the power of the study. Similarly, the model outputs can be further applied to social, ecological and economic models to quantify soil ecosystem services and link them to the welfare of farmers and their community.

CHAPTER III

APPLICATION OF BIOGEOCHEMICAL MODEL DNDC TO ASSESS THE EFFECTS OF ORGANIC FERTILIZATION SYSTEM IN THE SOIL ECOSYSTEM OF RICE PADDIES IN BEAUMONT, TEXAS

Introduction

The gradual rise in domestic organic rice production still lacks the capacity to meet market demands as US rice imports continue to increase (USDA-Rice Outlook, 2018). Lower yields reported due to restricted nutrient availability from the initial transition to organic fertilized farming is likely why local farmers fail to meet market demands (Berry et al., 2002; Wild et al., 2011). Transitioning to a completely organic system is more environmentally friendly compared to conventional agriculture in terms of C sequestration and soil fertility improvement (Mäder et al., 2002; Qin et al., 2010). This study applies the biogeochemical model, Denitrification and Decomposition (DNDC) to simulate the impacts of the first step towards organic farming, by looking at the effects of using an organic fertilizer over synthetic urea on soil ecosystem services such as nutrient cycling and soil C cycling.

DNDC model simulations have been a great tool to study the mercurial interactions within the soil chemical systems leading to greenhouse gases emissions from rice paddies (Li et al., 2000; Li et al., 2005; Chen et al., 2016; Fumoto et al 2010). It has also been used to study soil organic C and N dynamics in rice paddies (Tang et al., 2006; Pathak et al., 2006). Model validation through field analysis showed some limitations on matching simulated nitrous oxide production in Japan and Thailand but was accurate for methane emission simulations for rice fields in China (Cai et al., 2003). Furthermore, DNDC plant growth and decomposition sub-models simulate the changes in soil dissolved organic C content as mediated by soil microbial activity, which are strongly influenced by organic fertilizers (Mäder et al., 2002; Simmonds et al., 2015). However, the majority of DNDC model-based research on rice has been limited to simulations of conventional fertilized of rice production.

Methane soil methane emissions and C sequestration rates are a function of both N fertilizer type and application rates in rice fields (Cai et al., 1997; Zou et al., 2005; Qin et al., 2010; Pittelkow et al., 2013; Das and Adhya, 2014). Ammonium sulphate significantly reduced field methane emissions but increased nitrous oxide emissions with increasing application rates (Cai et al., 1997; Fumoto et al., 2010). Addition of organic amendment compared to conventional fertilizers is known to increase field methane flux. For example, application of wheat straw and rapeseed cake significantly increased field methane flux (Zou et al., 2005) and addition of dehydrated manure product showed lower C efficiency ratios compared to conventional rice paddies in China (Qin et al., 2010).

On the other hand, the type and source of organic fertilizers are known to have significantly different effects. Implementations of integrated nutrition comprising vermicompost and biofertilizers could be used to increase rice yields (Jeyabal and Kuppuswamy, 2001) but life cycle assessment showed more varying and unstable environmental effects from transitioning to organic rice farming (Hokazono and Hayashi, 2012). Clearly there are varying effects on the soil ecosystem from different organic amendments. This study will compare quality of soil from organic against conventional fertilized rice fields in Texas.

There is a high demand for sustainable and organic management alternatives that will maintain the soil health of rice paddies. Therefore, the objectives of this study are to 1) parameterize the DNDC model to simulate both organic and conventional fertilized rice production in high clay content soils of Beaumont, Texas, 2) validate simulated data against observed data collected from the greenhouse pot study of 2015, and 3) evaluate and compare grain yields, soil methane emission and soil organic C from both treatments. This study proposes to apply the model to simulate organic blood feed compost, Nature Safe, and the conventional, urea fertilized rice farming in Beaumont, Texas. Addition of organic amendment increases labile organic matter sustaining methane emissions but is also known to improve soil C sequestration in rice paddies, and this study will help determine the net C effect.

This paper demonstrates the versatility of the DNDC model to accurately simulate rice growth, with altering fertilization systems. Expected results include successful simulations of rice growth and the biogeochemical cycles pertinent to flooded rice production systems in Beaumont, Texas using DNDC. With increasing awareness among consumers towards personal health and environmental impacts of crop production (Daly and Farley, 2011; Snyder and Spaner, 2010), the study is designed to test the sustainability of switching to organic fertilization systems in rice production.

Materials and Methods

Site Description

The site was chosen due to availability of emission data from a 2015 rice greenhouse trial conducted at the Texas A&M Agrilife Research Extension Center (Fig 3.1) and to compare results with a previously conducted DNDC simulation study of a rice field trial in 1990 (Li et al., 2000). The field of study is located at 30.2°N and 94.4°W.

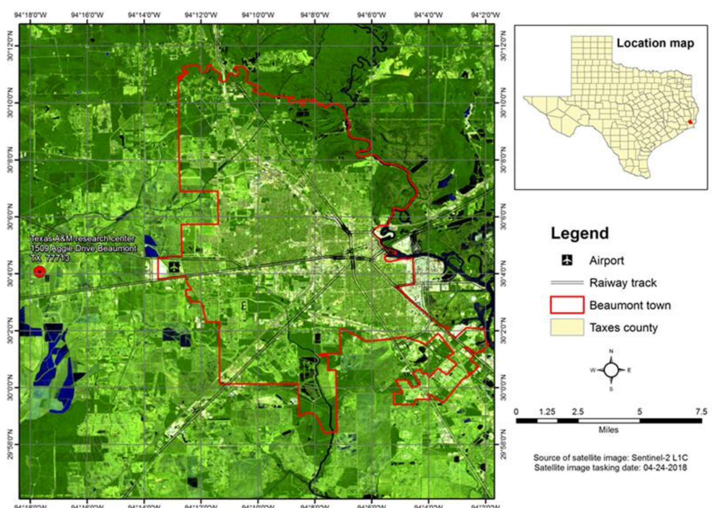


Figure 3.1. Map of Texas with the city of Beaumont in Jefferson County.

The soil in which model simulations and the greenhouse study were conducted is a League clay loam. The League series is a fine, smectitic, hyperthermic Oxyaquic Hapluderts. It is a very deep, poorly drained, very slowly permeable soil in the Gulf

Coast Prairie, Major Land Resource Area of Texas. The mean annual temperature ranges from 20.0 to 22.0 ºC and the mean annual precipitation ranges from 1219 to 1575 mm (NOAA https://www.ncdc.noaa.gov/cdo-web/). High clay content soils are ideal and are popularly used for rice production and native pasture. Rice is grown on this site generally on a 1 to 4 years rotation basis.

Greenhouse trial

The greenhouse experiment was conducted at the Texas A&M Agrilife Research Center at Beaumont, TX from May to August 2015. It was a three-factorial complete randomized design that examined the effect of rice variety (RiceTec XL753 and no plant control), nitrogen source (Nature Safe 13-0-0 vs urea) and six different N application rates (untreated control, 50, 100, 150, 200, and 250 kg N ha^{-1}) on grain yield and soil methane production. There were four replications of each treatment with a total of 96 pots split between organic fertilizer amendments and conventional urea treatment.

Table 3.1. Physical and chemical properties of the Beaumont clay loam used in the pot experiment.

Bulk density (g cm^{-3})	pH	Organic C (g C kg^{-1})	Total N (g N kg^{-1})	Soil Texture	Clay (%)	Silt (%)	Sand (%)
2.26	6.5	15	2.24	Clay Loam	29	31	40

The trial was conducted in polyethylene pots (22 cm in diameter and 22 cm deep). Each pot was filled with 4.5 kg of air dried soil (Table 3.1). The organic fertilizer, Nature Safe (13% N), was incorporated into the soil at the first day of seeding and urea (46-0-0) was applied on the weeks 2, 5 and 8 after planting. Four rice plants were kept per pot. The rice was grown under aerobic conditions with occasional irrigation until 36 days after planting and then a continuous flooding water depth of 3-cm from the surface was maintained. The pot trial management regimen is summarized in Table 3.2. Gas sampling was made from 8:00-11:00 am using chambers every week in two consecutive days and analysed by gas chromatography. A total of 50 mL soil was sampled per pot using a modified plastic cylinder by cutting the needle end at each plant growth stage at the depth of 0-20 cm. The collected soil samples were immediately freeze-dried for extraction and analysis of total water-extractable organic C and total N among other parameters.

Table 3.2. Management regiment for conventional and organic fertilizer treatments under the greenhouse pot experiment from May to August 2015.

Date	DAP	Management of the pots
May 13	-1	Application of Nature Safe (50, 100, 150, 200, and 250 kg N ha^{-1})
May 14	0	RiceTec XL 753 seed plantation
May 18	4	Observed germination
May 20	6	First irrigation event
May 22	8	Second irrigation event, and first soil sample taken
May 28	14	First split application of urea (20% of application rates 50, 100, 150, 200, and 250 kg N ha^{-1}), and second irrigation event
June 2	19	Third irrigation event
June 4	21	Fourth irrigation event
June 19	36	Permanently flooded to maintain a 3 cm height of standing water
June 22	39	Second soil sample taken
June 25	42	Second split application of urea (50% of application rates 50, 100, 150, 200, and 250 kg N ha^{-1})
July 8	55	Third soil sample taken
July 17	64	Third split application of urea (30% of application rates 50, 100, 150, 200, and 250 kg N ha^{-1})
August 12	90	Harvest
September 2	111	Fourth soil sample taken

Note: DAP, days after planting.

Gas emission samples were measured every week starting the third day after planting using gas-tight vials with fixed volumes. Sampling was extended to two days with 36 plants measured for emission each day in 30- and 60-minute intervals. The collected samples were immediately shipped to a USDA soil lab and analysed using a gas chromatograph to determine the concentrations of methane, carbon dioxide and nitrous oxide.

The flux for each gas was calculated using concentrations from the gas chromatograph analysis and the duration of gas sample collection using the following equation (Rolston, 1986):

$$F = \left(\frac{V}{A}\right) \times \left(\frac{\Delta C}{\Delta t}\right) \times \left[\frac{273}{273 + T}\right] \times \left(\frac{P}{760}\right) \times 10^4 \ \times 24$$

where F is the flux (in g gas ha^{-1} day^{-1}), V is the volume of the chamber (in m^3), A is the cross-sectional area of the chamber (in m^2), Δc/Δt is the change in gas concentration inside the chamber with respect to time ($\times 10^{-6}$ m^3 m^{-3} h^{-1}), T is the air temperature (in ºC), 273 is the correction factor between ºC and K, and P is the air pressure in mm Hg.

DNDC model simulation

About the model

The DNDC model is an amalgamation of three main sub-models: soil climate, crop growth, and soil biogeochemistry. The soil climate sub-model predicts soil temperature, moisture, pH, redox potential (Eh) and substrate concentrations based on the soil properties inputs as provided by the user. Similarly, the crop growth sub-model simulates the physiology of any crop of interest, in this case, rice, by using outputs from the soil climate sub-model. The user can provide information on maximum yield, biomass portioning, C to N ratio, season accumulative temperature, water demand, and N fixation capacity specific to the species under study. The crop growth will be simulated by the accumulative temperature, N uptake, and water stress at a daily time step. Outputs of daily photosynthesis, respiration, C allocation, and water and N uptake are provided to check results against field observations to make sure the crops are simulated correctly (Li et al., 1992; Li et al. 1994; Li et al. 2003). The most recent version of the DNDC model, the 'DNDC95.exe' and the 'DNDCgo.dll', was used to run simulations accurate for anaerobic crop production systems (recommended by Dr. Jia Deng, personal communication).

Foremost, the process of articulating the advantages of switching to an organic cropping system requires optimization of the DNDC model to ensure accurate simulations, with minimum error associated to variability in soil properties. To validate the DNDC model, we used the results from a greenhouse experiment conducted in 2015.

For the DNDC model simulations of the greenhouse experiment, daily minimum and maximum air temperature, and precipitation were needed to spin-up the soil C pools prior to the simulation. The National Oceanic and Atmospheric Administration website online climate database was used to retrieve the meteorological data collected at the Beaumont Research Center weather station (NOAA https://www.ncdc.noaa.gov/cdo-web/).

Modification and initialization of the model

Initial soil C pool

The model distributes total soil organic C into several pools such as residue, humads, and humus, and the simulation results are sensitive to the initial soil organic C added by the user. For this study, we assume soil organic C pools to be in a steady state since the source of soil was from paddy fields where rice cropping had been conducted in combination with a cover crop for at least 5 years. This assumption was used to run the model for 70 years to spin-up the DNDC model and to establish steady state for the total C pool.

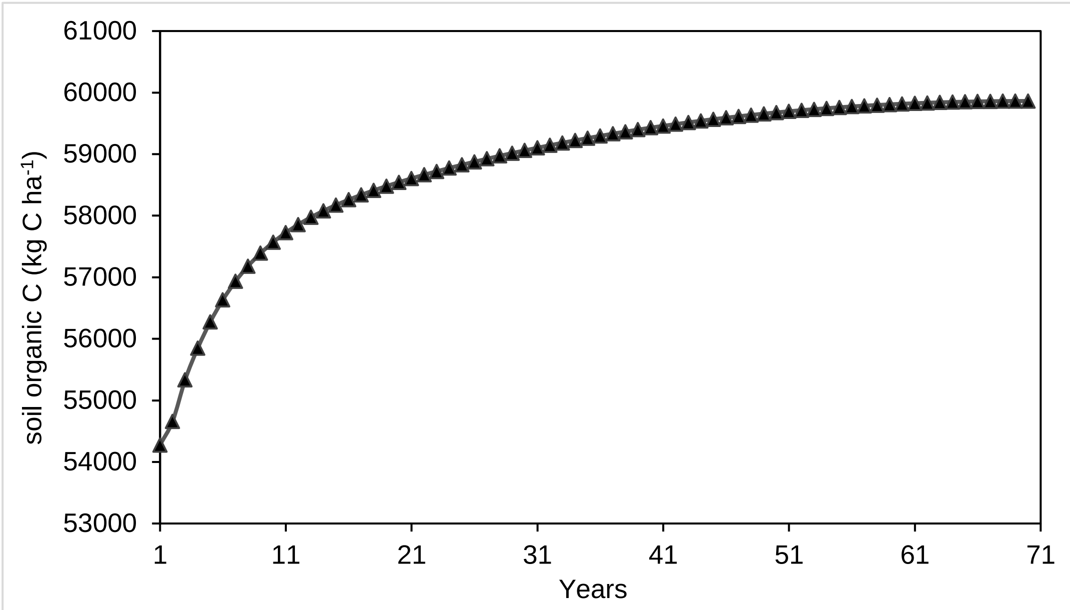


Figure 3.2. The Spin-up phase of DNDC simulation over 70 years

The model spin-up result (Fig. 3.2) does not have a high variation within a year because rice is grown for four months and rotated with cover crops for six months. Since the soil is constantly covered with vegetation, the non-labile soil organic C stays constant throughout the year.

Crop growth

Estimation of plant biomass fraction of root, stem, leaf and grain in addition to their C to N ratios for the hybrid cultivar XL753 was conducted at rice maturity stage. Samples stored from the 2015 experiment were retrieved and processed for root, stem, leaf, and grain total C and N analysis. Total C from plant tissues was determined using the Walkley-Black chromic acid wet oxidation method (Walkley and Black, 1934). Total C, N and C to N ratio for soil, plant root, stem, leaf, and grain are listed in Table 3.3.

Table 3.3. Total C and total N for soil and rice roots, stems, leaves, and grain.

Sample	Total Organic C ($g\ C\ kg^{-1}$)	Total organic N ($g\ N\ kg^{-1}$)
Soil	15.7	2.239
Stem/Leaf	464.7	13.3
Root	319.5	7.1
Grain	411.9	9.15

The DNDC model greatly underestimated grain yield during preliminary runs. Factors such as greenhouse air temperature and higher yielding hybrid cultivar had to be accounted for to have more appropriate yield simulation. The model was set to include a plastic film greenhouse construction that accounted for the altered temperature-moisture soil regimes. Furthermore, the optimal temperature for rice growth was adjusted to between 32 and 35 ºC based on the literature (Nagai and Makino, 2009; Ghadirnezhad and Fallah 2014; Shi et al., 2017). Similarly, the thermal degree days (TDD) of growth for the hybrid species was calculated using the following formula (Iwata, 1984):

$$Thermal\ Degree\ Days\ (TDD) = \frac{(Tmax + Tmin)}{2} - Tbase$$

where, T_{base} for rice was chosen to be 10 ºC based on Gao et al. (1992).

The TDD was estimated to be 2003.5 ºC days. Adjustment of the TDD along with the addition of the greenhouse feature of the model improved the correlation between observed and simulated grain yields.

Input parameters

Each parameter was modified to better fit the greenhouse pot experiment from 2015. The optimized input parameters are listed in Table 3.4. Atmospheric carbon dioxide was changed from a default value of 350 to the recent value of 400 ppm. Soil texture was set to a clay loam with 29% clay based on lab analysis using the Bouyoucos hydrometer method (sand/sillt/clay = 40/31/29). Soil organic C was changed according to lab measured TOC value (15.7 g C kg^{-1}) and model spin-up results. Initial nitrate concentrations were set to 0.36 and ammonium was set to 4.19 g N kg^{-1} based on baseline values measured before the greenhouse pot experiment. Under cropping parameters, maximum biomass was set to 10,000 kg ha^{-1} $year^{-1}$ based on previously reported biomass values (Lee et al., 1991). Biomass fraction and C/N ratio were determined based on lab analysis of plant root, stem and grain from plant samples collected from another field trial using the same cultivar Table 3.3. Finally, tillage, fertilization and manure amendment were added based on the greenhouse experiment data log.

Table 3.4. DNDC input parameters for rice production under organic and conventional fertilization systems in Beaumont, Texas.

Input parameter Value Unit	Conventional Value	Organic Value	Unit
	Climate		
Latitude	30.0	30.0	°N
N in rainfall	2	2	mg N L^{-1}
Air NH3 concentration	0.06	0.06	µg N m^{-3}
Atmospheric CO2 concentration	400	400	ppm
	Soil		
Land-use type	Rice paddy field	Rice paddy field	
Texture	Clay loam	Clay loam	
Bulk density	2.3	2.3	g cm^{-3}
Soil pH	6.5	6.5	
soil organic C content (0-10 cm)	0.015	0.015	kg C kg^{-1} soil
Initial nitrate	0.3	0.3	g N kg^{-1} soil
Initial ammonium	4.32	4.32	mg N kg^{-1} soil
	Management		
Number of cropping systems	1	1	
Span of cropping system	1	1	Year
Years in a cycle of cropping system	1	1	Year
Farming practices for year	1	1	
Number of crops planted	1	1	
Crop type	Paddy rice	Paddy rice	
Planting date	5/14	5/14	
Harvest date	8/12	8/12	
Fraction of above-ground residue left as stub	0.9	0.9	
Maximum biomass	10000	10000	kg C ha^{-1}
Grain fraction	0.40	0.40	
Leaf+stem fraction	0.44	0.44	
Root fraction	0.16	0.16	
Grain C to N ratio	45	45	
Leaf+stem C to N ratio	38	38	
Root C to N ratio	55	55	
Thermal degree days, °C	2000	2000	°C
Water demand	508	508	g water/g dry matter

Table 3.4 Continued

Input parameter Value Unit	Conventional Value	Organic Value	Unit
N fixation index	2.2	2.2	
Vascularity	1	1	
Tilling applications			
Tilling application 1: date	4/1	4/1	
Tilling application 1: depth	30	30	cm
Tilling application 2: date	8/15	8/15	
Tilling application 2: depth	30	30	cm
Fertilizer applications			
Fertilization 1: date	5/28		
Fertilization 1: rate	10		kg N ha^{-1}
Fertilization 1: type	Urea		
Fertilization 2: date	6/25		
Fertilization 2: rate	25		kg N ha^{-1}
Fertilization 2: type	Urea		
Fertilization 3: date	7/17		
Fertilization 3: rate	15		kg N ha^{-1}
Fertilization 3: type	Urea		
Manure application			
Manure 1: date	-	6/18	
Manure 1: amount	-	200	kg N ha^{-1}
Manure 1: C to N ratio	-	4	
Flooding application			
Flooding 1: date	6/19	6/19	
Draining 1: date	8/12	8/12	

Model validation

The data observed in the 2015 greenhouse trial was employed as a reference to compare simulated data using root-mean-square deviation that uses coefficient of determination (R^2) and root mean square error (RMSE). The RMSE will be calculated using the following equation,

$$RMSE = \sqrt{\frac{\sum_{i=1}^{n}(X\,obs, i - Xmodel, i)^2}{n}}$$

where $X_{obs,i}$ and $X_{model,i}$ are observed and modelled values of grain yields and methane emission, respectively, and n is the total number of samples.

