

CRANFIELD UNIVERSITY

Jesús Castillo Velázquez

Full Chain Analysis of Nitrogen Use Efficiency in Rice-Livestock
Systems in Uruguay: Identifying Opportunities for Optimizing N
Management

School of Water, Energy & Environment
PhD Environment and Agrifood

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Academic Year: 2019 - 2022

Supervisor: Professor Guy JD Kirk
Associate Supervisor: Dr Stephan M Haefele
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ABSTRACT

Traditionally the rice crop in Uruguay rotates with pastures for direct livestock grazing. This rotation has allowed a constant rice yield increase of $90 \text{ kg ha}^{-1} \text{ yr}^{-1}$ over the past 50 years, with yields averaging 8.4 Mg ha^{-1} in the last decade. Relatively little nitrogen (N) fertilizer is added ($80 \text{ kg ha}^{-1} \text{ yr}^{-1}$) and the system shows no sign of soil degradation. By contrast, the livestock component is conducted extensively with mostly (75-80%) unimproved pastures, with low animal productivity ($100 \text{ kg liveweight ha}^{-1} \text{ yr}^{-1}$). This thesis is concerned with how the system N balance is sustained at regional and national scales and if it can be maintained in the future. The objectives were to quantify the N balance (all N inputs – outputs), N surplus (all N inputs – N removed in food products) and N use efficiency ($\text{NUE} = \text{N in food products} / \text{all N inputs}$) of different rice-livestock-pasture rotations across Uruguay over time. Because historical records of N inputs and outputs are available at regional and national scales, it was possible to assess the whole system in the long term at a farm-gate level. The DNDC model was parameterised with data from a rice long-term experiment and used to complement the regional N balance data. Results showed a very high average NUE (55–60%) with N balances around neutrality (-6 to $+5 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) and low N surplus ($20 \text{ kg N ha}^{-1} \text{ yr}^{-1}$). These values were worse where pastures have been replaced by other cash-crops or rotations shortened. However, there is an opportunity to intensify the system, maintaining the good N balance by improving the livestock component with improved pastures and higher stocking rates to improve N cycling. Results showed the rice-livestock system of Uruguay is a model mixed farming system with several decades of integration.

Keywords:

Rice-livestock-pasture rotations, mixed farming systems, nitrogen balance, full chain NUE analysis.

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LIST OF ABBREVIATIONS

AND	Animal nitrogen deposition
BNF	Biological nitrogen fixation
C	Carbon
DNDC	DeNitrification-DeComposition model
EIA	Environmental impact assessment
Fch-NUE	Full-Chain nitrogen use efficiency
GHG	Greenhouse gas
HFSR	High fertility short rotation
IA	Index agreement
LCA	Life cycle assessment
LFLR	Low fertility long rotation
ME	Mean error
MEF	Model efficiency
MFMR	Medium fertility medium rotation
N	Nitrogen
NB	Nitrogen Budget
NBAL	Nitrogen Balance
NF	Nitrogen footprint
NSURP	Nitrogen surplus
NUE	Nitrogen use efficiency
RI-CONT	Rice continuous
RI-PAST	Rice pasture
RI-SOY	Rice soybean
SGR	Scenario with grassland
SGRN	Scenario with grassland plus nitrogen
SIP	Scenario improved pastures

1 INTRODUCTION

1.1 Nitrogen use efficiency as a metric to compare N managements.

Nitrogen (N) is the most important nutrient limiting crop yields around the world (Mueller et al., 2012). Since the green revolution, the quantity of N used has increased tenfold while grain/cereal production has only tripled, which shows the importance of N management in facing the future (Ladha et al., 2020). Food demand will continuous to increase over coming decades to keep pace with population and income growth, supported by greater use of N from fertilizers (FAO, 2017). Nitrogen use efficiency (NUE) is a key concept in N management, even more, when agricultural systems face a constant increase in N inputs, mainly of N as fertilizer. Overall, NUE refers to how N is utilized by crops and/or animals, quantifying the ratio of N inputs (e.g., fertilizers, biological fixation, atmospheric depositions, N in feed, soil N) to N outputs (e.g., grain yield, biomass production, animal food products) (Ladha et al., 2005; Oenema & Tamminga, 2005). Of the total N fertilizer consumed globally each year, rice production uses around 23% with NUE of less than 40% of the total N applied in grain or 50% in the whole crop (Ladha et al., 2005; Prasad et al., 2017).

Livestock systems reach lower NUE values compared with crops. The main N losses are linked to NH_3 volatilization and NO_3^- leaching associated with N fertilization of forages and manure management (Groenestein et al., 2019; Sutton et al., 2013). These reach more than $200 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ in European livestock production and represent around 70 and 80% of all the agricultural N flow to water bodies and ammonia emissions respectively (Leip et al., 2015). In comparison, because of low stocking rates (< 1 livestock unit ha^{-1}), N losses from extensive livestock systems are very low, typically around $10 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ (Oenema, 2006).

The increased trend in N fertilizer use in crop systems in the last decades and the expected demand in animal protein linked with more intensive livestock systems, will require further efforts to improve the efficient use of this nutrient (Ladha et al., 2020; Lassaletta et al., 2016). Excess reactive N in the environment is mentioned as one of the five main emerging threats facing humanity and the

planet, for its impact on the climate, the environment, and public health (PNUMA, 2019).

Therefore, the use of N in agricultural systems must be thought not only in terms of physical and economically productivity but also as a way to minimize negative externalities associated with environmental pollution and ensure the system sustainability and quality in the future (Fixen et al., 2015). The NUE considered as an established metric to benchmark N management could be useful in assessing N management in both food systems, facing the future challenges in terms of N.