Scatter plots with linear regression lines were plotted between observed and simulated results. Mean comparisons were made between treatments using the Student's t-test.

Model redox optimization

A part of the completely randomized designed greenhouse pot experiment was duplicated with four replications per treatment for soil redox and ion analysis. Soil samples were collected every day for the first week after flooding and were extracted with 0.002M $CaCl_2$ solution. The extracted samples were analyzed for nitrate and sulphate concentrations in solution using the Dionix ICS2000 ion chromatograph with a XYZ detector and for iron and manganese concentrations using an atomic absorption spectrometer. Results from the study were incorporated into DNDC and simulations were repeated.

Results and Discussion

Grain yield

The mean grain yields associated with using organic fertilizer were less than with conventional fertilizer, but the differences were not statistically different (p = 0.64) (Table 3.5). The comparable yields between organic and conventional systems may be due to similar N supply between the two fertilizers and shorter growing season in this greenhouse trial compared with that in fields. Also, grain yields of hybrid cultivars are less N sensitive and more efficient scavengers of plant available N (Richmond, 2017).

Grain yields increased with increasing fertilizer rate and were consistent with other studies using the hybrid cultivar with similar N input rates (Richmond, 2017). Although previous studies have showed lower grain yields from systems using organic fertilizer compared to conventional fertilizers (Fig. 3.3) (Rautaray et al., 2003), the type and rate of the organic amendment as well as the rice cultivar are known to have different effects on grain yields (Richmond, 2017; Wilson et al., 2013). The commercially packaged organic amendment used for this project, Nature Safe, previously had shown high yields, attributed to its influence on the rates of N and P mineralization in the soil (Lyamuremye, Dick and Baham, 1996; Siavoshi and Laware 2011). Other organic amendments such as application of cow dung and husk ash together showed an increase in rice grain yield compared to individual chemical fertilizers in a long-term analysis (Saleque et al., 2004). This effect was further supplemented by higher grain yields associated with the hybrid cultivar, RiceTec (RT) XL753, as demonstrated by studies in other locations (Lee et al., 2016).

Table 3.5. Observed and simulated grain yield for plants under six different N application rates of organic and conventional fertilizers.

N rate	Grain Yield			
	Organic N		Conventional N	
	Observed	Simulated	Observed	Simulated
kg N ha^{-1}	---------------------------- kg C ha^{-1} --------------------------			
0	3860	3463	3860	3463
50	4937	4033	5298	4869
100	5640	4289	6434	6297
150	6423	4662	7732	7685
200	7877	5828	8991	9295
250	8946	7001	9841	10801

Correlation between simulated and observed grain yields were high for both organic (r^2 = 0.96 and RMSE = 297) and conventional fertilized (r^2 = 0.99 and RMSE = 184) rice production systems (Fig. 3.3). The greatest difference in yields was seen with lower application rates (50 and 100 kg N ha^{-1}), but the gap narrowed significantly at higher rates (150, 200 and 250 kg N ha^{-1}). This indicates that yields can be maintained despite switching to organic fertilized systems with the hybrid cultivars at moderate to high N application rates. Lower grain yields from organic amendment at lower N application rates can be associated with less readily available nutrient in the initial transition from inorganic N fertilization to organic amendment due to slower nutrient release rates from organic materials (Gopinath et al., 2008). This yield lowering effect from the transition to organic sources of N has been previously reported to be mitigated by using higher application rates of organic N (Gopinath et al., 2008).

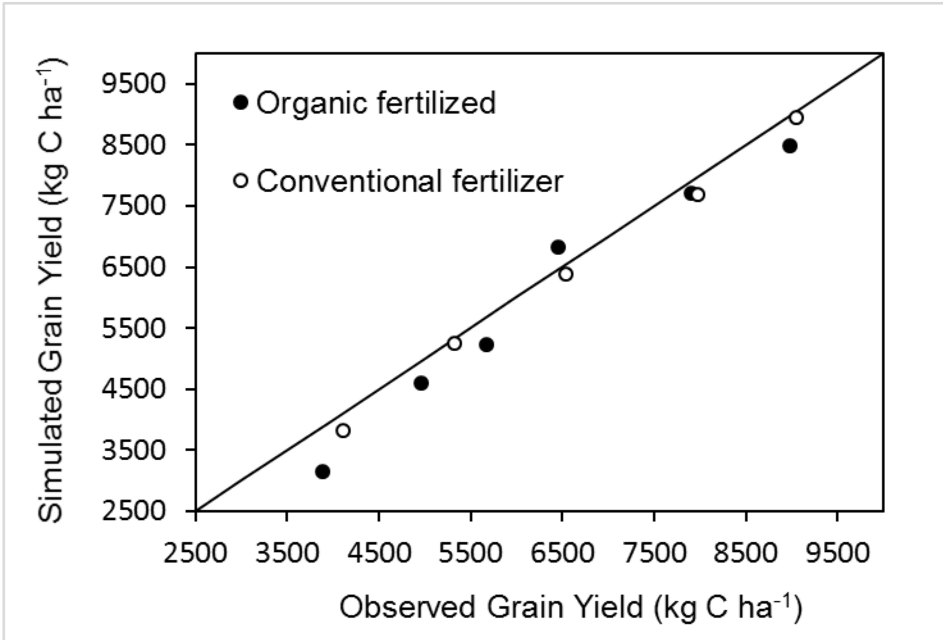


Figure 3.3. Scatter plot for the observed and simulated grain yields for organic (r^2 = 0.96) and conventional (r^2 = 0.99) fertilized rice production system.

The DNDC model was able to accurately simulate plant growth and crop yield for rice. The difference in model accuracy between treatments can occur because the model input feature for inorganic fertilizers provides specific fertilizer types including urea, which specifies the N content and release rate observed. However, the model input feature for manure application type consists of broader groups like farmyard manure, poultry waste, sewage sludge, and meat or blood meal, that might not representative of the true nature of commercially available organic amendment such as Nature Safe, which is a homogeneous, slow release organic fertilizer.

The model performed better at simulating conventionally fertilized rice production and did well with simulating some of the effects of organic fertilizer on grain production. Adding organic matter has an array of effects on the soil, and the DNDC model does not account for all of them. Some parameters important to plant growth not simulated by the model are changes in soil CEC and concentration and composition of other micronutrients available in the soil. As a result, creating rationales about effects of alternative treatments such as organic fertilizers on soil structure, infiltration rate, microbial population and composition, and their subsequent effect on crop growth was limited from model simulations.

Methane flux

Organic fertilizer increased soil methane emissions in comparison to conventionally fertilized systems (Fig. 3.4, 3.5 and 3.6). The greatest methane emission flux was observed from the organic fertilizer treatment applied at the highest rate, 250 kg N ha^{-1}. Similarly, observed seasonal methane emissions were higher for organic fertilizer (Fig. 3.6). However, mean comparisons showed no statistically significant differences between daily (p=0.75) and cumulative seasonal methane fluxes (p=0.17) observed from the two treatments and all six N application rates studied. Methane emissions were observed to increase with increasing application rates of fertilizer for both conventional and organic fertilizer (Fig. 3.4 and 3.6). Higher rates of organic fertilizer increase substrate availability for methanogenesis and result in higher emissions (Das and Adhya, 2014; Qin et al., 2010; Gon and Neue, 1995). However, unlike previous studies that have reported lower emissions with increasing conventional fertilizer rates (Cai et al., 1997; Zou et al., 2005), methane emissions from this greenhouse pot experiment increased with increasing urea application rates. Adding urea in an anaerobic system with no mid-season drainage prevents the ammonia from the urea being oxidized to nitrates as a result suppression of methane emissions in our continuous flooded system was not observed (Achtnich et al., 1995). In addition, we can predict that increasing urea input will have increased the amount of organic N and C which in turn simulated methanogens and increases the amount of methane produced.

Initial model simulation showed good correlation with the intensity of methane observed over the growing season but had problems simulating temporal variability (Fig. 3.5). Simulated methane emissions for both treatments started a week before observed flux in the trial. We proposed that this problem may be associated with the model predicted E_h of the soil, which jumped from a high positive to a high negative value in less than 24 hours after permanent flooding. Soil methanogenesis is the main process that dictates the rate of soil methane emissions, and it is a function of C substrates, soil organic matter degradation and soil E_h (Cao, Dent and Heal, 1995). Correction efforts through experimental measurements of E_h from the top 10 cm of the soils in the pot experiment showed that after permanent flooding, the system took seven days to reach the point of complete saturation and to reach redox values low enough to favor reduction of carbon over other competing electron acceptors in the system.

Errors in daily methane emission simulations have been previously reported where emissions for early growing seasons were underestimated and for late growing seasons were overestimated (Fumoto et al., 2008). The major issue associated with DNDC methane simulations was identified to be the model assumption of a homogenous soil ecosystem, whereas components such as rice residues, Fe oxides, and root biomass are heterogeneously distributed (Fumoto et al., 2008; Fumoto et al., 2010). Similarly, methanogenesis is known to be the inhibited by presence of other electron acceptors such as nitrate, sulphate and ferric ion due to increased competition for electrons (Achtnich et al., 1995). The model methane simulations do not account for varying concentrations of soil iron and sulfate ions, methane conductance by plants based on tiller density and surface oxidation of methane that determine rates and time for soil methane emissions (Fumoto et al., 2008).

The discrepancies in the model simulations of daily methane emissions were rectified once the experimentally measured concentrations of sulphate, nitrate, ferric and manganese ions in the soil along with soil pH and E_h under continuous flooded conditions were sent to the authors of the model (Fig. 3.4). Analysis of saturated soils showed that the both soil sulphate and nitrate ion concentrations declined rapidly but were still present in the first week after flooding which would delay methane emissions.

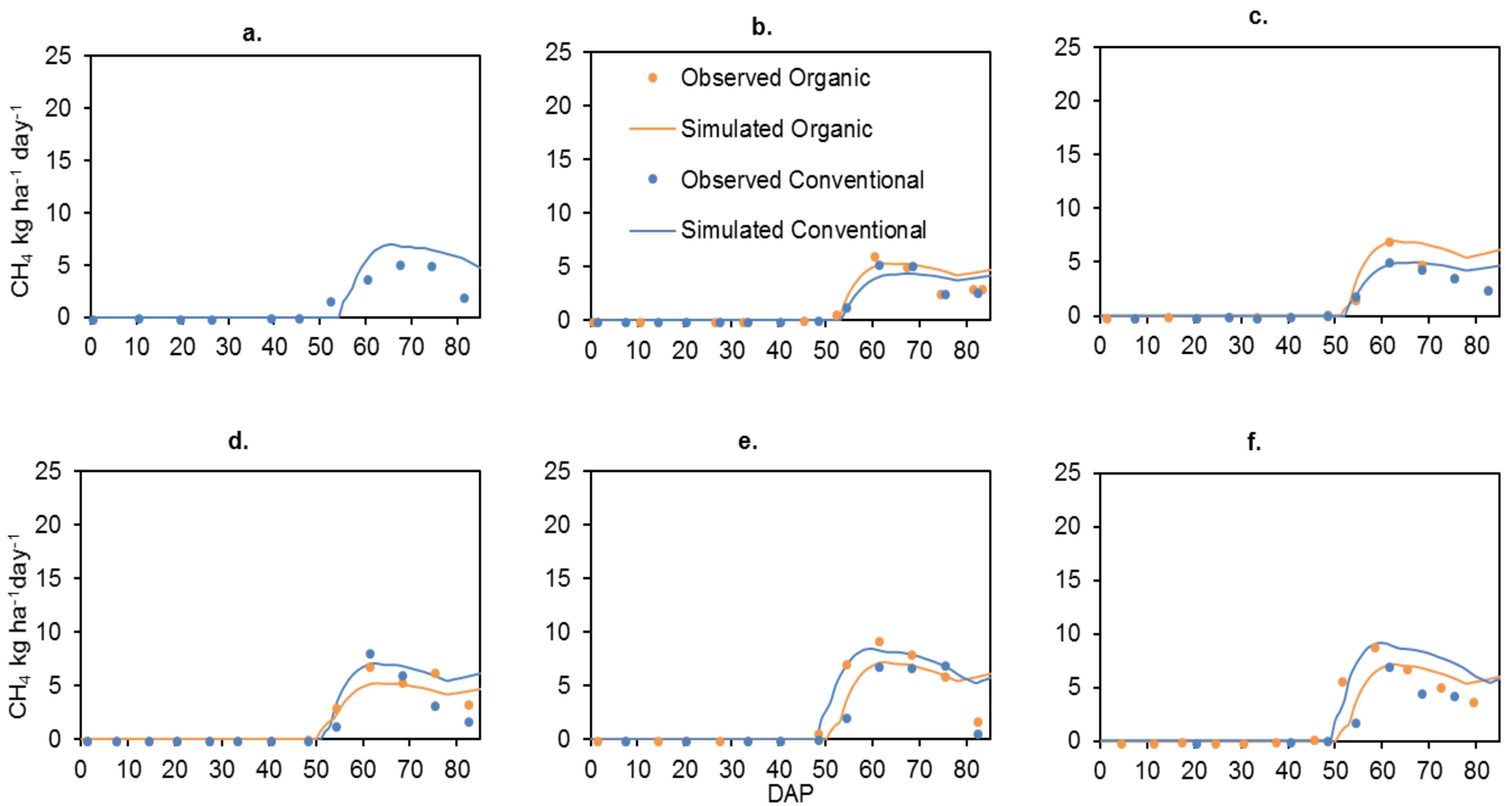


Figure 3.4. Corrected methane emission with in kg gas ha-1day-1 with observed flux in points and simulated flux in solid line for six different organic and conventional fertilizer application rates (kg N ha-1): a. Control; b. 50; c. 100; d. 150; e. 200 and f. 250

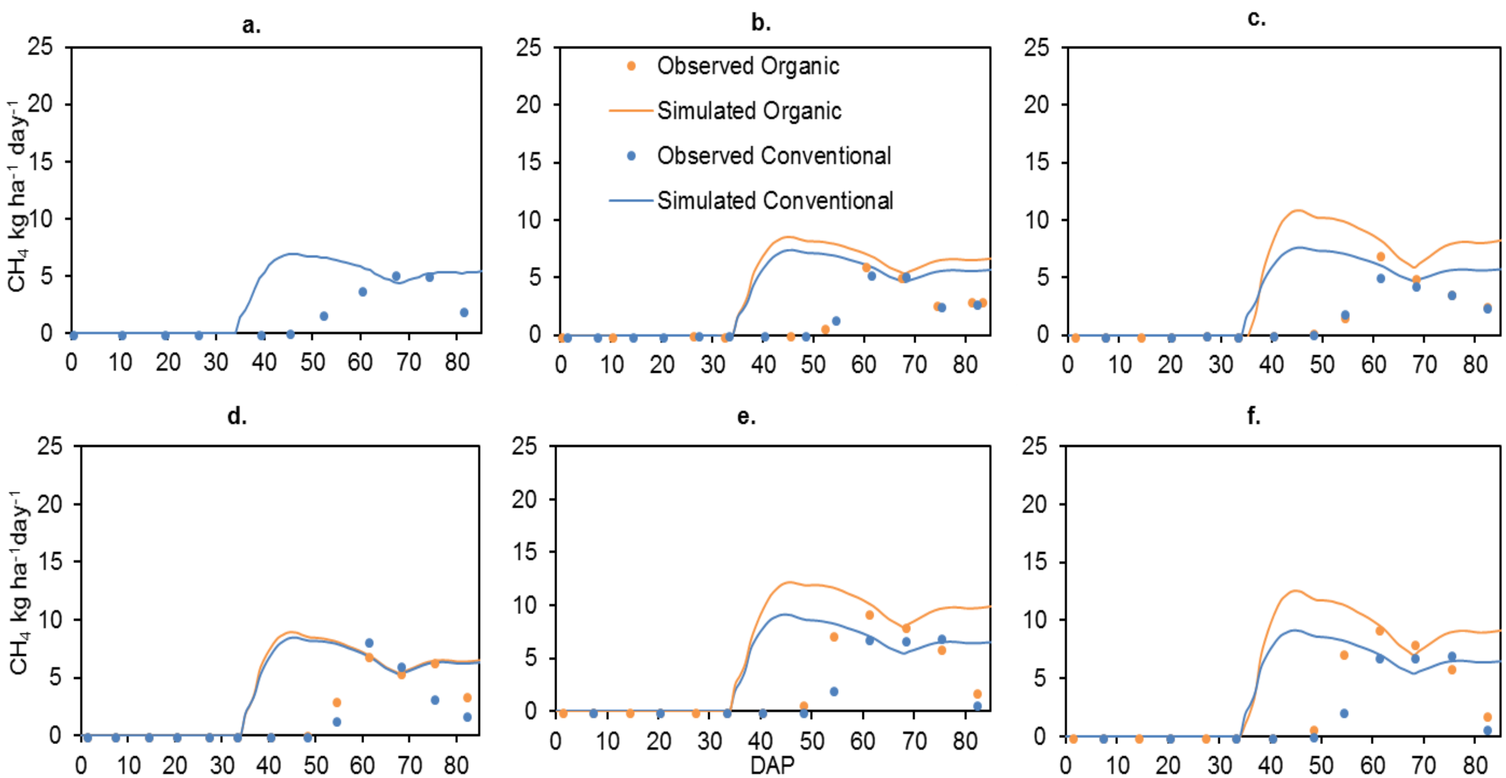


Figure 3.5. Methane emission with temporal error in kg gas ha-1day-1 with observed flux in points and simulated flux in solid line for six different organic and conventional fertilizer application rates (kg N ha-1): a. Control; b. 50; c. 100; d. 150; e. 200; f. 250

Correlation between simulated and observed cumulative methane emissions during rice growing days were good for both organic and conventional treatments (r^2 = 0.57) (Fig. 3.6). The cumulative methane emissions were underestimated for observed values, as measurements were only taken weekly, and extrapolations were made by averaging the measured values. Similar results from seasonal averages were observed in simulations of another pot experiment conducted under a screen house at International Rice Research Institute (Katayanagi, et al., 2012). The model overestimated seasonal methane flux because model measured direct methane emission and did not account for loss of CH_4 to surface oxidation (Katayanagi, et al., 2012). Furthermore, gas measurements were not taken on the day of harvest or the day of drainage, during which highest rates of methane release would be observed from disturbance to the soil. As a result, the observed cumulative gas emissions were generally less than simulated values (Fig. 3.4 and Fig. 3.6).