1.2 Rice systems around the world

Rice is the staple food for half of the world population, being produced in all the continents except Antarctica (Prasad et al., 2017). Annually, around 164 Mha are harvested worldwide of which 85% is in Asia, followed by Africa and the Americas with 10 and 4%, respectively, while the remaining area is found in Europe and Oceania. This extensive distribution around the globe implies a very wide range of ecosystems, including upland rice, deep-water rice, rainfed and irrigated rice (GRiSP, 2013). The latter is the most important ecosystem since it accounts for half of the world's rice area and 75% of total production (Prasad et al., 2017). Cropping systems range from continuous lowland rice to lowland rice combined with upland crops, particularly wheat, maize, grain legumes or vegetables (potato, garlic). This involves up to 300 kg ha⁻¹ of N fertilizer addition. Such cropping sequences are important in China, India, Pakistan and Bangladesh, among others (Ladha et al., 2020; Pathak et al., 2006), and are been increasingly adopted in countries where water scarcity and human population expansion occur (Alberto et al., 2014). Legume – rice crop sequences can spare N fertilizer per unit rice production compared with continuous rice or the wheat-rice sequence (Zhou et al., 2022), and can alleviate soil degradation (Liu et al., 2022). In other regions, where rice is produced as a cash-crop (e.g., USA, Brazil), rice has been increasingly associated with soybeans. Glyphosate resistant soybean varieties are increasingly used to control persistent weeds from rice (Popp et al., 2005; Ribas et al., 2021). There have been reports of both increased and

decreased N efficiencies and soil N mining in these rotations (Green & Blackmer, 1995; Nishida et al., 2013).

A much less common rice cropping system is that combining rice with livestock. Some examples are found in India and Japan (Asai et al., 2018; Deng et al., 2020). In general, this involves interacting products, by-products or residues (grains, brans, fibre, crop residues, manure) but the two components are on different portions of land. Integrated or mixed systems are intermediate between the extremes of extensive rangeland and intensive cropland, with both components often benefitting. This type of crop-livestock system integration is through rotation, with a temporal separation of livestock and crop production on the same area of land (Bell & Moore 2012). Time scales for both components range from short periods up to years, typically determined by the crop production stage. Such systems with a real integration between livestock and rice can be found in some parts of southern Brazil and across Uruguay. In southern Brazil, the system has moved from continuous rice to integrated rice-livestock production. These systems include different arrangements with rice of annual or perennial pastures, upland crops (soybean, maize, Sudan grass) and livestock for direct grazing. There are advantages over the less diversified systems in terms of crop yield, nutrient availability and economic income (Carlos et al., 2020; Vogel et al., 2021). The case of Uruguay is different because all the rice land is integrated in a stable pasture-livestock rotation, where both system components share the same land surface in different stages of the rotation. This is based on strong integration between rice and livestock farmers, the rice mill industry, the rice growers' associations and research institutes (Palmer, 2012). An overview of each component of this integrated system is presented next.

1.3 Components of the rice-livestock system in Uruguay

Rice production is conducted in association with pastures for direct livestock grazing (Deambrosi, 2003). This distinctive aspect has developed since the mid-1960s as a result of an international project for the joint agricultural development of eastern Uruguay and southern Brazil (Sombroek, 1969). Since then, livestock and pastures have been present in areas previously dominated by temporary

wetlands of low economic value, only used for rice. With that background and despite the relatively small production area (165 Mha), the Uruguayan rice sector reached a national productivity $> 8 \text{ Mg ha}^{-1}$ in the last decade, which places it among the most productive rice producing countries worldwide (Blanco et al., 2010; Singh et al., 2017). The high productivity and low domestic market consumption (5-10%) lead to the country ranging in the top-ten world rice exporters (Workman, 2018).

Interestingly, the high rice productivity has been achieved with relatively low amounts of N added as fertilizer, although the rates increased from 40 to 80 kg N ha^{-1} in recent years (Pittelkow et al., 2016). The numbers indicate that per unit of added fertilizer, more than 100 kg grain are produced, tripling what is reported internationally. However, despite the achieved high efficiency, the amount of N removed in the grains exceeds the N added as fertilizer. This led to the hypothesis that there must be other N contributions for the crop compensating for this difference, such as that from biological N fixation (BNF) by microorganisms in the rice floodwater or from the legume pasture in the rotation (Deambrosi & Blanco, 2008). Although the contribution of N fixing organisms in rice systems has been reported in many parts of the world and the N input could reach up to 40 kg N ha^{-1} (Fern et al., 2000; Quesada et al., 1997), yield increases are usually not above 0.5 Mg ha^{-1} (Banayo et al., 2012). Isotopic ^{15}N studies on rice fields of Uruguay did not detect a substantial contribution from this source (Castillo, 2018). On the other hand, local studies have shown that N from BNF of grasses mixed with white clover can reach up to 100 kg N ha^{-1} in half a year (Labandera et al., 1988), and between 80 and 400 kg ha^{-1} when legume cover-crops seeded prior to rice were included (Schulz et al., 1999). However, not all the rice area in Uruguay rotates with legume pastures, and for the areas that do, the frequency of legumes in each rotation cycle is variable (Deambrosi & Blanco, 2008; DIEA, 2022). Therefore, this N input could not be solely responsible for maintaining the N budgets of the system. Therefore, it is unclear how rice in these systems has been operating for several decades, increasing yield over time with relatively low N inputs from fertilizers.

Livestock production is the main economic activity of Uruguay and is conducted on almost all the national agricultural area (DIEA, 2020; Simoes & Hidalgo, 2011). Production numbers for the livestock component are less impressive than those for rice. However, the free-range grazing and animal welfare conditions across all the national herd enable that component to achieve high international prices and provide access to international high values markets. The livestock component is usually managed extensively at a low stocking rate averaging 0.8 livestock units ha⁻¹, achieving around 100 kg ha⁻¹ yr⁻¹ liveweight meat productivity (Dieguez & Pereira, 2020; Soares de Lima, 2009). These numbers are associated with a low to medium native grassland productivity, ranging from 2900 to 6000 kg dry matter ha⁻¹ yr⁻¹. Native grasslands determine an extensive but resilient system, tolerant to wide variation in climatic conditions. They are characterized by very low amounts of inputs, and based on a biophysical approach to match animal demand with pasture growth in conditions of high climate variability between years (Jaurena et al., 2021). Improved pastures on the other hand, contain mixtures of a Gramineae spp. (*Lolium multiflorum* Lam. or *Festuca arundinacea* Scherb.) and forage legumes (*Trifolium repens* L. and *Lotus corniculatus* L.), which can triple animal productivity (Berretta et al., 2000). A similar trend is found in livestock production strictly associated with the rice rotation.

Both components are strongly integrated based on economic agreements between rice and livestock farmers. Around 75% of rice farmers rent the land to livestock farmers (DIEA, 2003) with shared use of various resources. For example, the rice farmer can provide drinking water for animals in summer, frequently the livestock farmer provides cattle to the rice farmer to manage the rice stubble for the next rice crop growing season and/or the rice farmer facilitates the pasture sown after the harvest of the last rice of the rotation. Further on, the rice milling industry ensures the payment of the land rental to the livestock farmer, independent of the final grain yield or economic profit reached by the rice farmer. In addition, the rice farmer frequently contributes labour to the livestock farmer land such as for road maintenance, pasture seeding and fertilizer application.

But despite this strong integration, research has typically focused on each component separately. For example, for the livestock component there are some

proposals to increase animal productivity whilst reducing GHG emission (Jaurena et al., 2021; Kanter et al., 2016). This could also positively affect the rice component (reducing N fertilizer requirements) but a combined assessment considering both components has never been done. An example from the rice component is that research has focused on identifying the main N sources to the crop, exploring the yield response and NUE in different rotations with different N fertilizer recommendations at the crop level (Castillo, 2018). Related results showed that the Uruguayan rice component achieved consistent N fertilizer recoveries by the plant (apparent recovery efficiency), and the grain yield increased according to the extra plant N uptake (physiological efficiency), but the yield gain from applied N (partial factor productivity) was extremely high. This suggests there is a considerable contribution from unknown N sources, most likely linked to the livestock-pasture component, playing a key role in the achieved efficiency. But this and other components of the N cycle and N losses have not been properly investigated to develop N management strategies considering the whole integrated system. A schematic representation of the rice-livestock system showing the rotation variations found in a typical year is presented in Error! Reference source not found.. In the figure, the rice growing season is represented with the “A” letter while the different forages for direct livestock grazing are indicated by “D, F and H”, being “I” the representation of the beginning of a new pasture cycle through an aeroplane seeding. Letters “B, C, E and G” correspond to the different land preparation lengths according to each rotation type. Additional information on the chain labour management for each rotation variation could be found in Appendix **B.1**.

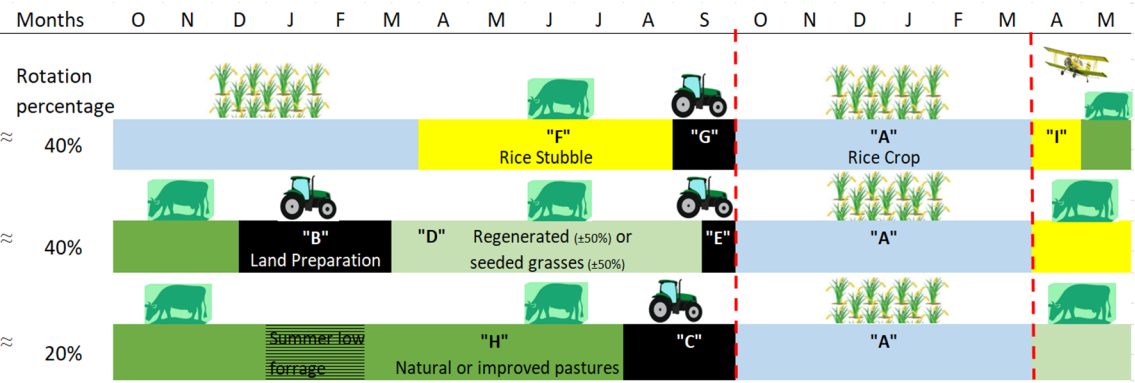


Figure 1-1 Schematic representation of the rice-livestock system of Uruguay and the annual average percentage of type of rotation seeded with rice each year. Vertical red dotted line indicates a typical rice growing season.

1.4 Quantifying NUE at the system level

There are many approaches to calculating NUE. Most are orientated to crop production/response, and follow improvement via fertilizer management or plant breeding and operate on a short-term scale (Dobermann, 2007). Such approaches are less useful for N cycling over time and within a cropping system. Related to that, Congreves et al., (2021) grouped common NUE indices used in agricultural research in fertilizer, plant, soil, isotope, ecology and system-based indices. A system-approach could enable comparisons that differ in soil properties, crop sequences and climate, among others, thus capturing differences in biophysical controls on N dynamics (Martinez-Feria et al., 2018). Consequently, a system NUE index (Full chain NUE, Fch-NUE) has been proposed. This index is useful for characterizing the complete food system, identifying all the processes prior to food processing or consumption. It also allows estimation of the gap between actual NUE and targeted NUE, based on, for example, best management practices or on the average values obtained in well-managed field experiments (EU Nitrogen Expert Panel, 2015). Besides the system level, this index could also be applied to the system components. The Fch-NUE is based on the mass balance concept, where for a given system:

$$N \text{ inputs} = N \text{ outputs} + N \text{ losses} + \text{changes in } N \text{ stocks} \quad (1-1)$$