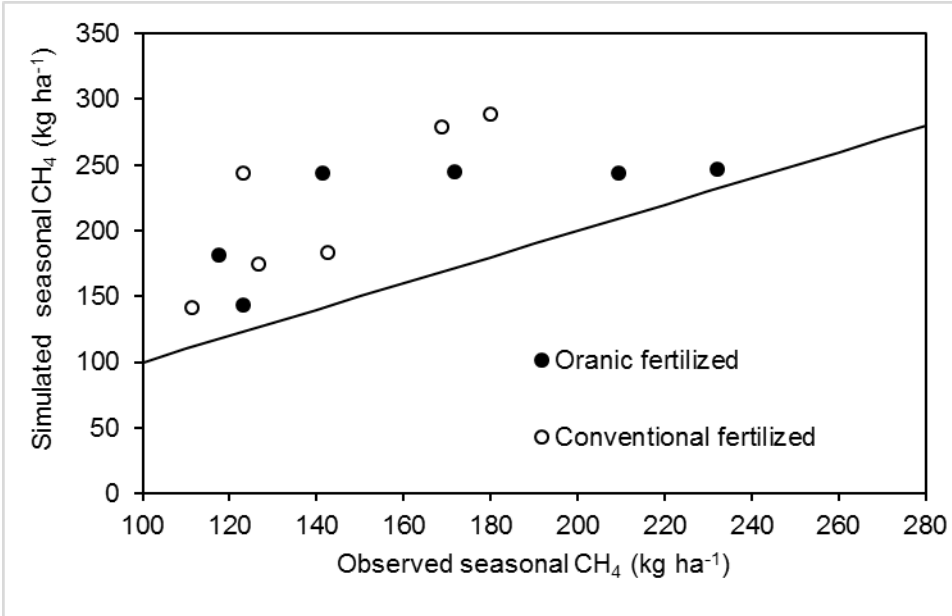


Figure 3.6. Scatter plot for observed and simulated seasonal CH4 emission for organic and conventional fertilized rice production system.

Simulated cumulative methane emissions are similar to observed values up to 150 kg N ha^{-1}, after which the trend has changed or reversed. Methane flux from 200 and 250 kg N ha^{-1} application rates were overestimated for conventional and underestimated for organic fertilized systems (Fig. 3.6). Model simulation of seasonal methane reaches a maximum threshold at 150 kg N ha-1 or more in organic fertilized fields due to limited numbers of plant tillers that conduct the gas from soil into atmosphere. Model simulated methane emissions can be improved by adding features such as soil initial CEC, electron acceptor concentrations, rice cultivar conductance, and concentration of methanogens present in the microbial population (Boote et al., 1996; Fumoto et al., 2008).

Soil organic C

Addition of organic fertilizer increased soil organic C with a linear trend with increasing N application rates through direct addition of organic matter (Table 3.6). Highest concentration of soil dissolved organic C was observed in the second growth stage for N application rates 150, 200 and 250 kg N ha^{-1}. Similarly, a drop in DOC was observed for all N application rates in the third growing stage except for the 250 kg N ha^{-1}. The second stage, the vegetative phase of rice growth, is characterized by an increase in the number of roots and explains the surge in soil DOC. The subsequent decrease in DOC during the reproductive stage is the result of diminishing root growth rates (Moldenhauer et al., 2001). Furthermore, DOC increases with N rates because adding organic matter in the soil increases decomposition rates that in turn adds to the amount of DOC in the system (Simmonds et al., 2011).

Table 3.6. Observed DOC from four growth stages under six different N application rates with organic fertilizer and simulated DOC.

N rate	DOC					
	Stage 1	Stage 2	Stage 3	Stage 4	Total DOC	Total DOC (sim)*
kg N ha^{-1}	---------------------------- kg C ha^{-1} ---					
0	1172	5367	4428	4321	15288	13030
50	3909	5424	4833	2887	17053	19584
100	1213	14976	5882	7479	29550	23951
150	1113	26507	5088	9666	42374	28004
200	1594	16504	4726	3045	25870	31693
250	6157	17102	15509	9038	47806	34971

* simulated stage 4.

Correlation of model predicted dissolved and observed DOC in the final growth stage was good (r^2 = 0.81) (Fig. 3.7). The model simulation of DOC accounts for both plant exudation and decomposition of soil organic C in the soil (Simmonds et al., 2011). DNDC calculates plant derived DOC using a plant growth index and root biomass and calculates the decomposition rate of each soil organic C pool (residue, humads, and humus) based on soil clay content, C to N ratio of bulk organic matter, temperature and moisture (Simmonds et al., 2011). Detailed computation of each source of soluble C substrates enhances the model's performance of simulating DOC. Based on high correlation between simulated and observed grain yield and methane emissions, model predicted values of soil organic C were used for both treatments although validation was provided for organic fertilizer only.

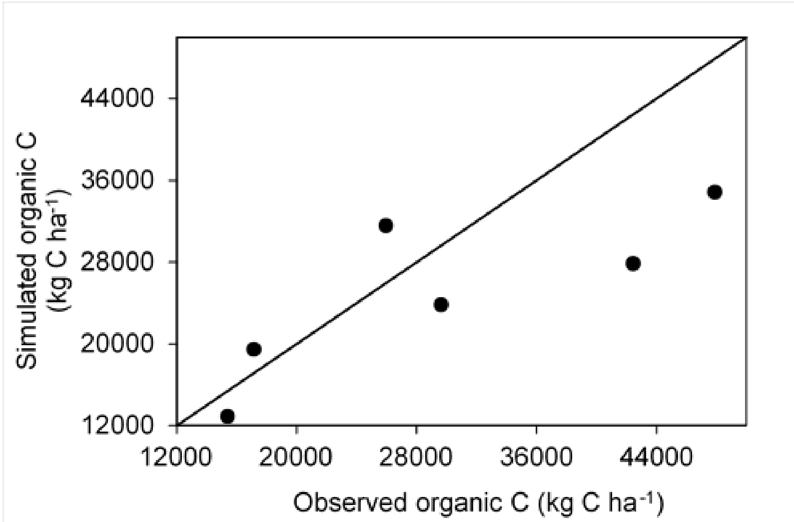
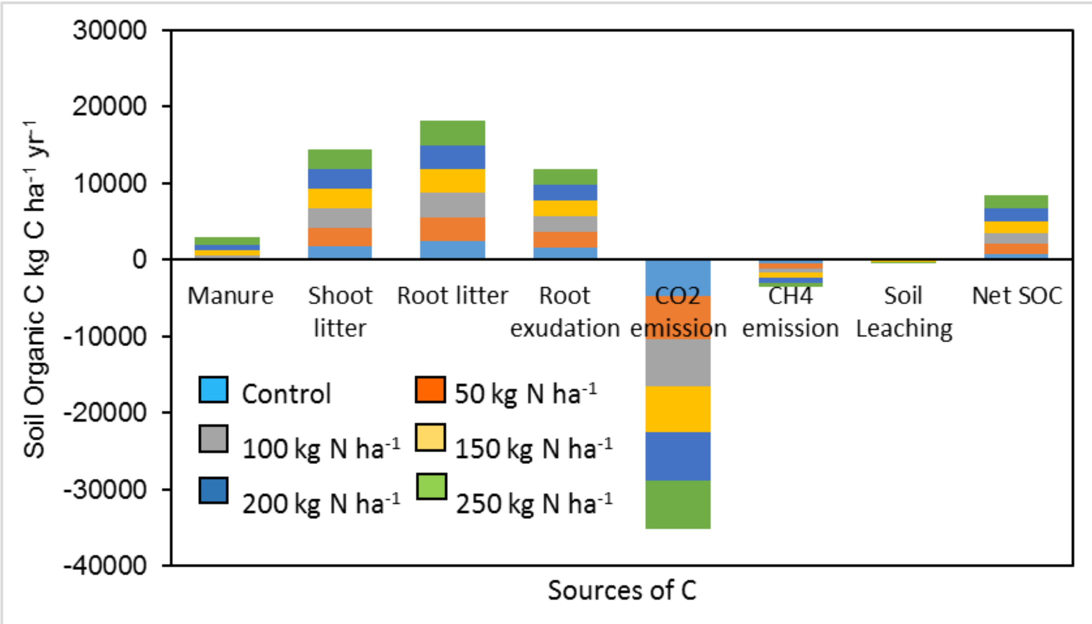


Figure 3.7. Scatter plot for observed versus predicted water extractable organic C as recorded in the final growth stage of organic fertilized rice growth

The DNDC model calculated the values of individual fluxes of C that are added to or removed from the soil under rice production and saturated conditions. These features of model outputs have been previously used to estimate soil organic C stored in the croplands of mainland China by accounting for C input sources such as manure, crop residue and root exudation, and output sources such as soil leaching, and carbon dioxide and methane flux (Tang, 2006).

A similar approach was applied in this study, and the use of organic amendment added greater than 1700 kg C ha^{-1} of C in soil with varying application rates (Fig. 3.8) whereas synthetic fertilizers only accounted for 1400 kg C ha^{-1} in a year (Fig. 3.8). Although methane emissions were relatively higher in organic fertilized fields, the addition of organic matter into the soil negates this increment and created a positive net C effect under a simple mass balance analysis.



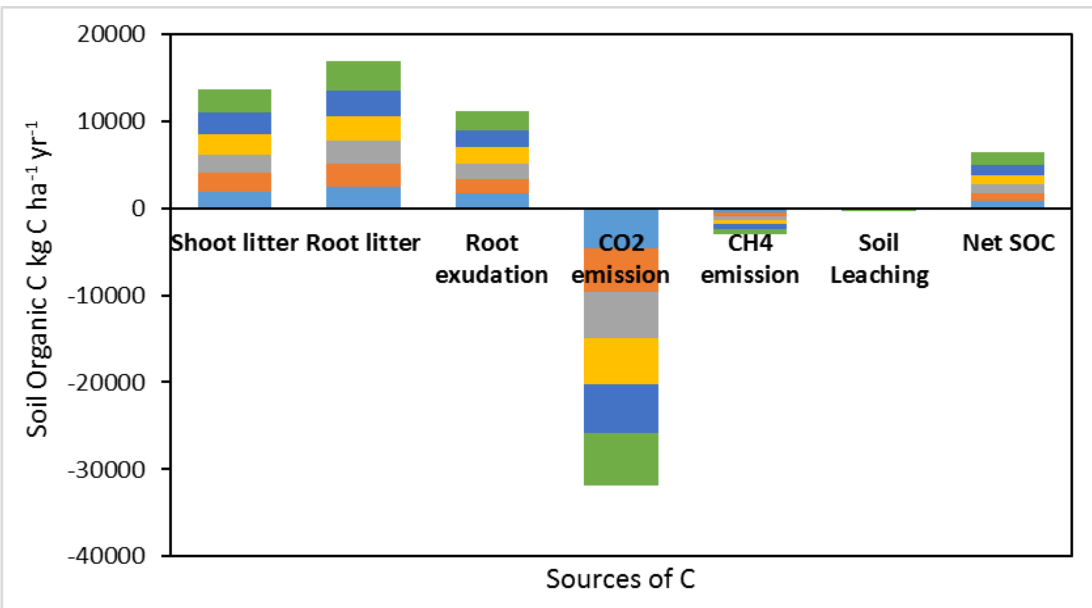


Figure 3.8. Model simulated C fluxes and net C balance in organic (top) and conventional (bottom) fertilized continuous flooded rice paddies of Beaumont, Texas from 2015.

Although adding organic amendment resulted in greater C storage in the soil, greenhouse gas emissions from the field were also much greater. We can predict that although increasing organic matter will improve soil structure, function and will enhance soil ecosystem services, these improvements come with a tradeoff. Methane has a longer atmospheric life and has 25-fold greater global warming potential in a 100-year scale (Pittelkow et al., 2013). Therefore, the decision of transitioning to organic fertilizer must be made with the impacts of higher methane emissions in mind.

Conclusion

The observed rice grain yields were significantly affected by the N rate but not by the source between synthetic and organic fertilizers. The yields increased linearly with increasing N rate within the tested range. However, the N source did not have a significant effect on the yields. Unlike the grain yields, methane emissions were slightly affected by N rates but more affected by the N source. Compared with synthetic fertilizer, organic fertilizer application emitted more methane. The seasonal methane emission ranged from 120 to 210 kg C ha^{-1}. Water extractable organic carbon was highest at the second growth stage. At each sampling stage, effects of N rate on DOC were observed.

Addition of organic fertilizer reduced grain yield and increased methane emissions, but the differences were not statistically significant. Overall, organic amendment with delayed flooding using a hybrid cultivar had comparable yields and methane flux but was more sustainable to soil health as it added organic C to the ecosystem of simulated rice paddies in Beaumont, Texas. However, decision to transition to organic fertilizer system must be made after comparing global warming potentials based on the difference in methane emissions and the impacts it will have on global climate changes at a fixed time scale. Further changes in management to reduce methane emission from organic fertilizer should be pursued.

DNDC is a good model for predicting soil trace gas emissions under a set farming management practices for various upland crops. However, it did show some discrepancies in simulated seasonal and daily methane emissions from rice production under saturated conditions. Although the model interface made data input very easy, it also made understanding of the algorithms behind the simulations problematic and over-parameterizing a possible issue (Li et al., 1992; Li et al., 2000). The model can be improved by adding more details in the input section for organic fertilizers, soil microbial activity index, and soil physical properties. Model strengths include simulation of each soil organic C flux in the soil, and detailed measurement of DOC in the soil liquid phase, which can be used to plot changes in soil organic C during different plant growth stages. Therefore, DNDC is useful to simulate soil chemical processes that lead to changes is DOC and soil methane emissions and should be improved by adding predictions of effects of management on soil biological and physical properties and relating them back to soil chemical processes.

CHAPTER IV

DNDC MODEL-BASED INVESTIGATION OF SOIL ECOSYSTEM SERVICES UNDER STRAW AMENDED UREA FERTILIZATION SYSTEM IN TEXAS RICE PRODUCTION

Introduction

Studies of the mercurial interactions within cropland ecosystems will help delineate the effects of varying farm management techniques across the five dimensions of soil security. In Jefferson County, over 35,000 acres of land are invested in rice cultivation and the soils invested provide ecosystem services that benefit the rice farmer and the local community (USDA-Texas Rice Yield, 2017). Unfortunately, as soil functions decline, and production of marketable goods increase, nonmarket services have gone into a heavy decline (Daly & Farley 2010). Past *capabilities* of rice producing soils include regulation of soil carbon (C), nitrogen (N) and other nutrient cycles. These capabilities are an important reference point that can be used to evaluate the current *condition* of rice paddy soils to sustain grain yields along with original soil functions (Bouma et al., 1998; McBratney et al., 2014).

This study evaluates the capability and condition dimensions of soil security as a function of rice farm management practices, and measures these biophysical changes using the biogeochemical model, Denitrification and Decomposition (DNDC). Unlike most previous studies that stop at quantification of biophysical changes, this research created an integrated social, economic, and ecological concepts (ISEEC) framework (Gunderson, 2001) that links physical measurements of soil ecosystem services (ES) to socio-economic implications (Olander, 2018). This is a vital bridge that links soil capability and condition to the soil security dimensions of *connectivity* by providing effective communication to rice stakeholders and of *codification* by assisting policy makers interested in greenhouse gas emissions from rice fields.

To establish the link between the biophysical and social dimensions of soil security, this project defines natural capital as the available stock of physical and biological resources that are replenishable and accessible (McBratney et al., 2014). The natural capital developed in the ISEEC framework for soil health analysis followed Dominati, et al. (2010), whereby capital stocks of soil properties that yield the flow of ES were recorded. Capital is the fourth dimension of soil security and interconnects natural capital with human, social and economic (financial and manufactured) capitals in a rice production system.

This study combines soil ES valuation models with DNDC simulations to study the effects of varying management practices in Texas rice fields. Previous studies have evaluated the conditions of rice paddy soil ecosystems using DNDC simulations, and predicted that shallow flooding and optimal chemical fertilizer had the highest greenhouse gas mitigation, N loss reductions and grain yield enhancement effects in China and Japan (Li et al. 2000; Li et al. 2002; Li et al., 2004; Li et al., 2006; Chen et al., 2016; Pathak et al., 2005; Pathak et al., 2006; Tang et al., 2006; Yu et al., 2011; Zhang et al., 2011; Chen et al., 2016; Fumoto et al., 2008; Fumoto et al 2010). Modeled results have been validated using field measurements from conventional and organic fertilized rice production systems (Babu et al., 2006; Babu et al., 2005; Cai et al., 2003). Field validations showed that the model overestimated daily methane emissions and had to be adjusted by changing the soil microbial index initially assumed by the model (Babu et al., 2005; Babu et al., 2006). Simulations of rice production in Texas have also been used to compare the effects of switching to a commercial organic fertilizer from urea in rice yields, methane emission and soil organic C (Chapter 2/Results).

Soil ES encompass both goods and services that are beneficial to people and are produced partly or wholly by a functioning ecosystem (Olander et al., 2018). This project studies both marketable goods and non-marketable services that are delivered from a rice soil ecosystem. Marketable provisions or goods are relatively simple to measure and can be further evaluated by monitoring market data of a commodity generated in rice production (Plummer, 2009). However, in case of establishing beneficial values of services, there are no such tangible goods to be marketed. Hence benefit values must be evaluated using indirect methods. Some of the commonly used techniques used for evaluating beneficial values of services are described in Table 4.1.

Table 4.1. Current methods for quantifying non-marketable ecosystem services.

Method	Description	Relevant Example
Avoided Cost (AC)	The benefit gained by a society from avoiding costs that would have been incurred in case of loss or absence of a service. [a]	Prevention of soil erosion provided by changes in soil tillage.
Replacement Cost (RC)	The cost of replacing an ecosystem service with a manmade technology. [a]	Soil nutrient cycling and waste treatment replaced by engineered treatment systems.
Factor Income (FI)	Income enhancing services and the amount gained. [a]	Improved grain yields because of maintained soil health.
Travel Cost (TC)	Services that people are willing to spend a certain traveling cost for; services will be associated with the cost people are willing to spend. [a]	Agricultural communities as a tourist attraction destination. Market for fresh produce, agricultural demonstrations and knowledge distribution
Hedonic Pricing (HP)	Positive effect on prices people are willing to pay for goods associated with certain services. [a]	A price people are willing to pay for farmlands based on the condition and capability of the soil.
Marginal Product Estimation (MP)	Sustainable value of outputs from a variation of (raw) material input, considered as a function of production, can be used to evaluate service demand. [a]	Quality and quantity of product yields from farmlands over time is a product of a raw materials from a healthy soil, such as, its minerals content and thickness.
Contingent Valuation (CV)	Value of services evaluated by creating hypothetical scenarios that pertain alternatives. [a]	Proposal to replace a wetland area with urban structures and weighing the pros and cons of replacing the wetland ecosystem.
Group Valuation (GV)	Demand of a service is established through complete public participation in open debates in an egalitarian society. [a]	Zoning ordinance of farmlands.

Table 4.1 Continued

Method	Description	Relevant Example
Benefit Transfer Method (BTM)	Valuation of services at one location based on estimates used from other services. [b]	To calculate the specific value coefficient V_{Ck} (\$ ha^{-1} yr^{-1}) for rice paddies, value coefficients of cropland, 92\$ ha^{-1} $year^{-1}$, can be combined with some shared features from freshwater wetlands.[c]

a. Costanza et al. 2006 (New Jersey's Ecosystem Services) b. Plummer, M. L., 2009 c. Kreuter et al., 2001

Ecological studies are complex and often require an interdisciplinary approach to attain a meaningful output. This project applies DNDC outputs to create meaningful consequences of management alterations by using soil ES that are useful to the farmers and their community. Rice farmers and the community surrounding the farms are the stakeholders in rice production who would be negatively affected by declining soil functions and subsequent soil ES. Therefore, it has become crucial for soil scientists to develop a language that is relevant and understood by farmers and other stakeholders that make decisions that directly influence soil security (Bagnall and Morgan, 2018). The key to a comprehensive communication between two parties is to identify ecosystem goods and services that people perceive to be directly related to their benefit (Ringold, et al. 2013). They are collectively termed as linking indicators and are studied to ease social and possibly economic evaluation of ecological changes (Boyd, et al. 2015).

The concept of linking indicators has been further developed into a framework to identify stand-alone indicators of ES with benefit-relevant implications on decision-making (Olander, et al. 2018). This guideline combines biophysical measures of an ecosystem with social or economic measures of its value as well as preference towards human well-being. This concept of benefit-relevant indicators was applied in this project following the parameters of the final service concept (Ringold et al., 2013).

The overall goal of the study is to develop a framework that quantifies soil ES gained or lost from adding a straw amendment with synthetic urea fertilizer in rice production in Texas that are relevant to the stakeholders using DNDC simulations. This paper is novel in its development of an approach to apply model simulated data to quantify benefit-relevant soil ES from rice production. The specific objectives of the study are to: 1) develop a combined DNDC and ISEEC framework that translates soil biophysical outcomes of rice production to soil ES, 2) isolate benefit-relevant indicators of soil ES based on causal links to stakeholders in rice production, and 3) quantify soil ES using biophysical values from DNDC simulations.

Soil security connects rice stakeholders, in many platforms including commodity production and ecosystem services (Campbell and Paustian, 2015). The project quantifies this connection by extracting benefit-relevant indicators and focuses the model outputs on rice production to better communicate how soil ES impacts a farm community. We demonstrate a framework for better security of the community soil and its agronomic biophysical functions.

Materials and Methods

DNDC model simulations

DNDC model for this soil ES project was calibrated using field observed data collected by Sass and Fisher in 1991. The study measured the effects of supplementing synthetic urea fertilizer with an organic amendment, Paspalum straw, on field methane emission and soil organic C storage (Sass and Fisher, 1991) in three fields with varying planting dates (Table 4.2). The study was conducted on the high yielding rice cultivar "Jasmine 85" in Beaumont. A primary soil for rice production, a Verland silty clay loam, classified as a fine montmorillonitic, thermic Vertic Ochraqualf, was the soil farmed in this study.

Table 4.2. Field events, in days after planting, for the three fields representing different planting dates. These events were used as input into the DNDC calibration (Sass and Fischer, 1991).

	Field 1	Field 2	Field 3
Straw amendment addition	-4	-4	-3
First urea application	-1	0	-1
Planting date	0 (April 13)	0 (May 18)	0 (June 18)
Second urea application	39	31	32
Permanent flood	41	31	31
Third urea application	75	67	84
Draining	126	114	107
Harvest	139	130	119

For purposes of calibrating DNDC with Sass and Fisher (1991) results, dates of planting, harvest, flooding, drainage, fertilizer and/or organic straw amendment, were changed according to each treatment as reported in their study. Weather (NOAA https://www.ncdc.noaa.gov/cdo-web/) and crop growth model inputs were kept constant for all simulations.

Parameterization of maximum crop biomass improved the correlation between measured and modeled yield. Maximum biomass for the urea fertilized system was set to 7500 kg C ha^{-1} based on previous observed yields for Jasmine 85 treated with urea (Issaka, Buri and Wakatsuki, 2009). This value was reduced to 7000 kg C ha^{-1} for urea with organic straw amendment case simulations to better fit the lower observed yields in the 1991 study. Similarly, the microbial activity index (0-1) was adjusted to 0.8 from a default value of 1 to correlate modeled estimates of seasonal methane emission levels to those measured by Sass and Fisher (1991). Details about management for the DNDC model simulations are listed in Table 4.3.

Table 4.3. DNDC input parameters for rice production simulations under urea fertilizer and urea with straw amendment in Beaumont, Texas.

Input parameter Value Unit	Synthetic Urea	Field 1	Unit
Latitude	30.0	30.0	°N
N in rainfall	2	2	mg N L^{-1}
Air NH_3 concentration	0.06	0.06	µg N m^{-3}
Atmospheric CO_2 concentration	350	350	ppm
Land-use type	Rice paddy field	Rice paddy field	
Texture	Silty clay loam	Silty clay loam	
Crop type	Paddy rice	Paddy rice	
Planting date	4/13	4/13	
Harvest date	8/30	8/30	
Fraction of above-ground residue left as stub	1	1	
Maximum biomass	7500	7000	kg C ha^{-1}
Thermal degree days	3266	3266	°C
Fertilizer applications			
Fertilization 1: date	4/12	4/12	
Fertilization 1: rate	56	56	kg N ha^{-1}
Fertilization 1: type	Urea	Urea	
Fertilization 2: date	5/22	5/22	
Fertilization 2: rate	67	67	kg N ha^{-1}
Fertilization 2: type	Urea	Urea	
Fertilization 3: date	6/27	6/27	
Fertilization 3: rate	67	67	kg N ha^{-1}
Fertilization 3: type	Urea	Urea	
Manure application			
Manure 1: type	none	Straw	
Manure 1: date	-	4/9	
Manure 1: amount	-	77.5	kg N ha^{-1}
Manure 1: C to N ratio	-	45	

After calibration, DNDC model simulations of grain yield and seasonal soil methane emissions were combined from the planting dates and correlated with observed values. The r^2 values were 0.97 and 0.98, from grain yield and methane emissions respectively (Fig. 4.1). Large correlation coefficients and linear behavior between simulated and observed grain yields and soil methane emission suggest the model simulations are valid for rice growth and soil functions under saturated conditions.

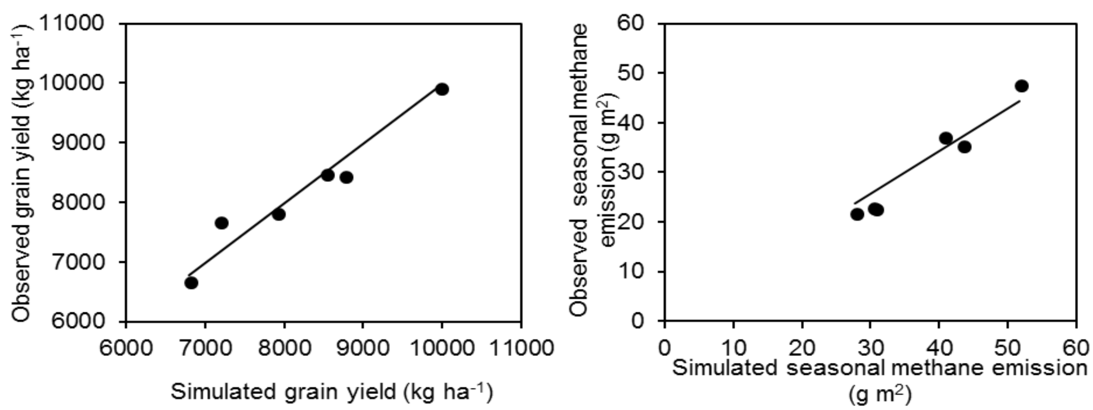


Figure 4.1. Scatter plot to show correlation between simulated and observed grain yield (left) and seasonal methane emissions (right) from three fields with varying planting dates with observed data from Sass and Fisher 1991.

The DNDC model was then used to simulate rice production fertilized with urea and urea with straw amendment over 22 years. Minimum and maximum daily air temperatures and daily precipitation from 1991 to 2012 were obtained from the NOAA Beaumont weather station and used for the model (NOAA https://www.ncdc.noaa.gov/cdo-web/). Results from each year were categorized into three groups using the ratio of annual evapotranspiration (ET) to precipitation. They were, low precipitation years (>1), medium precipitation years (0.81 to 0.99) and high precipitation years (<0.81). Changes in soil organic C content and N content were calculated by conducting a mass balance between flows of C and N into (manure, fertilizer, shoot and root litter) and out of (soil leaching, crop uptake, greenhouse gas emissions and volatilization) the soil. Student t tests for mean comparisons were conducted using SAS (SAS Institute, 2009) between annual average values of grain yield, methane emission, soil residual N, soil organic C and soil carbon dioxide equivalent emissions content simulated for urea fertilization and urea with straw amendment fertilization. ANOVA was used to check for treatment effect for each category of ET and precipitation ratio from the 22-year simulation.

Mechanism for ecosystem service (ES) quantification

This project followed the classic framework of ES valuation consisting of four steps (Freeman, 2002). The first step was to determine the soil natural capital and functions that are affected by changing farm management practices. Paddy soil ecosystem was evaluated by using the quantity of stock or 'Natural Capital' and the rate of flow of soil functions under rice production (McBratney et al., 2014). Soil natural capital studied includes soil organic C content, temperature, pH, residual N content, and redox potential. Soil functions flowing from these stocks include mineralization of N and C, microbial decomposition and denitrification and soil C and N sequestration. For ease of distinction between soil functions and soil ecosystem services, the soil functions are defined as processes that transform the soil stocks into the services and are grouped together with the natural capital under ecological changes (Dominati et al., 2010). The soil stocks and functions then provide beneficial flows of ecosystem services that satisfy human needs (Dominati et al., 2010). Effects of adding organic straw amendment with urea in rice production were compared to urea by itself using DNDC simulated soil stocks and functions pre-and-post-production.

The second step was to assess the changes in soil natural capital and functional after treatments and identify soil ES that were directly affected. Every ecosystem generates an eclectic range of services among which only those services that flowed directly from the defined capital and its functions were chosen. Further screening was conducted to isolate benefit-relevant indicators and soil ES that had a proximal link to the welfare of the stakeholders were chosen. These soil ES included: regulation of C storage, soil nutrient cycling, greenhouse gas emissions (nitrous oxide and methane), provision of food (crop yield), and provision of raw materials (conserved top soil).

The third step was to assess the value of each service change based on causal links to benefit-relevant indicators. This step adds socio-economic meaning to soil ecological changes. The fourth step of soil ES valuation was to accumulate individual values and demonstrate the combined effect of each treatment over 22 years, for communication with the stakeholder population. Connecting socio-economic implications with biophysical changes in soil structure and ES because of management created a holistic picture that was used to evaluate the effects of adding organic straw amendment with urea on the soil security of rice paddies.

Benefit-relevant indicators

The DNDC model outputs from the 22-year simulation, were processed to isolate stakeholder-specific and benefit-relevant indicators that carry value when presented to the stakeholders.

The ecosystem of interest was defined as rice paddies under anaerobic conditions and the beneficiaries (stakeholders) were identified as the rice farmers and the local community of Beaumont, Texas. The attributes of the rice paddy ecosystem valued by the beneficiaries included rice production capacity, nutrient recycling and soil conservation. These attributes were directly linked to ecosystem services such as regulation of grain production, methane emission, N retention and soil organic C storage. The DNDC model was applied to quantify changes each indicator through model simulations under varying management conditions.

The measure of both benefits and disservices were connected to social outcomes based on what is valued by the farmers, if there is a demand for the service, and how a service may be used or enjoyed (Olander, et al. 2018). This link was established using the 'service causal chain' as described in Fig. 4.2.

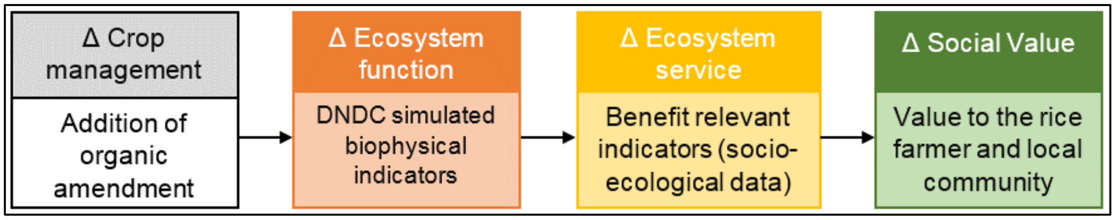


Figure 4.2. Ecosystem service causal chain to show the movement of a crop management action on the soil ecology (in orange), the soil ecosystem services (in yellow) and the change in benefit-relevant indicators proximal to the social welfare of rice-planting stake.

The sequence of cause and effect begins at the point of management change, which in this study, is the addition of an organic straw N amendment with urea. The stress or action then follows a series of changes in the soil ecosystem stocks, and the flow of soil functions and services. Changes in amount of soil stocks and the flow of soil functions are grouped as ecological changes and are represented in orange, the services directly flowing from these soil stocks and functions are grouped as soil ecosystem services and are represented in yellow. Finally, the ecosystem services that have a direct influence on human welfare flowing from the soil are coined to be benefit-relevant indicators and are represented in green (Fig. 4.2).

Direct method

A direct cost evaluation method was used to calculate total gained yield, and the global warming potentials of the two treatments. Total grain yields were compared between urea and urea with organic straw amendment from a single year and a 22-year DNDC simulation. Market price comparisons of adding straw amendment to urea fertilized rice could be misleading, since it does not represent the life cycle analysis for both the amendment and urea. Therefore, the benefit-relevant indicator, grain yield, was assessed based on difference in mass of grain observed instead of total income from sale.

Global warming potential of soil under the two treatments were calculated by converting methane and nitrous oxide emission into carbon dioxide equivalent values. The conversion was made using global warming potentials of 28 and 265 respectively for comparison at a 100-year scale (EPA- Greenhouse Gas Emissions). EPA provides an estimated range for the global warming potentials of methane and nitrous oxide, we used the lower end of the range for convenience. Similarly, net soil organic C added to the soil was also converted to equivalent carbon dioxide values by using percentage of C in the molecular weight of the gas. Total carbon dioxide equivalent emissions were calculated for comparison, by adding equivalent emissions of the greenhouse gases and reducing the equivalent C stored in the soil.

The integrated method

For analysis of any complex agronomic system including rice production, it is necessary to integrate social, economic and ecological concepts (ISEEC) to evaluate changes in management practices (Gunderson, 2001; Fox et al., 2009). The model was used to facilitate the holistic approach that considers the effects of management changes over time (Fig. 4.3). Integration of soil ES assessment between service types over time provided better information for decision making. This may further lead to re-organization and/or emergence of new states of the socio-economic and natural capitals as well as function (Flood, 2002).

The ISEEC holistic model (Fig. 4.3) is a framework that was initially developed for rangeland sustainability (Fox et al., 2009). It was used to evaluate four states, the initial and final soil natural capital, outlined in green, and the initial and final socio-economic capital, outlined in blue (Fig.4.3). Links between the ecological and social subsystems was established using ecosystem services flowing between two systems and are outlined in purple. The final states of an agricultural system including both biological and socio-economic stocks and processes, were studied for any changes after management alterations over time.

We examined the states pre-and-post rice production using both single-year and 22-year DNDC model simulations. Changes from the initial to final state were determined by quantifying soil properties for natural capital assessment (Dominati, et al 2010) and benefit-relevant indicators for the socio-economic capital assessment (Olander, et al 2018). Quantification of soil ES that were benefit relevant was used to link soil and socio-economical functions that define the state of our rice production ecosystem.

Soil natural capital and socio-economic capital transition from initial to final states through changes in composition and functions. In our case study, the rice soil natural capital (Fig. 4.3), includes simulated soil components that are important in regulating soil ES flow from rice paddies, such as soil organic C content, soil residual N concentration, and soil redox potential.

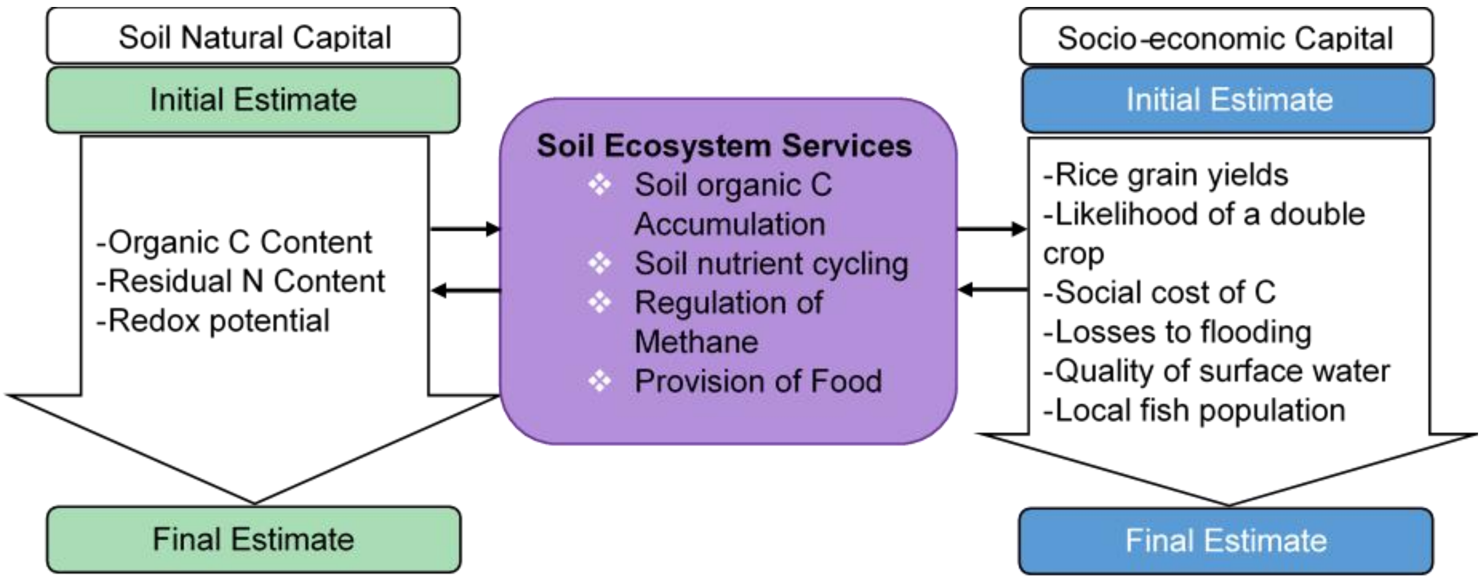


Figure 4.3. Integrated social, economic and ecological concept (ISEEC) to measure and link changes via soil ecosystem services (in purple) between rice soil natural capital (in green) and socio-economic capital (in blue) of stakeholders in rice farming across a temporal scale.

Soil is the second largest pool of sequestered C in the global C cycle, and soil stored C plays an important role in mitigating anthropogenic emissions of C (Stockmann et al., 2013). Similarly, soil N also plays an important role by influencing the C and N ratio available to the soil microbial population which in turn affects the flow of soil functions such as rate of N mineralization and nutrient cycling of added fertilizer or organic amendment (Masunga et al., 2016; Ponnamperuma, 1984). Furthermore, under saturated conditions typical of rice farms, there are drastic changes in the soil redox potential as oxygen depletes (Achtnich, et al. 1995; Ponnamperuma, 1984). The redox potential of the paddy soils is included as a soil natural capital as it is a key factor that determines rates of soil functions of reduction reactions and the soil service of regulating methane emissions (Achtnich, et al. 1995; Ponnamperuma, 1984).

Finally, biophysical changes within a rice paddy ecosystem were linked to socio-economic capital by describing linking or benefit-relevant indicators (Fig. 4.3). The causal links created through soil ES assessment (Olander et al., 2018) culminated in evaluation of the socio-economic capital invested by the farmers and their community in rice production. These include, rice grain yield, fertility of the field, and the local climate as governed by global climate changes.

Results and Discussion

Simulated grain yield

Addition of straw amendment with urea resulted in lower grain yields in all planting dates. Grain yields reduced with addition of amendment by 15, 8 and 13% from early to late planting dates, respectively (Fig. 4.4).