In Equation (1-1), all N inputs, outputs and losses of the system are considered while changes in N stocks include soil N to system N changes (food, feed, etc). This concept could be used as an indicator of N depletion or accumulation in soils. The Fch-NUE is calculated as follows:

$$Fch - NUE = \frac{N \text{ in harvested products}}{N \text{ inputs}} \quad (1-2)$$

In the numerator of Equation (1-2), N in harvested products includes all the N removed from the land (grain, forage) in a cropping system, while elements

considered for animal systems include milk, meat, eggs, and wool, among others. Inputs in the denominator include N in fertilizers, atmospheric deposition, biological N fixation, the difference between import and export N, and as in Equation (1-1), changes in N stock. These last two inputs are relevant when a country boundary is used instead of the farm-gate (EU Nitrogen Expert Panel, 2015).

The calculated NUE values at system level should be compared against defined efficiency thresholds adjusted for each component (crop or animal system). Proposed NUE limits for crop systems frequently found in the literature are $90 < \text{NUE} > 50\%$ (EU Nitrogen Expert Panel, 2015) while the respective values for animal systems depend on the production type. For example, specialized dairy systems including the N content of milk, meat and crops, can have NUE thresholds similar to crops (Gerber et al., 2013). For cattle production from pastures, NUE thresholds are lower, around $25 < \text{NUE} > 10$ (Garrett et al., 2017).

Considering all the Fch-NUE factors, an ideal case would be the combination of a high N output in harvested products, a high NUE and a low amount of N surplus (NSURP). NSURP is expressed here as the difference between all N inputs and outputs, which indicates potential N losses to the environment.

But this is not the only approach for estimating NUE at a system level. Erisman et al., (2018) argued that the focus of the assessment defines the best approach to be used, and the most common approaches are life cycle analysis (LCA), N footprint (NF), N budget (NB) and environmental impact assessment (EIA), all of them with their specific strengths and weaknesses. While a shared strength is the flexible system boundaries, a shared weakness is the inconsistent allocation methods, so equal conditions must be ensured when system comparisons are conducted (Ladha et al., 2020). In terms of data needed to conduct the analysis, all the approaches are data-intensive except the NB. The link with environmental impacts is better considered by the LCA and EIA while the consideration of internal N fluxes and mass flow mass are better assessed by the NB and NF approaches. The difficulty of comparing impacts and site-specific orientation are disadvantages of the LCA and EIA, while the NF and NB approaches are better when applied to

a regional level and above. In a review of 36 peer-reviewed papers across 30 countries, Erisman et al. (2018) found that 44, 30, 18 and 8% of the studies corresponded with the NB, LCA, NF and EIA, respectively, concluding that the NB approach is the most suitable for Fch-NUE due to the direct estimation of NUE, N outputs and NSURP.

Regardless of the selected approach, all these studies rely on the availability of the necessary data. Several of the reviewed studies have been conducted in European countries where data availability is very good. Some others used FAOSTAT and EUROSTAT as data sources. However, even in developed European countries, some important data can be lacking. In these cases, modelling of biogeochemical system processes can be helpful to fill the data gaps. Several models for such purposes are available, and the most frequently used models include DAYCENT, DSSAT, DNDC, CERES-Rice (Nurulhuda et al., 2018). In developing countries, data availability is often poor, preventing precise NUE analysis. The case of Uruguay is an exception because it has several governmental agencies and public organizations with detailed datasets for the national crop-livestock system, as presented next.

1.5 Agricultural data availability in Uruguay

Any study intending to evaluate a component or productive system in its different stages within the food production/processing chain requires broad information and good-quality data, even more so when an approach of sustainable intensification is followed. And even in countries where good-quality data is available, if the flow of that data between other actors in the system is poor, such as between farmers, extensionists, policy makers, workers and industries, or if the knowledge silos are not connected, it becomes a serious limitation (Baethgen et al., 2016).

In Uruguay, there are several government agencies or agencies with government participation that operate at the level of most stages of the production/processing chain for each component. For the livestock component, the animal stock is surveyed annually for each farm of each county and then aggregated at the county level (SNIG, 2022). In addition, there is information available on the number of heads, animal category, weight, and industrial yield of the animals sent to the

abattoir. This allows calculation of the output of the food product, which is associated with the region or county where the animal comes from (INAC, 2020). This is possible due to the traceability of the entire national cattle stock since 2006, beginning at birth of the animal until its final shipment to the abattoir, through the use of an electronic chip recording all movements in the life of each animal of the national livestock stock (Abraham et al., 2014; Freeman & Orejas, 2015). Pasture data is available through annual surveys which show the area of each pasture type (native grasslands, improved pastures, annual forages) for each farm, again aggregated at county or region level (DIEA, 2020). Additionally, forage productivity data for the main ecoregions of the country are available through remote sensing (Lezama & Paruelo, 2022) which can be used to calculate livestock forage availability and, therefore, N recycling.

Data on the rice component is collected both by government agencies and by the rice mill companies themselves, who share the information with research institutes to be analysed and discussed. Therefore, precise data of the seeded area, total product inputs, grain yield, rice area for pasture seeding, among others, is available annually (INIA, 2020). The good quality of the records allowed to develop the national agricultural information system (SNIA) which includes the data mentioned and more. This project, developed by the Institute for Climate and Society (IRI, Columbia University) together with the agricultural and livestock ministry of Uruguay, integrates existing data and produces new information, products and tools for improving climate risk management and to assist decision and policy making in the public and private agricultural sectors (Baethgen et al., 2016).