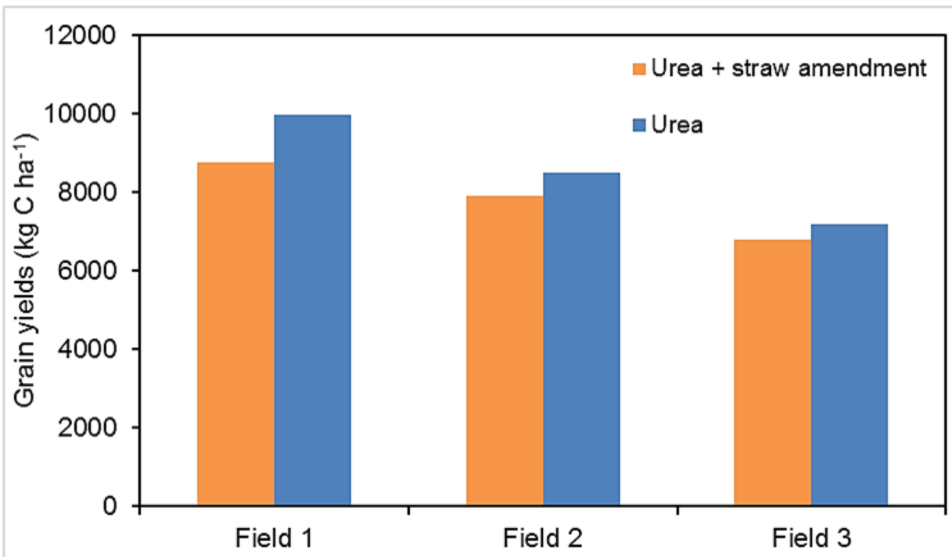


Figure 4.4. Comparison of total grain yield from fields with varying planting dates with organic amendment and conventional fertilizer.

For the 22-year simulation, Student t-tests showed no significant difference between grain yields from urea and urea with straw amendment fields for years with low and high ET to precipitation ratios (p=0.2 and p=0.07), but a significant difference in grain yields between treatments for medium precipitation years (p=0.0005) did exist. Grain yield produced per year was lower by 5.4% for the straw amendment treatment in years with medium ET to precipitation ratios. The yield decreased from adding the organic straw amendment to urea, was caused by the extra amount of C introduced from the amendment, change in soil C to N ratio, and mineralization of the type of N source added. Focused plant breeding in combination with management alternatives may be able to improve cereal yields in organic farming conditions (Mader et al., 2002).

Analysis of variance between years with low, medium and high precipitation regiments also showed an effect on grain yield for organic ($p = 0.02$) and conventional fertilized ($p = 0.03$) systems (Fig. 4.5). Least Significant Difference (LSD) mean separation analysis showed significant differences between yields from low and high, and medium and high precipitation years. No significant differences were seen between low and medium precipitation years. Lower precipitation years are associated with higher temperatures which has a negative effect on rice yields and a reciprocate effect on seasonal methane emissions (Matsui, et al. 1997).

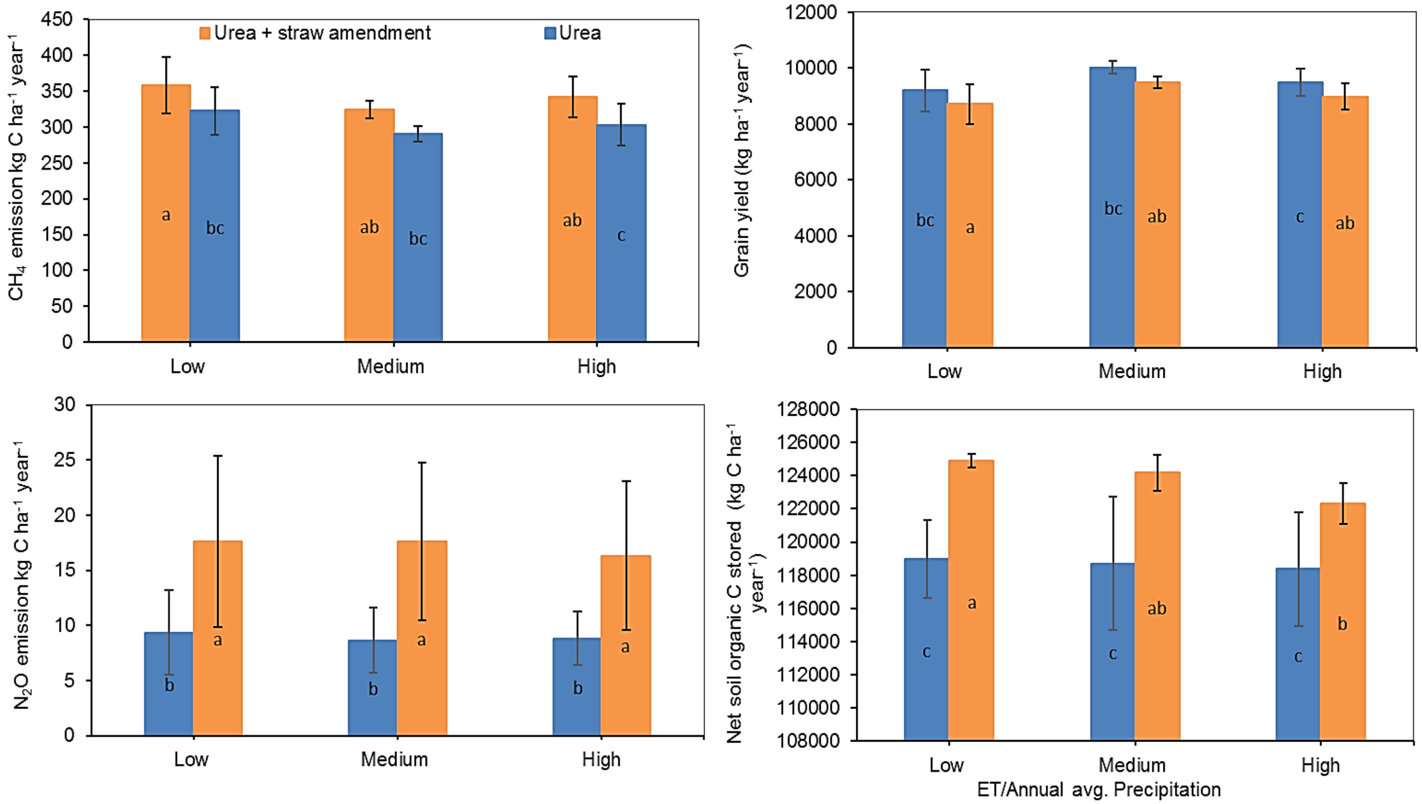


Figure 4.5. Annual average methane emission, grain yield, soil organic C, and nitrous oxide emissions between organic amendment and conventional fertilizer with low, medium, and high precipitation years based on their ET-to- precipitation ratios.

Simulated soil C cycling

Addition of organic straw amendment with urea resulted in greater soil methane and carbon dioxide emissions and a greater final soil organic C content. Content of C lost as carbon dioxide and methane are the two largest C fluxes released by the soil into the atmosphere and interact with the final amount of organic C stored in the soil.

Straw amendment treatment resulted in increased seasonal methane emissions between treatments by 28, 41 and 15.2% from early to late planting dates, respectively (Fig. 4.6). Later planting dates had lower emissions because of shorter flood periods. This phenomenon can be accounted for by analyzing the concentration of various electron acceptors in the saturated system. Saturated soils typically start producing methane gas only after ten days to two weeks when other available electron acceptors (O_2, NO_3^-, Fe^{3+}, Mn^{4+}, and SO_4^{2-}) are completely consumed (Achtnich, et al. 1995; Ponnamperuma, 1984).

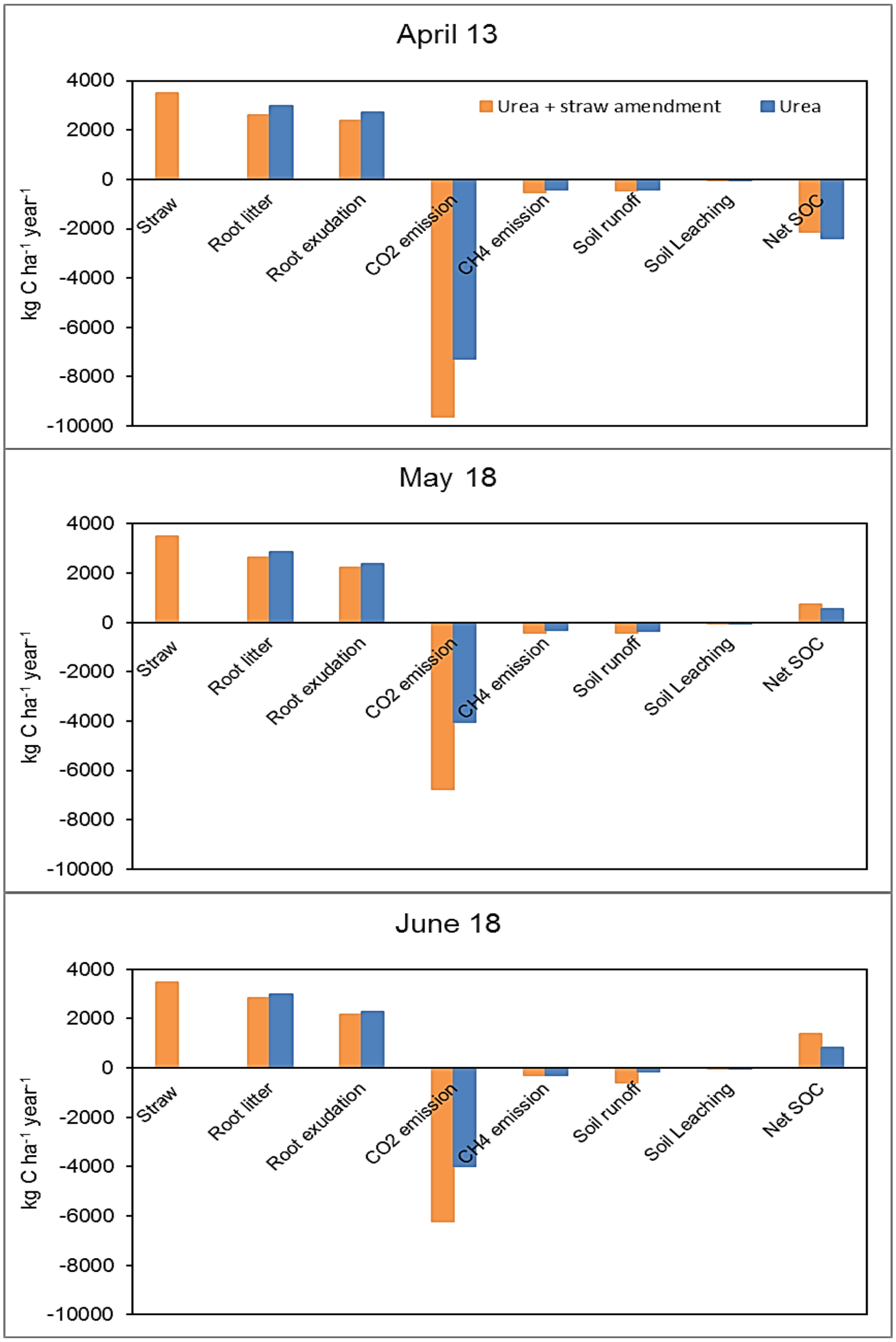


Figure 4.6. Mass balance of soil organic C in a paddy soils and net C flux comparisons between urea and urea with organic straw amendment in fields with varying planting dates (positive and negative values indicate net flux into the soil and out of the soil, respectively).

The 22-year simulation of seasonal gas emissions showed strong effect from the straw amendment ($p = 0.0003$) on methane production. The difference in seasonal methane levels between years was also significant ($p = 0.02$).

Analysis of variance between low, medium and high precipitation years based on their ET-to-precipitation ratios also showed an effect on methane emissions ($p = 0.03$) (Fig. 4.5). Highest emissions were observed during low precipitation years which also had a higher average temperature during growing season in comparison to high and medium precipitation years (Fig. 4.5). DNDC methane simulations are shown to be highly sensitive to changes in temperature (Matsui, et al. 1997; Sass and Fisher 1991). Decline in precipitation in combination with higher temperature has previously been reported to increase methane emissions in rice production (Daulat and Clymo, 1998; Chin et al., 1999). Increase in temperature was associated with increase in number of methanogens in saturated rice paddy soils, and enhanced micronutrient uptake and methanogenic activity (Chin et al., 1999; Seneesrisakul et al., 2018).

The model calculates the values of individual fluxes of C being added or removed from the soil, which allowed us to do a mass balance analysis of C in the crop growing season (Fig. 4.6). DNDC has been successfully used to estimate soil organic C storage in most croplands of mainland China by accounting for C input sources such as manure, crop residue and root exudation, and output sources such as soil leaching, carbon dioxide and methane flux (Tang, 2006). Our analysis showed that addition of urea with straw amendment increased the total input of C into the soil and had a net positive C flux resulting in greater C storage in comparison to urea alone (Fig. 4.6).

Significant increase in carbon dioxide and methane emissions to the atmosphere negated the amount of added organic amendment and lowered net C flux. Fields with later planting dates had shorter growing periods and lower emissions of carbon dioxide and methane, which in turn resulted in a greater net C storage (Fig. 4.6). Increased amounts of soil C addition was observed with later planting dates because of lower methane emission leaving the soil. No significant differences were observed in soil organic C between low, medium and high precipitation years (Fig. 4.5).

Average annual soil organic C simulations were plotted for both urea with straw amendment and urea alone (Fig. 4.7). The soil organic C curves for both treatments have an asymptotic trend, however, the increase in soil organic C reached a maximum point when the simulation was extended for two more years (Fig. 4.8).

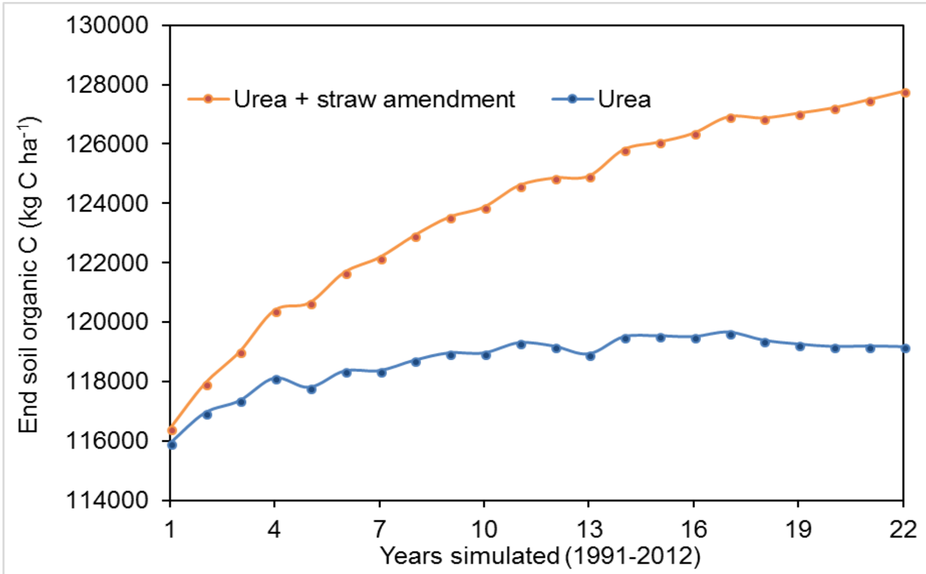


Figure 4.7. Model simulated soil organic C trends over 22 years for conventional fertilizer and organic amendment treatments.

A mass balance analysis of soil organic C showed a loss of soil organic C for the conventional fertilizer treatment (- 36 kg C ha^{-1} year^{-1}) and addition of soil organic C for the organic amendment treatment (+282 kg C ha^{-1} year^{-1}) (Fig. 4.8). The additional source of organic C from organic amendment compensated for other C fluxes that remove C from the soils and created a net positive flux of C added into the soil in comparison to conventional fertilizer. Soil C sequestration was observed after the 22-year of simulation and not in the single year, because the effect of adding an organic amendment to soil organic C content is slow and would occur only through long-term studies (Diacono and Montemurro, 2009).

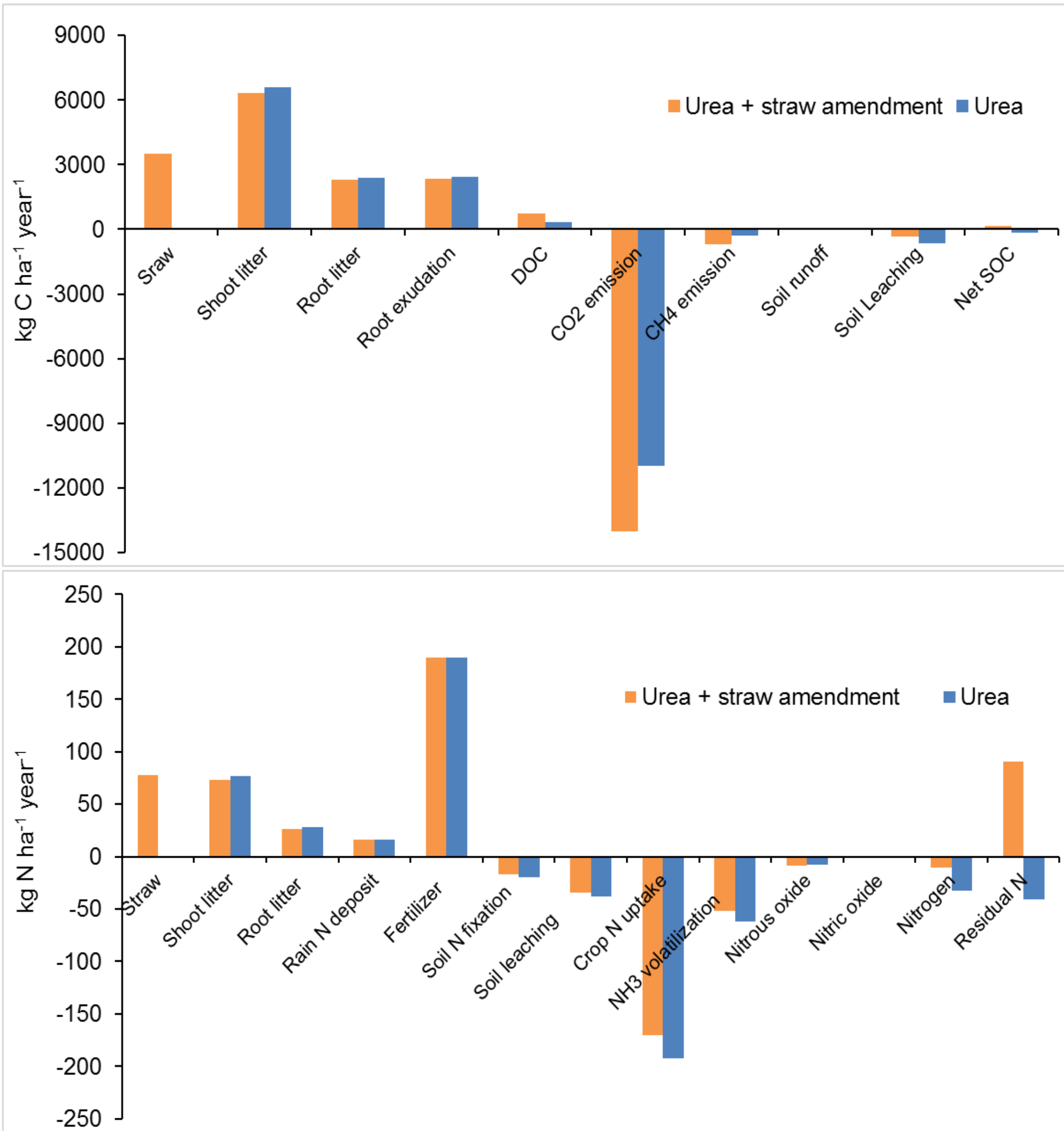


Figure 4.8. Mass balance of annual soil organic C sources and sinks in a rice paddy soil ecosystem and net C flux comparisons between conventional fertilizer and organic amendment averaged from 22 years simulated data.

Simulated soil N cycling

Addition of organic matter increased residual soil organic N content and soil nitrous oxide emissions in all planting dates. As an additional source of organic N, the organic amendment treatment increased residual N content by less than 1% for all three planting dates.

For the 22-year simulation, analysis of variance of the final soil organic N contents showed strong effect from adding organic amendment ($p<0.0001$). Annual end soil organic N from organic amendment was significantly higher than conventional fertilizer and both showed asymptotic behavior when simulated for 22 years (Fig. 4.9). However, comparisons based on ET-to-precipitation categories, showed no correlations with nitrous oxide emissions (Fig. 4.5).

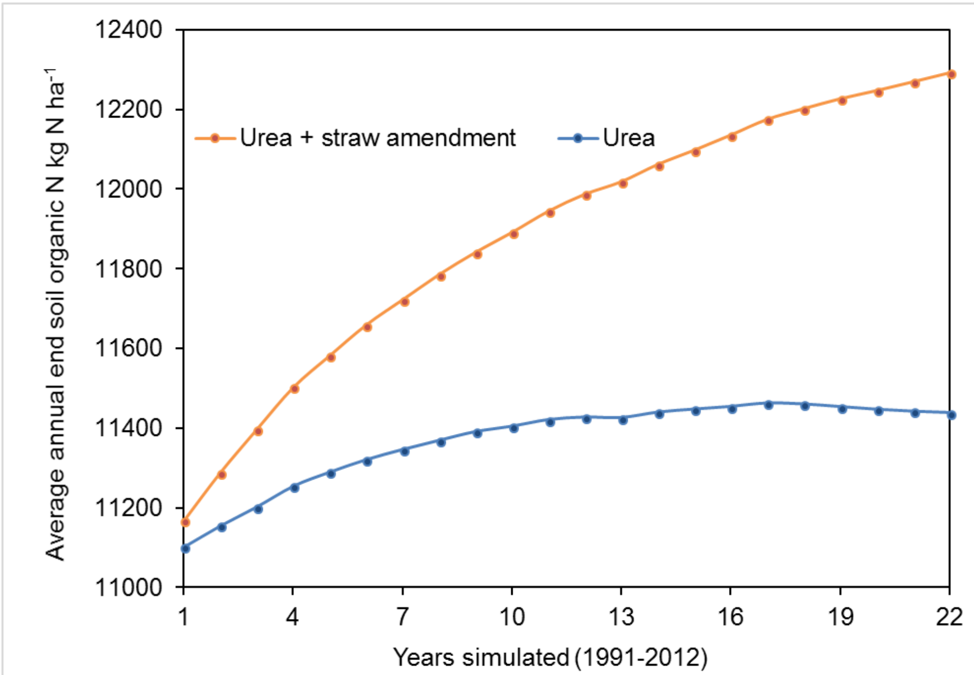


Figure 4.9. Model simulated soil organic N trends over 22 years for conventional fertilizer and organic amendment treatments.

DNDC simulated values of various fluxes of addition and removal of soil organic N were used to analyze soil N cycle. Residual soil N at the end of each rice production year was calculated using mass balance analysis of soil organic N (Fig. 4.8). Addition of organic amendment resulted in soil organic N augmentation by about 7.5 kg N ha^{-1} year^{-1}, whereas application of only conventional fertilizer resulted in loss of soil organic N by about 19 kg N ha^{-1} year^{-1}. Our results were consistent with previous studies that demonstrate increase in soil organic N storage for future crops in fields amended with organic fertilizers (Diacono and Montemurro, 2009).

On the other hand, organic amendment significantly increased soil nitrous oxide emission over 22 years (Fig. 4.5). Addition of organic amendment increased amounts of dissolved soil organic C which in turn simulated production of nitrous oxide before saturation and N_2 near saturated conditions (Bronson et al., 1993).

Ecosystem services assessment

Biophysical analysis with DNDC model simulations helped delineate a series of changes in soil properties. Our study showed adding organic amendment led to increase in soil stocks such as total soil organic C, and functions such as mineralization rates of the soil organic C, and production of dissolved organic C and C substrates involved in soil redox reactions that lead to subsequent variations in soil E_h. Changes in soil organic C capital was observed as a direct result of adding organic amendment. On the other hand, organic amendment increased the field methane emissions, and subsequently, the global warming potential of rice paddies. The linking indicator of changes in soil atmospheric C emissions for policy makers are changes in the social cost of carbon dioxide from rice fields. For farmers, the linking indicators are the effects of climate change in rice grain yields (lower yields leading to reduced local income). For the local community and farmers, the linking indicators are the vulnerability of soils with low organic C to flood events (Fig. 4.10).

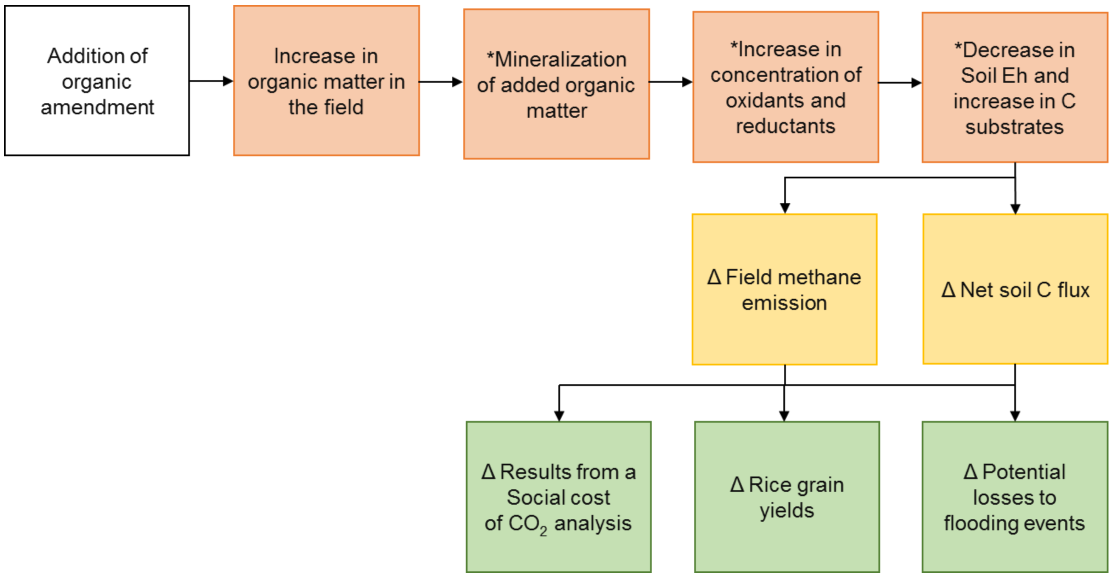


Figure 4.10. Ecosystem service causal chain shows how addition of organic amendment leads to the ecological changes (in orange) in soil C mineralization and soil Eh leading to changes in soil methane regulating services (in yellow), from which benefit-relevant indicators (in green) were chosen for their proximity to human well-being.
*represents soil functions flowing from soil stocks

The EPA and other federal agencies use integrated assessment models to measure social cost of carbon dioxide (SC-CO_2 EPA, 2016). The SC-CO_2 provides a comprehensive estimate of climate change damages on human welfare by considering changes in net agricultural productivity, property damages from increased flood risk, and cost of changes to the energy system (Newbold et al., 2010). This model is popularly used to value the climate impacts of new regulations by including long-term damage done by a ton of carbon dioxide emitted in a given year (Tol, 2008). The parameters considered in the measurement of the SC-CO_2 are consistent with our benefit-relevant indicator approach and can be considered important information to be relayed to policy makers. Results from the SC-CO_2 analysis provide present and future values of the marginal social damages from increased atmospheric greenhouse gas input.

Similarly, the individual components assessed within the SC-CO_2 are useful candidates to serve as benefit-relevant indicators of greenhouse gas emissions to local people. Adding organic matter increased greenhouse gas emissions, which will influence climate and can cause negative global effects. Changes in temperature and intensity of precipitation highly influence the soil moisture profile, which is an important soil property in agriculture since soil-stored water provides water to plants between rainfall events (Lobell and Gourdji, 2012). Also, studies have shown reductions in rice yield from increasing average air temperatures in combination with increasing atmospheric CO_2 (Matsui, et al. 1997). Although variations were observed among cultivars, rice pollen viability and production are known to decline as daytime maximum temperature exceeds 33°C (Hatfield and Prueger, 2015). Similarly, meta-analysis for ozone influence on rice showed significant reduction in net photosynthesis, biomass, grain number and mass (Hatfield, et al. 2011). Therefore, impacts of climate change on rice productivity is a good indicator that is relevant to the well-being of rice farmers.

Although organic amendments increased atmospheric greenhouse gas emission, the amendment also improved soil organic C storage. Coastal erosion, inundation and episodic flooding are recorded to have increased both in intensity and frequency as a direct result of global warming and increasing sea levels (McGranahan et al., 2007). The US Southeast coastal area which includes most of Texas rice production sites, is highly vulnerable to flooding from sea level rise due to its low relief. The coastal flood risk assessment database shows that the eastern coastline of Texas and Louisiana fall under high risk zones (Gornitz et al., 1994). Since greenhouse gas emissions are directly linked to increasing sea levels communication of estimated potential losses from coastal flooding is a relevant indicator for mitigating such emissions.

In contrast to increasing soil atmospheric C, soil C storage provides ecosystem services that are benefit-relevant to local community (McBratney, et al., 2014). Increased organic C input into the soil is known to improve soil structure and function. Examples include increases in soil cation exchange capacity, and improved soil structure resulting in increased infiltration rates or water capturing function of soil (Dominati, et al. 2010). These are common soil properties that, in turn, help insure long-term productivity of crops and can be reported in terms of increased or lower temporal volatility of grain yields. Improvement of soil structure also enhances a landscape’s capacity to capture and store water, which results in reduced runoff and soil erosion (Stockmann, et al. 2013). Implications of reduced erosion that are relevant to local stakeholders can be reported in terms of biodiversity of aquatic animals available to fishermen (Olander, et al. 2018). Finally, increased soil organic matter and stronger soil structure will reduce sediments and total suspended solids in local waterways and reservoirs, which will improve ground water recharge, control pollution loads from farms and upkeep local aesthetics (Dominati, et al. 2010). Therefore, the benefit-relevant indicators of increased C storage in rice fields for the farmers are the long-term sustainability of rice yields, and for the community, are the maintenance of aquatic biodiversity by mitigating eutrophication, and of the local waterway aesthetics by reducing total suspended solids added through soil erosion (Fig. 4.11).

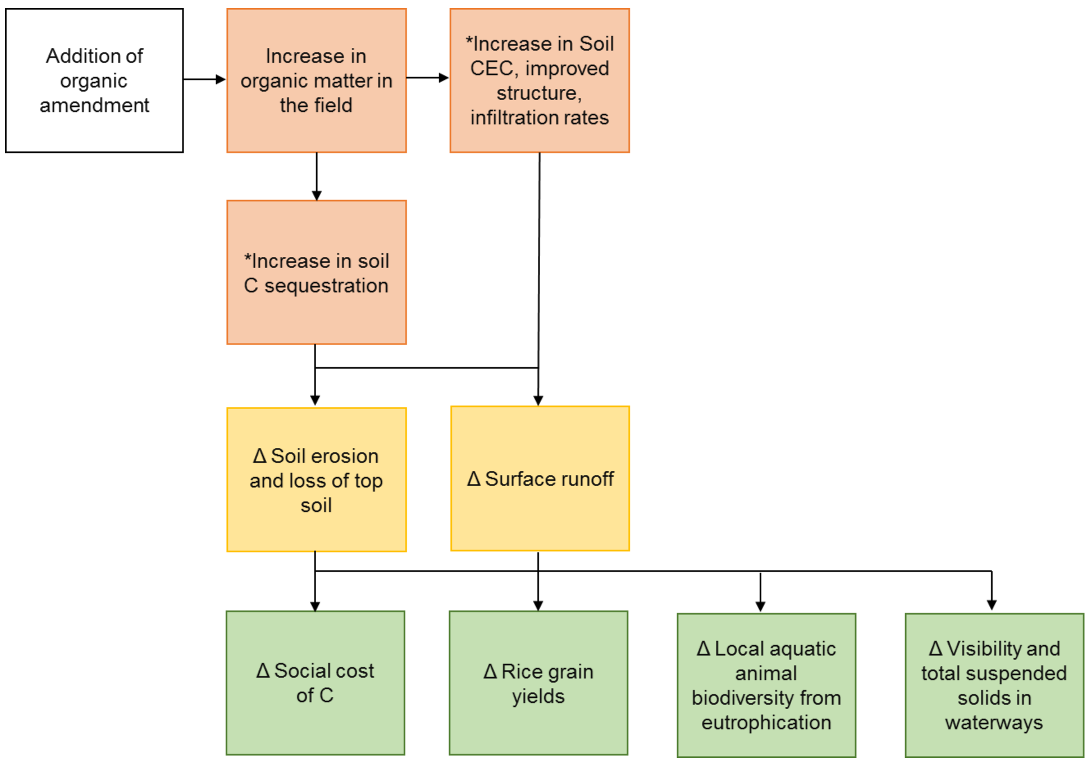


Figure 4.11. Ecosystem service causal chain shows how addition of organic amendment leads to the ecological changes (in orange) in soil C mineralization and soil Eh leading to changes in soil erosion and runoff (in yellow), from which benefit-relevant indicators (in green) were chosen for their proximity to human well-being.
*represents soil functions flowing from soil stocks

Another major effect that observed from converting to an organic amended system was the change in soil functions such as soil microbial activity and production of plant available N. The overall effect was accumulation of residual N in the soil natural capital which can be available for cropping a second double crop such as soybeans, which is a common double cropping system with rice in southeast Texas (Jones and Belmar, 1989; Belmar, Jones and Starr, 1987; Havlin, et al. 1990; Spurlock, et al. 2016). Previous studies have shown that historically organic farmers were able to recommend N input rates based on anticipated residual N (Bulluck, et al. 2002). Therefore, residual N in soils can be relayed to farmers in terms of reduced N input for a second double crop or for the next rice planting, and to local people in terms of aquatic biodiversity changes offsets from mitigation of eutrophication (Fig. 4.12). However, adding organic material increased soil nitrous oxide emissions, which have a very high greenhouse gas effect and will increase the social cost and global warming potential of organic rice farming.

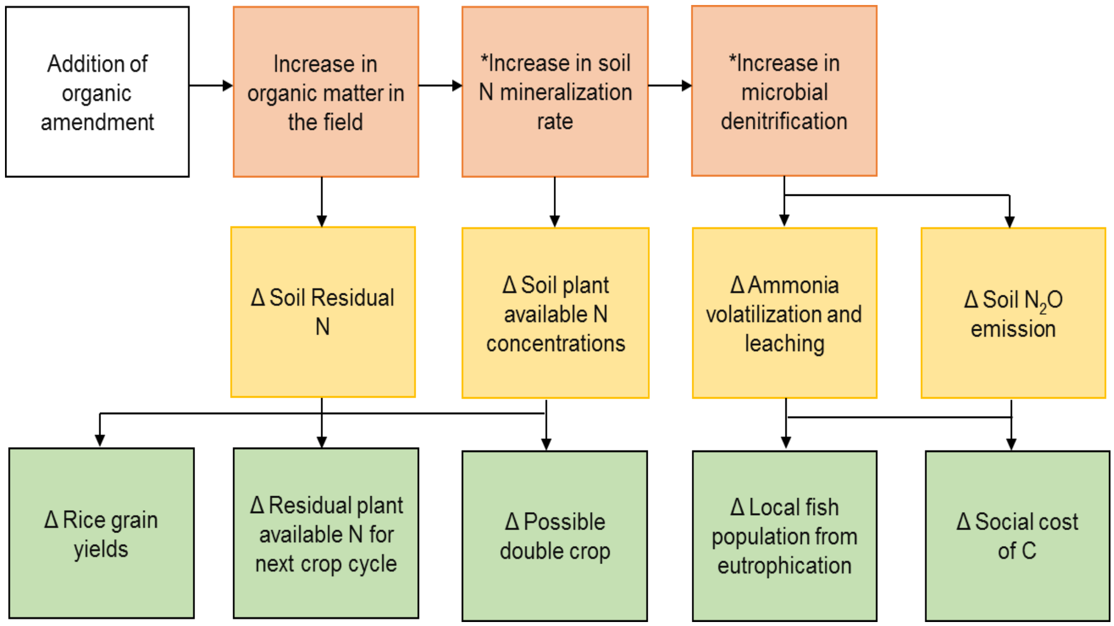


Figure 4.12. Ecosystem service causal chain shows how addition of organic amendment leads to the ecological changes (in orange) in soil N mineralization and microbial denitrification rates leading to changes in soil nitrous oxide regulating services (in yellow), from which benefit-relevant indicators (in green) were chosen for their proximity to human well-being.
*represents soil functions flowing from soil stocks

There are both positive and negative effects of adding organic amendment in rice production. Increase in methane and nitrous oxide emissions will have a high global climate impact but increased soil residual C and N content will improve soil security. Further discussion on the tradeoffs of fertilizer alterations in rice production are presented using the integrated method.

The integrated analysis

Overall effects of the organic amended N fertilizer treatment from varying planting dates and from the 22-year simulation have been summarized in Table 4.4. Analysis of the effects include quantification of soil ecosystem services including, rice grain yields, methane regulation, soil organic C storage and soil organic N storage. Benefit-relevant indicators and their linking natural and socio-economic capitals, as previously established (Fig. 4.10, 4.11 and 4.12), were predicted based quantified soil ecosystem services Table 4.5. By adding values to each component of the model, we demonstrated how a natural or anthropogenic disturbance might concurrently influence the states of both the capitals invested in a system. Global warming potential and its translation to equivalent amount of emissions observed by driving an average of 15,000 miles on a 2015 Ford F150 is shown in Fig.4.13.

Table 4.4. Quantification of soil ES (rice grain yield, methane regulation, soil organic C storage and soil residual N concentration) reported in percentage changes gained (+) or lost (-) for organic amended fields in comparison to fields with urea treatment.

One-year simulations				
Planting dates	Rice grain yield	Methane regulation	Net soil organic C concentration	Residual N concentration
April 13	(-15%)	(+27.5%)	(+0.67%)	(+0.64%)
May 18	(-8%)	(+41%)	(+0.87%)	(+0.65%)
June 18	(-13%)	(+15.2)	(+0.95%)	(+0.66%)
22 years simulation				
ET/Precipitation Ratio	Rice grain yield	Methane regulation	Net soil organic C concentration	Residual N concentration
Low precipitation	(-5.4%)	(+11%)	(+4.9%)	(+5.2%)
Medium precipitation	(-5.4%)	(+12%)	(+4.6%)	(+4.8%)
High precipitation	(-5.4%)	(+13.8%)	(+4.3%)	(+4.5%)
Overall	(-5.4%)	(+13.3%)	(+7.2%)	(+7.5%)

Table 4.5. Final status analysis of the natural and socio-economic capital for organic amended and conventional fertilized rice farms by using the benefit-relevant indicators.