Nevertheless, studies at a production system level require additional information that is often not available in the general country-wide statistics related to agronomic aspects (e.g., pasture botanical composition for the calculation of N recycled by animals and biological N fixation). For this reason, data from long-term platforms are necessary. The National Agricultural Research Institute (INIA) of Uruguay established various long-term platforms that can be used for these purposes and has extensive experience in handling such data since the early sixties when INIA started its first long-term experiment, currently the oldest in

South America (Debreczeni & Körschens, 2003). One of the last platforms was established in 2012, corresponding to the rice rotation with pasture and other crops, including livestock as part of the treatments. Relevant information includes the pasture composition, pasture productivity, persistence, concentration of nutrients in forage, stubble quantity and nutrients extracted in grain, among others. In recent years, measurements of N_2O losses were added though there are still no measurements of NH_3 volatilization losses, neither at national nor the rice-livestock system level. Therefore, this important N budget factor could only be estimated with biogeochemical models. For this, data from the long-term experiments can be used for model calibration. In summary, there is abundant data at regional and national scales to conduct the analyses in this study.

1.6 Aims and Objectives

The overall aims were to assess the current and past N budgets and NUE of the rice-pasture-livestock system of Uruguay at a component and system level, focusing on the stages of the rotation sensitive to alternative N management options.

1.6.1 Objective 1

To investigate the suitability of the DeNitrification-DeComposition (DNDC) model to simulate crop responses and the main N loss processes in different rice-pasture rotations.

1.6.2 Objective 2

To investigate N balance, NUE and N surplus over 10 years of a long-term experiment with different rice-pasture rotations.

1.6.3 Objective 3

To investigate the N balance, NUE and N surplus of the rice-livestock system in Uruguay at the country level from 2004–21.

1.6.4 Objective 4

To investigate regional differences in N balance, NUE and N surplus of rice-livestock rotations across Uruguay from 2004–22.

Figure 1-2 shows the different assessed study levels for the fulfilment of each specific objective.

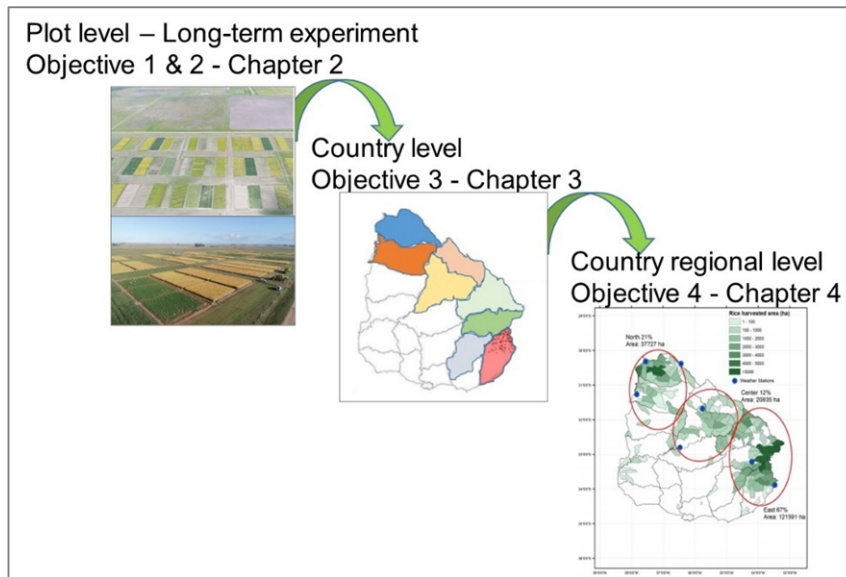


Figure 1-2 Flow diagram of the different assessed study levels.

1.7 Hypotheses

The following section outlines the hypotheses of the whole thesis, which are aligned with each specific objective.

1.7.1 Hypothesis 1

The available agronomic and management data recorded from the long-term experiment and complementary trials on that platform for rice, soybean, and pastures in rotation will allow the parameterization and validation of the DNDC model for these scenarios.

1.7.2 Hypothesis 2

Shifting from the business-as-usual rice-pasture rotation to more intensive rice rotations will affect N balances, NUE and N surplus.

1.7.3 Hypothesis 3

The country has followed a trajectory from high NUE and low yields under very low N inputs, declining NUE and increasing N surpluses as yields increased with increasing N inputs, reaching the current status at with improvements in NUE will require fine-tuning of the system's components or the level of the full nutrient chain.

1.7.4 Hypothesis 4

Even with relatively small N fertilization additions to rice, the NBAL has been around neutrality while NUE has reached high values in all the rotations over the period investigated. However, differences among rotations due to different pasture lengths and management practices are expected. Because of small N outputs in animal products, the livestock component reached positive and stable NBALs and medium to low NUE across the period, resulting in positive NBALs and medium NUE in the whole system.

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2 MEASURED AND MODELLED NITROGEN BALANCES IN LOWLAND RICE-PASTURE ROTATIONS*

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2.1 Abstract

Rotational rice systems, involving pastures, other crops and/or livestock, are common in temperate South America, exemplified by the rice-pasture-livestock system of Uruguay which combines very high rice yields with tight nitrogen (N) balances. The generally good nutrient use efficiency in these systems provides a template for nutrient management in other mixed farming systems, if the underlying processes can be sufficiently well quantified and understood. Here, we studied N balances in rice—non-rice rotations in a long-term experiment in