Natural capital	Urea + organic straw amendment	Urea
Soil organic C	282 kg C ha^{-1} $year^{-1}$ sequestered into the soil	22.4 kg C ha^{-1} $year^{-1}$ Soil organic C lost from the soil
Soil methane flux	342 kg methane emitted ha^{-1} $year^{-1}$	306 kg methane emitted ha^{-1} $year^{-1}$
Net residual N	90.53 kg N ha^{-1} $year^{-1}$ added into the soil	42.2 kg N ha^{-1} $year^{-1}$ lost from the soil
Socio-economic capital		
Soil productivity	Yields predicted to increase from 9037 kg C ha^{-1} $year^{-1}$ in long-term	Average grain yield of 9553 kg N ha^{-1} $year^{-1}$ predicted to decline in long-term
Climate changes	Higher global warming potential with 27 tons of CO_2 produced	Lower global warming potential, 21 tons of CO_2 produced
Soil structural integrity	Improved structure leads to protection against flood risk, property damage and erosion	Poor structure increased vulnerability to flood loss and erosion of up to 28,000 kg soil ha^{-1} $year^{-}$
Quality of local surface water	Minimum total suspended solids, low nutrient loading and high biodiversity Preserved aesthetics and fishing opportunities	Increased total suspended solids, high nutrient loading and hampered biodiversity Change in surface water and reservoir clarity and reduced fish population
Production opportunity	Residual soil N can support a second crop Lower investment for fertilizing a second crop	Loss of soil organic N requires additional N for a second crop Additional investment required to support a second crop

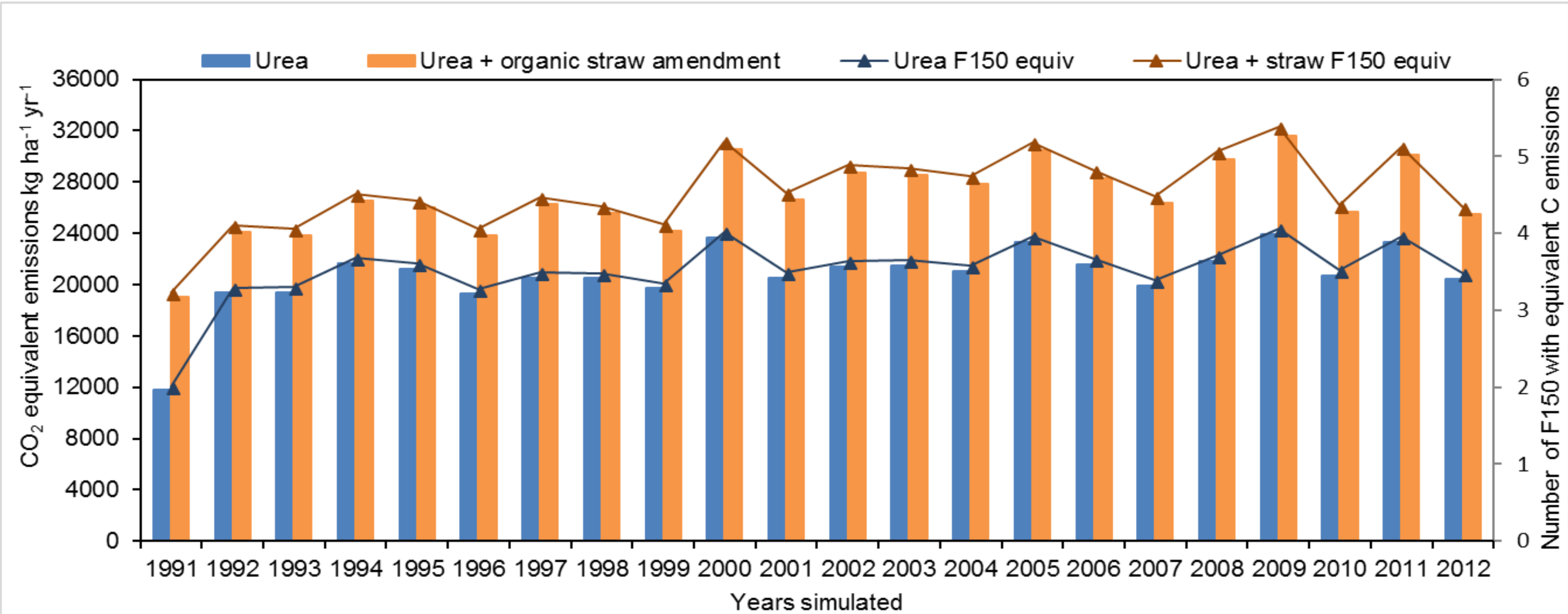


Figure 4.13. Global warming potential of rice farms under urea and urea with organic straw amendment calculated using net emissions of carbon dioxide from 22year DNDC simulation and their equivalent number of Ford 2015 F150s for an annual average of 15,000 miles.

Post calibration, model simulations were conducted for 22 years to predict the long-term effects of adding organic amendment. Soil organic C and N content increased by less than 1% in one year and up to 7% greater storage of organic C and N was observed for fields with organic amendment in 22 years. Previous studies have shown that the effects of adding organic materials on the soil properties of a cropland intensify over time and long-term field studies have shown increase on soil organic C and N by up to 90% in comparison to chemical fertilizer treatments (Diacono and Montemurro, 2009).

However, organic amendment increased methane and nitrous oxide emissions and the overall global warming potential from rice fields (Fig. 4.13). Thus, a high impact on the social cost of C occurred because of methane and nitrous oxide warming potentials are 28-fold higher and 263-fold higher than carbon dioxide in a 100-year time scale respectively (Boucher et al., 2009; Shine et al., 2005). Therefore, when organic amendment is added to the field, increased net carbon dioxide emissions is equal to driving 4 to 5 Ford 2015 F150s for an annual average of 15,000 miles. Similarly, urea fertilized fields emit greenhouse gases equivalent to 3 to 4 Ford 2015 F150s for an annual average of 15,000 miles. On average, the increase in emissions from adding organic amendment to conventional, urea-only, rice farming was equal to driving an additional two F150 trucks, annually.

Global warming is measured in a 100-year scale (Lal, 2004) and previous studies have shown terrestrial C pools to act as a sink for as much as 50 ppm of atmospheric carbon dioxide for 100 to 150 years with improved management (Lal, 2010).

Furthermore, additional benefits from increasing soil organic C such as advancing global food security, soil water retention capacity and nutrient cycling must be considered as a tradeoff when evaluating global warming potential. Increase in nitrous oxide emission, which has a much higher global warming potential, with organic amendment resulted in higher warming effects. Mitigating emission through alternative routes such as change in water regiment to intermittent flooding or no flooding would prove beneficial to avoid this problem (Zhang et al., 2009; Itakura et al., 2013).

Impacts of global warming and climate change from field greenhouse gas emissions can be related to rice farmers through provisioning soil ES and final grain yields. A previous study from the International Rice Research Institute, Philippines, reported that grain yield in rice declined by 10% for each 1ºC increase in growing-season minimum temperature in dry seasons (Peng et al., 2004). Simulation of annual C storage over 22 years showed an increase of 282 kg C ha^{-1} $year^{-1}$ and had a net C sequestration effect with organic amendment. On the contrary, conventionally-fertilized rice farms had lower methane and nitrous oxide emissions from the field (Table 4.5) but had lost soil organic C at the rate of -22.4 kg C ha^{-1} $year^{-1}$.

A previous study on the effects of incorporating manure mulch versus no manure showed a difference of up to 4 cm of surface runoff after a rainfall event in Iowa (Pimentel et al., 1995). Furthermore, reduced runoff and erosion control can help offset an estimated 28,000 kg ha^{-1} $year^{-1}$ of soil lost from croplands in Texas (USDA- 2011 RCA Appraisal). Furthermore, the USDA has been providing incentives for environmental markets that trade in water quality credits that can be earned by landowners through reducing nutrient runoff and sediment loads. Credits earned can be traded to water utilities and industrial polluters under the regulations of the Clean Water Act (USDA- 2011 RCA Appraisal).

Adding organic amendment is predicted to increase grain yield, protect against flood losses and support double cropping systems, but will also cause an increased global warming effect through higher methane and nitrous oxide emissions. It will be up to the stakeholders to decide the viability of the tradeoff between higher atmospheric greenhouse gas emissions and improved soil structure and function when adopting organic amendments for rice cultivation. Decisions will be dictated by the change in the most influential benefit-relevant indicator depending on people's individual preference.

Conclusion

DNDC model simulations were calibrated using a previously published field study conducted by Sass and Fisher (1991) that measured and compared field-emitted methane concentrations between conventional fertilizer and organic amendment, from three rice fields located within the Texas A&M University's AgriLife Research Extension Center in Beaumont, Texas. Data from both single-year and 22-year simulations showed a significant effect of organic amendment on methane emissions, nitrous oxide emissions, carbon dioxide equivalent emissions, grain yield, soil organic C and soil organic N.

Application of organic amendment resulted in increase in seasonal methane emissions; however; mass balance calculations showed a positive net soil organic C in comparison to conventional fertilizer. Clearly, adding organic amendment has both positive soil health effects from C and N storage in the soil and negative effects on the atmosphere and global climate through greater greenhouse gas emissions.

The DNDC model simulations did not consider the variation in soil physical and chemical properties caused by accumulating soil organic C from the added organic amendment. The model assumes that soil structure, infiltration rate, cation exchange capacity, runoff, and grain yield were not sensitive to additional soil organic C accumulated over the years from the additional organic amendment. This disregard for change in soil condition prevents a thorough assessment of how organic fertilized rice systems truly change soil functions. However, it does record the steady increase in soil organic C and field methane emissions and provides a good starting point to quantify the changes in soil ES from changes in fertilizer management in rice farming.

Biophysical effects of organic amendment were translated in terms of soil ES to ease the communication to the stakeholders of rice production. The natural capital invested in rice farming was valued using the model simulated changes between management practices. Individual soil ES flowing from the capital were assessed using a causal chain approach to identify linking indicators or benefit-relevant indicators. Therefore, changes in soil organic C and soil greenhouse gas emissions were linked to social cost of C, rice grain yields, potential property losses to flooding events, suspended solid visible in local waterways and their aquatic biodiversity. Similarly, changes in soil organic N was linked to reduction in N input for next crop cycle, capacity of soil to sustain a double crop and evasion of eutrophication of local waterways from nutrient loading.

Associating the value of soil ES in terms of dollars can be misleading and can over or underestimate the effects of changing management practices on the natural capital and its subsequent effects on the socio-economic capital. This project was designed to combine the use of DNDC a biogeochemical model with benefit-relevant indicators and ISEEC framework. The combination allowed for interpretation of soil physical and chemical changes from adopting alternative management in rice production in terms of social implications that are considered important by the farmers and their local community.

Although interpretations from model simulations can be useful for their detailed output, the error associated with them and their accuracy are always difficult to determine. Field validations are not true representations of predictions beyond measured parameters and years. Furthermore, this study showed that DNDC simulations are not sensitive to changes in soil components affected by changing fertilizer management. Therefore, true interpretations of transitioning to organic amended rice production can be made only through model improvements to include cumulative effects from changes in soil biological, chemical and physical properties.

CHAPTER V

SUMMARY OF DNDC APPLICATION TO SIMULATION OF SUSTAINABLE RICE PRODUCTION AND QUANTIFICATION OF SOIL ECOSYSTEM SERVICES

Summary and Conclusion

DNDC model has been extensively modified since its development to better simulate anaerobic conditions under which rice is typically produced. Its application towards the delineation of sustainable rice production in the past has been mainly focused on analyzing the effects of changing water management. This project used the model to simulate the effects of switching from urea fertilized to an organic fertilizer system as well as the effects of adding an organic straw amendment to ureafertilized rice production systems in Beaumont, Texas. Greenhouse experimental data and previously published field data from Sass and Fisher (1991) were used to calibrate the model.

The model-simulated results showed discrepancies at a temporal scale with daily methane emissions that had to be rectified using experimental data of concentrations of electron acceptors in the soil under saturated conditions. Final validation with the greenhouse pot experiment showed high correlation for grain yields, daily methane emissions, and dissolved soil organic C concentrations.

Calibration of the model with published field data was used to make a 22-year simulation, and the simulated data of rice grain yields, soil C cycling, and soil N cycling were used to quantify soil ecosystem services. Model simulated data provided a base for calculation of global warming potential and its equivalent carbon dioxide emissions for urea-fertilized and organic amended systems.

Detailed soil ecosystem services analysis showed that the model lacked a holistic algorithm to accounted for changes in soil physical and biological properties. A true picture of the benefits of adding organic amendment to soil CEC, soil structure, infiltration rate, microbial activities, and other physical and biological soil functions and stocks were lacking in this analysis. Therefore, there is room for improvement in the DNDC model algorithms and input features that would result in improved simulations of changes in the soil ecosystem brought about by management approaches.

REFERENCES

Achtnich, C., F. Bak, and R. Conrad. 1995. Competition for electron donors among nitrate reducers, ferric iron reducers, sulfate reducers, and methanogens in anoxic paddy soil. Biology and Fertility of Soils 19:65-72.

Ali, M.A., M.G. Farouque, M. Haque, and A. Kabir. 2012. Influence of soil amendments on mitigating methane emissions and sustaining rice productivity in paddy soil ecosystems of Bangladesh. Journal of Environmental Science and Natural Resources 5:179-185.

Armstrong, R.D., G. Millar, N.V. Halpin, D.J. Reid, and J. Standley. 2003. Using zero tillage, fertilisers and legume rotations to maintain productivity and soil fertility in opportunity cropping systems on a shallow Vertosol. Australian Journal of Experimental Agriculture 43:41-153.

Babu, Y.J., C. Li, S. Frolking, D.R. Nayak, A. Datta, and T.K. Adhya. 2005. Modelling of methane emissions from rice-based production systems in India with the denitrification and decomposition model: Field validation and sensitivity analysis. Current Science 89:1904-1912.

Babu, Y.J., C. Li, S. Frolking, D.R. Nayak, and T.K. Adhya. 2006. Field validation of DNDC model for methane and nitrous oxide emissions from rice-based production systems of India. Nutrient Cycling in Agroecosystems 74:157-174.

Bagnall, D. and Morgan C.L.S. 2018. Approach to valuing soil ecosystem services with linking indicators. *In* Global Soil Security: Soil Science-Society Interfaces. CRC Press, Florida.

Banger, K., H. Tian, and C. Lu. 2012. Do nitrogen fertilizers stimulate or inhibit methane emissions from rice fields?. Global Change Biology 18(10):3259-3267.

Bastiaans, L., M.J. Kropff, N. Kempuchetty, A. Rajan, and T.R. Migo. 1997. Can simulation models help design rice cultivars that are more competitive against weeds?. Field Crops Research 51:101-111.

Belmar, S.B., R.K. Jones, and J.L. Starr. 1987. Influence of crop rotation on inoculum density of Rhizoctonia solani and sheath blight incidence in rice. Phytopathology 77:1138-1143.

Boote, K.J., J.W. Jones, and N.B. Pickering. 1996. Potential uses and limitations of crop models. Agronomy Journal 88:704-716.

Boucher, O., P. Friedlingstein, B. Collins, and K.P. Shine. 2009. The indirect global warming potential and global temperature change potential due to methane oxidation. Environmental Research Letters 4:044007.

Bouma, J. 1989. Using soil survey data for quantitative land evaluation. Advances in soil science 9:177-213.

Bouman, B.A.M., and H.H. Van Laar. 2006. Description and evaluation of the rice growth model ORYZA2000 under nitrogen-limited conditions. Agricultural Systems 87:249-273.

Boyd, J., P. Ringold, A. Krupnick, R.J. Johnson, M.A. Weber, and K.M. Hall. 2015. Ecosystem services indicators: Improving the linkage between biophysical and economic analyses. International review of environmental and resource economics 8:359-443.

Bronson, K.F., H-U. Neue, E.B. Abao, and U. Singh. 1993. Automated chamber measurements of methane and nitrous oxide flux in a flooded rice soil: I. Residue, nitrogen, and water management. Soil Science Society of America Journal. 61:981-987.

Bulluck Iii, L.R., M. Brosius, G.K. Evanylo, and J.B. Ristaino. 2002. Organic and synthetic fertility amendments influence soil microbial, physical and chemical properties on organic and conventional farms. Applied Soil Ecology 19:147-160.

Cai, Z., G. Xing, X. Yan, H. Xu, H. Tsuruta, K. Yagi, and K. Minami. 1997. Methane and nitrous oxide emissions from rice paddy fields as affected by nitrogen fertilisers and water management. Plant and Soil 196:7-14.

Cai, Z., T. Sawamoto, C. Li, G. Kang, J. Boonjawat, A. Mosier, R. Wassmann, and H. Tsuruta. 2003. Field validation of the DNDC model for greenhouse gas emissions in East Asian cropping systems. Global Biogeochemical Cycles.

Campbell, E.E., and K. Paustian. 2015. Current developments in soil organic matter modeling and the expansion of model applications: A review. Environmental Research Letters 10:123004.

Cao, M., J.B. Dent, and O.W. Heal. 1995. Modeling methane emissions from rice paddies. Global Biogeochemical Cycles 9:183-195.

Campbell, C.A., R.P. Zentner, F. Selles, V.O. Biederbeck, B.G. McConkey, B. Blomert, and P.G. Jefferson. 2000. Quantifying short-term effects of crop rotations on soil organic carbon in southwestern Saskatchewan. Canadian Journal of Soil Science 80:193-202.

Chen, H., C. Yu, C. Li, Q. Xin, X. Huang, J. Zhang, Y. Yue, G. Huang, X. Li, and W. Wang. 2016. Modeling the impacts of water and fertilizer management on the ecosystem service of rice rotated cropping systems in China. Agriculture, Ecosystems & Environment 219:49-57.

Chin, K.J., T. Lukow, and R. Conrad. 1999. Effect of temperature on structure and function of the methanogenic archaeal community in an anoxic rice field soil. Applied and Environmental Microbiology 65:2341-2349.

Choudhury, A.T.M.A., and I.R. Kennedy. 2004. Prospects and potentials for systems of biological nitrogen fixation in sustainable rice production. Biology and Fertility of Soils 39:219-227.

Daly, H.E., and J. Farley. 2011. Ecological Economics: Principles and Applications. p. 111–123. *In* From Empty World to Full World. Island Press, Washington.

Das, S., and T.K. Adhya. 2014. Effect of combine application of organic manure and inorganic fertilizer on methane and nitrous oxide emissions from a tropical flooded soil planted to rice. Geoderma 213:185-192.

Daulat, W.E., and R.S. Clymo. 1998. Effects of temperature and watertable on the efflux of methane from peatland surface cores. Atmospheric Environment 32:3207-3218.

De Datta, S.K. 1981. Systems of rice culture. p. 221-251. *In* Principles and practices of rice production. Wiley-Interscience Publication, Toronto.

Diacono, M., and F. Montemurro. 2009. Long-term effects of organic amendments on soil fertility. Pp. 761-786. *In* Sustainable Agriculture Volume. Springer, Dordrecht.

Dominati, E., M. Patterson, and A. Mackay. 2010. A framework for classifying and quantifying the natural capital and ecosystem services of soils. Ecological Economics 69:1858-1868.

Fitzgerald, G.J., K.M. Scow, and J.E. Hill. 2000. Fallow season straw and water management effects on methane emissions in California rice. Global Biogeochemical Cycles 14:767-776.

Flood, J. 2002. Capital markets, globalization and global elites. p. 114-134. *In* Transnational legal processes-globalization and power disparities. Cambridge University Press, United Kingdom.

Fox, W.E., D.W. McCollum, J.E. Mitchell, L.E. Swanson, U.P. Kreuter, J.A. Tanaka, G.R. Evans, H. Theodore Heintz, R.P. Breckenridge, and P.H. Geissler. 2009. An integrated social, economic, and ecologic conceptual (ISEEC) framework for considering rangeland sustainability. Society and Natural Resources 22:593-606.