Uruguay, with the aim of parameterizing and testing the DNDC model of N dynamics for such systems for use in future work. The experiment includes three rotations: continuous rice (RI-CONT), rice-soybean (RI-SOY) and rice-pasture (RI-PAST). We considered 9 years of data on N balances (*NBAL*), defined as all N inputs minus all N outputs; N surplus (*NSURP*), defined as all N inputs minus only N outputs in food products; and N use efficiency (*NUE*), defined as the fraction of N inputs removed in food products. We parameterized DNDC against measured yield and input and output data, with missing data on N losses inferred from the N balance and compared with literature values. The model performance was assessed using standard indices of mean error, agreement and efficiency. The model simulated crop yields and rice cumulative N uptake very well, and soil N reasonably well. The values of *NBAL* were +45 and -20 kg N ha⁻¹ yr⁻¹ in RI-CONT and RI-SOY, respectively, and close to zero in RI-PAST (-6 kg N ha⁻¹ yr⁻¹). Values of *NSURP* decreased in the order RI-CONT >> RI-SOY > RI-PAST (+115, +25 and +13 kg N ha⁻¹ yr⁻¹, respectively). Values of *NUE* (84, 54 and 48% for RI-SOY, RI-PAST and RI-CONT, respectively) decreased as *NBAL* increased. The sensitivity of DNDC's predictions to the agronomic characteristics of the different crops, rotations and water regimes agreed with expectations. We conclude that the DNDC model as parameterised here is suitable for exploring how to optimise N management in these systems.

2.2 Introduction

Increased global trade of crops and livestock as well as the separation of crop and livestock production has led to large imbalances in nutrient budgets across the globe (Grote et al., 2008; Uwizeye et al., 2020). To improve local and global nutrient balances, future food systems should include a return to mixed farming systems with integrated crop and livestock production (Asai et al., 2018; Garrett et al., 2020; Peterson et al., 2020; De Faccio Carvalho et al., 2021). Such systems have a greater potential for efficient nutrient use and cycling than the intensive single commodity systems that have become dominant globally over the last decades (Ghimire et al., 2021; Martin et al., 2016).

An example of a well-integrated system is the national rice-livestock system in Uruguay. This has been practiced for over 50 years, attaining a high level of production for rice (mean grain yields $> 8 \text{ Mg ha}^{-1} \text{ y}^{-1}$) and an average level for livestock ($120 \text{ kg liveweight gain ha}^{-1} \text{ y}^{-1}$) (Castillo et al., 2021) for the prevailing production conditions (e.g. climate, pasture management). Analysis of country-level statistics over the last 16 years showed tight positive nitrogen (N) balances of $+2\text{--}3 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ for both the rice and livestock components as well as the whole system (Castillo et al., 2021). Similarly, De Oliveira Denardin et al. (2020, 2022), have recently shown that rice in rice-livestock systems in Brazil is less dependent on P and K fertilization and have higher yields than mono-cropped rice. This is remarkable given that N inputs to agricultural systems globally range from greatly excessive to inadequate, generating imbalances from environmental pollution to soil N mining (Ladha et al., 2020).

To understand how to optimise nutrient management in such systems, a detailed understanding of local and regional variation in system properties and processes is needed. A good assessment requires detailed data on nutrient inputs and outputs, as well as on loss processes. Regional and national datasets on nutrient balances are inevitably incomplete, especially for gaseous emissions, which are expensive and hard to quantify in detail (Katayanagi et al., 2012). Despite having good data on rice, pasture and livestock production and N budgets (Kanter et al., 2016; Pittelkow et al., 2016), data on gaseous N losses and process modelling of such losses are scarce for the Uruguayan rice-pasture system. Irisarri et al. (2012) and Tarlera et al. (2016) reported very low nitrous oxide (N_2O) emissions ($< 2 \text{ kg N-N}_2\text{O ha}^{-1} \text{ season}$) from rice across the country. However, the main gaseous N loss process in rice systems is generally volatilization of ammonia (NH_3), which can reach up to 50–60% of the applied N (Chen et al., 2015; Chowdary et al., 2004; Wang et al., 2018; Xu et al., 2013). Reliable modelling of NH_3 volatilization is therefore needed to fill-in missing data, and to interpret and extrapolate results. Models for this purpose need to capture all the relevant processes equally well.

There are particular challenges in modelling N dynamics in rice-pasture systems, because very different processes operate in the flooded rice phase compared

with the non-flooded pasture phase. Under flooded conditions, high rates of loss can occur by (1) NH_3 volatilization from the floodwater layer due to daytime increases in floodwater pH (by up to 2 units) as dissolved CO_2 is removed in photosynthesis of floodwater algae; and (2) nitrification-denitrification processes in adjacent oxic and anoxic regions in the soil-floodwater system (Buresh et al., 2008; Kirk, 2004). In the alternating pasture phase, N dynamics depend particularly on (1) biological N fixation (BNF) associated with legume species; (2) the effects of grazing animals; and (3) the retention of fixed N in crop residues and soil organic matter for following crops (Ledgard, 2001; Peoples & Craswell, 1992).

Potential models of rice field N dynamics include DayCent, but it does not calculate NH_3 volatilization (Del Grosso et al., 2015; Necpálová et al., 2015; Gurung et al., 2021), and CERES-Rice, which does calculate NH_3 volatilization but its application to N balances in Uruguayan rice-pasture systems was not promising (Pravia, 2009). The DeNitrification-DeComposition (DNDC) model (Li, 2000) is a widely used process-oriented simulation model of soil C and N biogeochemistry with a focus on agro-ecosystems. Originally developed for simulating GHG emissions from agricultural systems in the USA, the DNDC model has been calibrated and used worldwide (Abdalla et al., 2022; Kesik et al., 2005). The model is dynamic and can capture complex agro-ecosystem interactions for simulating GHG emissions from croplands and other ecosystems. The DNDC model has also been used to simulate crop grain yield and N uptake in lowland rice systems (Babu et al., 2006; Katayanagi et al., 2013; Zhao et al., 2020) as well as in aerobic “upland” crops (Abdalla et al., 2022; Jiang et al., 2021; Zhang et al., 2018).

Our objectives were to parametrise and test the DNDC model for C and N dynamics in rice-pasture rotations in Uruguay, and to use the model to examine the components of the N balance in these rotations. We parameterised and tested the model against data from a no-till long-term experiment on direct seeded rice-pasture rotations in Uruguay, with nine years of measurements of yields and the components of the N balance. To the best of our knowledge, this is the first time the DNDC model has been used for such purposes. If a good fit between

predicted and observed data can be achieved, the model will allow us to predict the trajectory of the existing systems and the effects of altering the current management, such as by the intensification of rice cultivation and introduction of non-traditional crops such as soybeans.

2.3 Materials and Methods

2.3.1 The long-term experiment

The experiment is located at the Instituto Nacional de Investigación Agropecuaria (INIA) Treinta y Tres, Paso de la Laguna Experimental Station in the East of Uruguay (33° 16' 22.2''S; 54° 10' 23.1W). This is a temperate grassland ecoregion with a mesothermic humid climate with daily mean temperatures of 22.6 ± 0.54 °C in summer and 12.0 ± 0.82 °C in winter. For the period 1971 to 2021, annual average rainfall was 1354 ± 283 mm and annual total potential evapotranspiration was 1048 ± 196 mm. The dominant soil type at the site is an Argialboll (Durán et al., 2006) on slopes $\leq 0.5\%$. **Table A-1** gives the main soil properties at the beginning of the experiment in 2012. In the 30 years before the experiment, the area was under a rice-pasture rotation with 2 years of rice followed by 3 years of improved legume pasture for direct livestock grazing. The experiment occupies an area of 7.2 ha, comprising 60 plots, each 60 m long and 20 m wide. There are six rotations in a randomized complete block design with three replications (full details in **6A.1.1** and **Table A-2**). For this study, we selected the three most contrasting rotations in terms of the frequency of rice cultivation and N additions: (a) continuous rice-legume pasture cover crop each year (RI-CONT), (b) rice-pasture cover crop-soybean-legume pasture cover crop every two years (RI-SOY) and (c) rice-pasture cover crop-rice-grazed pasture for 3.5 years every 5 years (RI-PAST) (Error! Reference source not found.).

Table 2-1 Components of the three selected rotations in the long-term experiment in spring-summer (SS) and autumn-winter (AW) seasons from 2012 to 2021. RG = Ryegrass (*Lolium multiflorum* L.); TF = Tall fescue (*Festuca arundinacea* L.), WC = White clover (*Trifolium repens* L.); BT = Birdsfoot trefoil (*Lotus corniculatus* L.); EC = Egyptian clover (*Trifolium alexandrinum* L.).

	2012		2013		2014		2015		2016		2017		2018		2019		2020		2021			
	SS	AW	SS	AW	SS	AW	SS	AW	SS	AW	SS	AW	SS	AW	SS	AW	SS	AW	SS	AW		
RI-PAST ^a	Rice 1	RG	Rice2	TF + WC + BT grazed by sheep								Rice1	RG	Rice2	TF + WC + BT grazed by sheep							
RI-SOY ^b	Rice	RG	Soy	EC	Rice	RG	Soy	EC	Rice	RG	Soy	EC	Rice	RG	Soy	EC	Rice	RG	Soy	EC		
RI-CONT	Rice	EC	Rice	EC	Rice	EC	Rice	EC	Rice	EC	Rice	EC	Rice	EC	Rice	EC	Rice	EC	Rice	EC		

^aRice1, Rice 2 and the first year of a TF + WC + BT pasture were grown in SS from 2012 to 2021

^bRice and soya crops were grown in the Rice-Soybean and Soybean-Rice sequences in SS from 2012 to 2021

2.3.2 Field measurements

Since the beginning of the long-term experiment, data on all management variables for crops and pastures were recorded annually. Rice and soybean yields, nutrient removals with the grain and seasonal pasture production and botanical composition were measured each season and year for each rotation sequence over the main plot (1200 m²). Biological N fixation by the pasture and soybean was estimated based on both, crop and pasture measurements and literature data. For pastures, dry matter production was measured seasonally each year and the botanical composition was recorded. The BNF was estimated multiplying each aboveground legume pasture species dry matter production (Mg ha⁻¹ yr⁻¹) by a fixation coefficient (kg N Mg DM⁻¹) from the literature and including an estimation of the belowground N in nodules, stolon, or roots. For soybean, BNF was estimated based on measurements in 2019-20 and 2020-21 growing seasons of the harvest index and aerial biomass partitioning at R6 stage and N concentration (g kg⁻¹) of grain and plant. We assumed that from the total amount of plant N at R6, 75% was derived from atmospheric N (Alves et al., 2003), and to consider N in roots and nodules, we multiplied that number by 1.5 following Herridge et al. (2008). Similarly, N removal in animal tissue were estimated from animal production and literature data. Ovine production was estimated as a function of measured forage (kg DM ha⁻¹ yr⁻¹) and a grazing efficiency of 60% (Crempien, 1983) multiplied by meat N concentration (g kg⁻¹). For wool consideration, 2 kg animal yr⁻¹ was assumed. The assumption of 15 kg DM animal day⁻¹ was defined produce 1 kg meat. Climate data (daily maximum and minimum air temperature, precipitation, solar radiation, wind speed and air humidity) were obtained from a weather station at the site, with 50 years of records. All the coefficients mentioned above are presented in **Table A-3**.

Aboveground rice N uptake and NH₄-N in the soil solution during flood irrigation (N treatment of 100 kg ha⁻¹) were measured in sub-plots (40 m²) in 2019–20 and 2020–21. For rice N uptake, total aerial plant samples were taken 7 times during the growing season each year, from 15 d after flooding about every 15 d up to harvest. Plant samples were dried at 60°C for 48 h and N concentration measured

by the Dumas method. Soil solution was sampled seven times from immediately after flooding up to 15 d before harvest at 20 cm depth, and analysed colorimetrically (Nelson, 1983). Emissions of N₂O from crops and pastures were measured in the main plots by the closed chamber technique (Minamikawa et al., 2012; Rochette & Eriksen-Hamel, 2008) every 15 d on average, starting immediately after flooding in the rice crop and 1 wk after establishment in pasture and soybean crops.