Freeman III, A. M. 2002. How much is Nature really worth? An economic perspective. *In* Valuing nature: The Shipman Workshop papers. Brunswick, ME: Bowdoin College. https://yosemite.epa.gov/SAB/sabcvpess.nsf/e1853c0b6014d36585256dbf005c5b71/46046a4532069ed485256e3d00426b22/$FILE/VALNATPP.pdf

Fumoto, T., K. Kobayashi, C. Li, K. Yagi, and T. Hasegawa. 2008. Revising a process-based biogeochemistry model (DNDC) to simulate methane emission from rice paddy fields under various residue management and fertilizer regimes. Global Change Biology 14:382-402.

Fumoto, T., T. Yanagihara, T, Saito, and K. Yagi. 2010. Assessment of the methane mitigation potentials of alternative water regimes in rice fields using a process-based biogeochemistry model. Global change biology 16:1847-1859.

Gao, L., Z. Jin, Y. Huang, and L. Zhang. 1992. Rice clock model—a computer model to simulate rice development. Agricultural and Forest Meteorology 60:1-16.

Ghadirnezhad, R., and A. Fallah. 2014. Temperature effect on yield and yield components of different rice cultivars in flowering stage. International Journal of Agronomy. doi:10.1155/2014/846707.

Ghosh, S., D. Majumdar, and M.C. Jain. 2003. Methane and nitrous oxide emissions from an irrigated rice of North India. Chemosphere 51:181-195.

Gilhespy, S.L., S. Anthony, L. Cardenas, D. Chadwick, A. del Prado, C. Li, T. Misselbrook, R.M. Rees, W. Salas, A. Sanz-Cobena, and P. Smith. 2014. First 20 years of DNDC (DeNitrification DeComposition): Model evolution. Ecological Modelling 292:51-62.

Giltrap, D.L., and A.E. Ausseil. 2016. Upscaling NZ-DNDC using a regression based meta-model to estimate direct N_2O emissions from New Zealand grazed pastures. Science of the Total Environment 539:221-230.

Gon, H.A.C., and H.U. Neue. 1995. Influence of organic matter incorporation on the methane emission from a wetland rice field. Global Biogeochemical Cycles 9:11-22.

Gopinath, K.A., S. Saha, B.L. Mina, H. Pande, S. Kundu, and H.S. Gupta. 2008. Influence of organic amendments on growth, yield and quality of wheat and on soil properties during transition to organic production. Nutrient Cycling in Agroecosystems. 82:51-60.

Gornitz, V.M., R.C. Daniels, T.W. White, and K.R. Birdwell. 1994. The development of a coastal risk assessment database: vulnerability to sea-level rise in the U.S. southeast. Journal of Coastal Research 12:327-38.

Grace, P.R., J.M. Oades, H.A. Keith, and T.W. Hancock. 1995. Trends in wheat yields and soil organic carbon in the permanent rotation trial at the Waite Agricultural Research Institute, South Australia. Australian Journal of Experimental Agriculture 35:857-864.

Gunderson, L.H. 2001. Panarchy: understanding transformations in human and natural systems. p. 3 - 24. Island press, Washington.

Hatfield, J.L., and J.H. Prueger. 2015. Temperature extremes: effect on plant growth and development. Weather and climate extremes 10:4-10.

Havlin, J.L., D.E. Kissel, L.D. Maddux, M.M. Claassen, and J.H. Long. 1990. Crop rotation and tillage effects on soil organic carbon and nitrogen. Soil Science Society of America Journal 54:448-452.

Hokazono, S., and K. Hayashi. 2012. Variability in environmental impacts during conversion from conventional to organic farming: a comparison among three rice production systems in Japan. Journal of Cleaner Production 28:101-112.

Issaka, R.N., M.M. Buri, and T. Wakatsuki. 2009. Effect of soil and water management practices on the growth and yield of rice in the forest agro-ecology of Ghana. Journal of Food, Agriculture and Environment 7:214-218.

Itakura, M., Y. Uchida, H. Akiyama, Y.T. Hoshino, Y. Shimomura, S. Morimoto, K. Tago, Y. Wang, C. Hayakawa, Y. Uetake, and C. Sánchez. 2013. Mitigation of nitrous oxide emissions from soils by Bradyrhizobium japonicum inoculation. Nature Climate Change 3:208.

Iwata, F. 1984. Heat unit concept of crop maturity, p. 351-370. *In* Physiological aspects of dryland farming. Gupta, U.S (Ed) Oxford and IBH, New Delhi.

Jeyabal, A., and G. Kuppuswamy. 2001. Recycling of organic wastes for the production of vermicompost and its response in rice–legume cropping system and soil fertility. European Journal of Agronomy 15:153-170.

Jiao, Z., A. Hou, Y. Shi, G. Huang, Y. Wang, and X. Chen. 2006. Water management influencing methane and nitrous oxide emissions from rice field in relation to soil redox and microbial community. Communications in Soil Science and Plant Analysis 37:1889-1903.

Jones, R.K., and S.B. Belmar. 1989. Characterization and pathogenicity of Rhizoctonia spp. isolated from rice, soybean, and other crops grown in rotation with rice in Texas. Plant Disease 73:1004-1010.

Lal, R. 2004. Soil carbon sequestration impacts on global climate change and food security. Science 304:1623-1627.

Lal, R. 2010. Managing soils and ecosystems for mitigating anthropogenic carbon emissions and advancing global food security. Bioscience 60:708-721.

Lee, C., E. Castaneda-Gonzalez, D.L. Frizzell, J.T. Hardke, Y.A. Wamishe, and R.J. Norman. 2016. Rice culture. Arkansas Rice Research Studies 1991:277-280.

Li, C., A. Mosier, R. Wassmann, Z. Cai, X. Zheng, Y. Huang, H. Tsuruta, J. Boonjawat, and R. Lantin. 2004. Modeling greenhouse gas emissions from rice-based production systems: Sensitivity and upscaling. Global Biogeochemical Cycles. doi:10.1029/2003GB002045.

Li, C., J. Qiu, S. Frolking, X. Xiao, W. Salas, B. Moore, S. Boles, Y. Huang, and R. Sass. 2002. Reduced methane emissions from large-scale changes in water management of China's rice paddies during 1980–2000. Geophysical Research Letters 29(20). doi:10.1029/200GL015370.

Li, C., S. Frolking, and R. Harriss, 1994. Modeling carbon biogeochemistry in agricultural soils. Global biogeochemical Cycles 8(3):237-254.

Li, C., S. Frolking, S., and T.A. Frolking. 1992. A model of nitrous oxide evolution from soil driven by rainfall events: 1. Model structure and sensitivity. Journal of Geophysical Research: Atmospheres 97:9759-9776.

Li, C., W. Salas, B. DeAngelo, and S. Rose. 2006. Assessing alternatives for mitigating net greenhouse gas emissions and increasing yields from rice production in China over the next twenty years. Journal of Environmental Quality 35(4):1554-1565.

Li, C.S. 2000. Modeling trace gas emissions from agricultural ecosystems. Nutrient Cycling in Agroecosystems 58:259-276.

Lobell, D.B., and S.M. Gourdji. 2012. The influence of climate change on global crop productivity. Plant Physiology 160:1686-1697.

Lyamuremye, F., R.P. Dick, and J. Baham. 1996. Organic amendments and phosphorus dynamics: II. Distribution of soil phosphorus fractions. Soil Science 161:436-443.

Mäder, P., A. Fliessbach, D. Dubois, L. Gunst, P. Fried, and U. Niggli. 2002. Soil fertility and biodiversity in organic farming. Science 296:1694-1697.

Masunga, R.H., V.N. Uzokwe, P.D. Mlay, I. Odeh, A. Singh, D. Buchan, and S.D. Neve. 2016. Nitrogen mineralization dynamics of different valuable organic amendments commonly used in agriculture. Applied soil ecology 101:185-193.

Matsui, T., O.S. Namuco, L.H. Ziska, and T. Horie. 1997. Effects of high temperature and CO_2 concentration on spikelet sterility in indica rice. Field Crops Research 51:213-219.

Matthews, R.B., R. Wassmann, and J. Arah. 2000. Using a crop/soil simulation model and GIS techniques to assess methane emissions from rice fields in Asia. I. Model development. Nutrient Cycling in Agroecosystems 58:141-159.

McBratney, A., D.J. Field, and A. Koch. 2014. The dimensions of soil security. Geoderma 213:203-213.

McGranahan, G., D. Balk, and B. Anderson. 2007. The rising tide: assessing the risks of climate change and human settlements in low elevation coastal zones. Environment and urbanization 19:17-37.

Moldenhauer, K.E.W.C., P. Counce and J. Hardke. 2001. Rice growth and development. pp.9-20. *In* Arkansas rice production handbook. University of Arkansas Division of Agriculture Cooperative Extension Service, Little Rock.

Muthayya, S., J.D. Sugimoto, S. Montgomery, and G.F. Maberly. 2014. An overview of global rice production, supply, trade, and consumption. Annals of the New York Academy of Sciences 1324:7-14.

Nagai, T., and A. Makino. 2009. Differences between rice and wheat in temperature responses of photosynthesis and plant growth. Plant and Cell Physiology 50:744-755.

Neue, H. U. 1993. Methane emission from rice fields. Bioscience 43:466-474.
Newbold, S.C., C. Griffiths, C. Moore, A. Wolverton, and E. Kopits. 2010. The “social cost of carbon” made simple. Environmental Protection Agency National Center for Environmental Economics. Working Paper Series 10-07.

Oades, J.M. 1984. Soil organic matter and structural stability: mechanisms and implications for management. Plant and soil, 76:319-337.

Olander, L.P., R.J. Johnston, H. Tallis, J. Kagan, L.A. Maguire, S. Polasky, D. Urban, J. Boyd, L. Wainger, and M. Palmer, 2018. Benefit relevant indicators: Ecosystem services measures that link ecological and social outcomes. Ecological Indicators 85:1262-1272.

Pathak, H., C. Li, and R. Wassmann. 2005. Greenhouse gas emissions from Indian rice fields: calibration and upscaling using the DNDC model. Biogeosciences 2: 113-123.

Pathak, H., C. Li, R. Wassmann, and J.K. Ladha. 2006. Simulation of nitrogen balance in rice–wheat systems of the Indo-Gangetic Plains. Soil Science Society of America Journal 70: 1612-1622.

Paul, E.A., and G.P. Robertson. 1989. Ecology and the agricultural sciences: A false dichotomy?. Ecology 70:1594-1597.

Peng, S., J. Huang, J.E. Sheehy, R.C. Laza, R.M. Visperas, X. Zhong, G.S. Centeno, G.S. Khush, and K.G. Cassman. 2004. Rice yields decline with higher night temperature from global warming. Proceedings of the National Academy of Sciences of the United States of America 101:9971-9975.

Pimentel, D., C. Harvey, P. Resosudarmo, K. Sinclair, D. Kurz, M. McNair, S. Crist, L. Shpritz, L. Fitton, R. Saffouri, and R. Blair. 1995. Environmental and economic costs of soil erosion and conservation benefits. Science 267:1117-1123.

Pittelkow, C.M., M.A. Adviento-Borbe, J.E. Hill, J. Six, C.V. Kessel, and B.A. Linquist. 2013. Yield-scaled global warming potential of annual nitrous oxide and methane emissions from continuously flooded rice in response to nitrogen input. Agriculture, ecosystems & environment 177:10-20.

Plummer, M.L. 2009. Assessing benefit transfer for the valuation of ecosystem services. Frontiers in Ecology and the Environment 7:38-45.

Ponnamperuma, F.N. 1984. Effects of flooding on soils. p. 9-45. *In* Flooding and Plant Growth. Academic press, Florida.

Qin, Y., S. Liu, Y. Guo, Q. Liu, and J. Zou. 2010. Methane and nitrous oxide emissions from organic and conventional rice cropping systems in Southeast China. Biology and Fertility of Soils 46:825-834.

Rautaray, S.K., B.C. Ghosh, and B.N. Mittra. 2003. Effect of fly ash, organic wastes and chemical fertilizers on yield, nutrient uptake, heavy metal content and residual fertility in a rice–mustard cropping sequence under acid lateritic soils. Bioresource Technology 90:275-283.

Richmond, T.L. 2017. The Effect of Preflood Nitrogen and Flood Establishment Timing on Rice Development, Nitrogen Uptake and Grain Yield. PhD diss., University of Arkansas, 2017.

Rolston, D. 1986. Methods of soil analysis: part 1 Physical and mineralogical methods. Madison (WI): Soil Science Society of America, 47:1103-1119.

Salas, W., S. Boles, C. Li, J.B. Yeluripati, X. Xiao, S. Frolking, and P. Green. 2007. Mapping and modelling of greenhouse gas emissions from rice paddies with satellite radar observations and the DNDC biogeochemical model. Aquatic Conservation: Marine and freshwater ecosystems 17:319-329.

Saleque, M.A., M.J. Abedin, N.I. Bhuiyan, S.K. Zaman, and G.M. Panaullah. 2004. Long-term effects of inorganic and organic fertilizer sources on yield and nutrient accumulation of lowland rice. Field 53-65.

Sass, R.L., F.M. Fisher, P.A. Harcombe, and F.T. Turner. 1990. Methane production and emission in a Texas rice field. Global Biogeochemical Cycles 4:47-68.

Sass, R.L., F.M. Fisher, Y.B. Wang, F.T. Turner, and M.F. Jund. 1992. Methane emission from rice fields: the effect of floodwater management. Global Biogeochemical Cycles 6:249-262.

Seneesrisakul, K., T. Sutabutr, and S. Chavadej. 2018. The Effect of Temperature on the Methanogenic Activity in Relation to Micronutrient Availability. Energies 11:1-17.

Shi, W., G. Xiao, P.C. Struik, K. SV Jagadish, and X. Yin. 2017. Quantifying source-sink relationships of rice under high night-time temperature combined with two nitrogen levels. Field Crops Research 202:36-46.

Shine, K.P., J. S. Fuglestvedt, K. Hailemariam, and N. Stuber. 2005. Alternatives to the global warming potential for comparing climate impacts of emissions of greenhouse gases. Climatic Change 68:281-302.

Shirato, Y. 2005. Testing the suitability of the DNDC model for simulating long-term soil organic carbon dynamics in Japanese paddy soils. Soil Science & Plant Nutrition 51:183-192.

Siavoshi, M., and S.L. Laware. 2011. Effect of organic fertilizer on growth and yield components in rice (*Oryza sativa L.*). Journal of Agricultural Science 3:217-224.

Sigren, L.K., S.T. Lewis, F.M. Fisher, and R.L. Sass. 1997. Effects of field drainage on soil parameters related to methane production and emission from rice paddies. Global Biogeochemical Cycles 11:151-162.

Simmonds, M.B., C. Li, J. Lee, J. Six, C. Kessel, and B.A. Linquist. 2015. Modeling methane and nitrous oxide emissions from direct-seeded rice systems. Journal of Geophysical Research: Biogeosciences 120:2011-2035.

Snyder, C. and D. Spaner. 2010. The sustainability of organic grain production on the canadian prairies- A review. Sustainability. 2:1016-1034. doi:10.3390/su2041016.

Spurlock, T.N., C.S. Rothrock, W.S. Monfort, and T.W. Griffin. 2016. The distribution and colonization of soybean by Rhizoctonia solani AG11 in fields rotated with rice. Soil Biology and Biochemistry 94:29-36.

Stockmann, U., M.A. Adams, J.W. Crawford, D.J. Field, N. Henakaarchchi, M. Jenkins, B. Minasny, A.B. McBratney, V.D. Courcelles, K. Singh, I. Wheeler. 2013. The knowns, known unknowns and unknowns of sequestration of soil organic carbon. Agriculture, Ecosystems & Environment 164:80-99.

Sullivan, P. 2003. Organic rice production. National center for appropriate technology transfer for rural areas. http://www.attra.ncat.org/attra-pub/rice.html

Tang, H., J. Qiu, E.V. Ranst, and C. Li. 2006. Estimations of soil organic carbon storage in cropland of China based on DNDC model. Geoderma 134: 200-206.

Tilman, D., J. Fargione, B. Wolff, C. D'antonio, A. Dobson, R. Howarth, D. Schindler, W.H. Schlesinger, D. Simberloff, and D. Swackhamer. 2001. Forecasting agriculturally driven global environmental change. Science 292:281-284.

Tol, R.S.J. 2008. The Social Cost of Carbon. *In* The Oxford Handbook of the Macroeconomics of Global Warming. Oxford University Press, United Kingdom.

USDA NASS. Annual Crop Production. January 2018. https://www.nass.usda.gov/Statistics_by_State/Texas/Publications/Current_News_Release/2018_Rls/spr_annual_crop_prod_2018.pdf

USDA NASS. Organic Survey Highlights. September 2015. https://www.agcensus.usda.gov/Publications/2012/Online_Resources/Highlights/Organics/2014_Organic_Survey_Highlights.pdf

USDA NASS. Texas Rice Yield. February 2017. https://www.nass.usda.gov/Statistics_by_State/Texas/Publications/Charts_&_Maps/zrice_yp.php

USDA NASS. 2016 Certified Organic Survey. http://usda.mannlib.cornell.edu/usda/current/OrganicProduction/OrganicProduction-09-20-2017_correction.pdf

USDA-ERS. Rice Outlook 2018. https://www.ers.usda.gov/webdocs/publications/88422/rcs-18d.pdf?v=43208

USDA RCA. 2011 RCA Appraisal. https://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1044939.pdf

US Environmental protection Agency (EPA) Fact sheet. Social cost of carbon. December 2016.

Walkley, A., and I.A. Black. 1934. An examination of the Degtjareff method for determining soil organic matter, and a proposed modification of the chromic acid titration method. Soil science 37:29-38.

West, T.O., and W.M. Post. 2002. Soil organic carbon sequestration rates by tillage and crop rotation. Soil Science Society of America Journal 66:1930-1946.

Wilson Jr, C.E., K. Moldenhauer, R. Cartwright, and J. Hardke. 2013. Rice cultivars and seed production. by Hardke J. University of Arkansas Division of Agriculture Cooperative Extension Service, Little Rock, AR:21-30.

Yu, D.S., H. Yang, X.Z. Shi, E.D. Warner, L.M. Zhang, and Q.G. Zhao. 2011. Effects of soil spatial resolution on quantifying CH_4 and N_2O emissions from rice fields in the Tai Lake region of China by DNDC model. Global Biogeochemical Cycles 25(2).

Zhang, H., Y. Xue, Z. Wang, J. Yang, and J. Zhang. 2009. An Alternate Wetting and Moderate Soil Drying Regime Improves Root and Shoot Growth in Rice All rights reserved. Crop Science 49:2246-2260.

Zhang, Y., C. Li, X. Zhou, and B. Moore III. 2002. A simulation model linking crop growth and soil biogeochemistry for sustainable agriculture. Ecological Modelling 151:75-108.

Zhang, Y., Y.Y. Wang, S.L. Su, and C.S. Li. 2011. Quantifying methane emissions from rice paddies in Northeast China by integrating remote sensing mapping with a biogeochemical model. Biogeosciences 8:1225-1235.

Zhao, Q., S. Brocks, V.I.L. Wiedemann, Y. Miao, and G. Bareth. 2015. Regional application of the site-specific biochemical process-based crop model DNDC for rice in NE-China. Annual Meeting of the German Society for Photogrammetry, Remote Sensing and Geoinformation (35. Wissenschaftlich-Technische Jahrestagung der DGPF), Cologne, Germany. March. 2015. https://www.researchgate.net/publication/275340071

Zou, J., Y. Huang, J. Jiang, X. Zheng, and R.L. Sass. 2005. A 3-year field measurement of methane and nitrous oxide emissions from rice paddies in China: Effects of water regime, crop residue, and fertilizer application. Global Biogeochemical Cycles. doi:10.1029/2004GB002401.