2.3.3 Parameterising DNDC

The default atmospheric CO₂ value was adjusted to 390 ppm for 2012, increasing by 2.5 ppm each year thereafter (NOAA, 2022). Nitrogen concentration in rainfall was set at 0.35 mg l⁻¹ (Zunckel et al., 2003). Soil clay content, pH and organic carbon content were as measured at the start of the experiment (**Table A-1**). The depth of the water retention layer (0.6 m) and the drainage efficiency (50%) were estimated based on the site conditions. Bulk density, porosity, soil hydraulic conductivity and available water potential values were calculated by the model from the input soil property variables.

Annual crop and cultural practices were included into the cropping management sub-model. Grazing by sheep was allowed for in the RI-PAST rotation with grazing frequency (rotational grazing averaging 8 d occupancy and 20 d of regrowth) and stocking rate (28 animals ha⁻¹ on average) as managed in the experiment. To simulate the start of a chemical fallow period in pasture cover crops and pastures in the final productive year, we selected the “crop termination tillage” option, which does not alter the soil surface.

The model was calibrated by adjusting the crop parameters listed in **Table 2-2** to obtain best fits between measured and simulated crop and pasture yields and N balances (total N uptake, grain and straw N partitioning, and soil NH₄-N concentrations) in the different rotations over the nine growing seasons. All other parameter values were either as determined above or the DNDC default values.

We lacked measurements of NH₃ volatilization, but it is an important part of the N budget (Results). We therefore assessed the sensitivity of simulated NH₃

volatilization and grain yields to the main soil parameters affecting volatilization (carbon and clay contents and pH) and N fertilization rate. We varied each parameter by -30 to +30 % of the standard value with an increment of 10%, consistent with variability across the main rice regions in Uruguay. We also checked the sensitivity of NH₃ volatilization and grain yield to the main crop development parameters (**Table 2-2**) by varying values by -15 and +15%. That range covers the expected variability in biomass fraction and C to N ratio in all rice plant components, and thermal degree days accumulation.

Table 2-2 DNDC cropping parameter settings used for calibration.

	Rice	Soybean	Egyptian clover	Ryegrass	Mixed perennial pasture
Range of maximum grain production (kg ha ⁻¹ yr ⁻¹)	7790–13908	1210–3170	23–46	17–43	63–155 ^a 80–224 ^b 69–92 ^c 14–46 ^d
Biomass fraction (grain/leaf/stem/root)	0.45/0.21/ 0.22/0.12	0.31/0.25/ 0.24/0.2	0.01/0.4/ 0.4/ 0.19	0.01/0.4/ 0.4/0.19	0.02/0.35/ 0.35/0.28
C to N ratio (grain/leaf/stem/root)	35/60/80/85	10/45/45/24	15/10/21/30	15/25/25/30	19/19/19/19
Thermal degree days (TDD)	2800-3140 ^e	2700	1700	1700	4000
N fixation index (N plant/N from soil)	1.05	4	10	1	1.34

^{a, b, c, d} based on measured aboveground biomass fraction for pastures in its 1st, 2nd, 3rd and 3.5th year

^e range of thermal degree days for rice for the different used rice varieties across the years

2.3.4 Model performance

Model accuracy during calibration and validation stages was tested through three indices following (Yang et al., 2014): mean error (*ME*), index of agreement (*IA*) and modelling efficiency (*MEF*):

$$ME = \frac{\sum_{i=1}^n (P_i - O_i)}{n} \quad (2-1)$$

$$IA = 1 - \frac{\sum_{i=1}^n (P_i - O_i)^2}{\sum_{i=1}^n (|P_i - \bar{O}| + |O_i - \bar{P}|)^2} \quad (2-2)$$

$$MEF = 1 - \frac{\sum_{i=1}^n (P_i - O_i)^2}{\sum_{i=1}^n (O_i - \bar{O})^2} \quad (2-3)$$

where P_i and O_i are the predicted and observed values respectively in season i , $\bar{P}$ and $\bar{O}$ are the respective mean values, and n is the number of seasons. If *ME* is above or below 0, the model underestimates or overestimates the observed data, respectively. The dimensionless *IA* index ($0 \leq IA \leq 1$) is used to represent the degree of deviation from zero. The *MEF* values ($-\infty$ to 1), also dimensionless, assess the goodness-of-fit of the model, with $MEF = 1$ indicating a perfect fit and $0-1$ denoting acceptable fit. For $MEF < 0$, goodness of fit must be assessed with a *t*-test.

The good agreement (see results) when comparing simulated and measured field data during the two parametrization stages gave us confidence to use other N parameters simulated by the DNDC model but not measured, to conduct a more complete N budget analysis in the different crop rotation systems.

2.3.5 Nitrogen balances

The N balance, surplus and NUE were assessed for the three rotations based on (Erisman et al., 2018). The nitrogen balance (*NBAL*) was calculated as:

$$NBAL = \sum N_{\text{inputs}} - \sum N_{\text{outputs}} \quad (2-4)$$

where N_{inputs} = N in fertilizers, BNF, atmospheric N deposition, and N_{outputs} = N in food products, N gas losses (NH_3 , N_2O , NO_2 and NO), and N leached.

All N inputs not retained in food products were considered as potential N loss to the environment and was defined as surplus (*NSURP*):

$$NSURP = \sum N_{\text{inputs}} - N_{\text{food products}} \quad (2-5)$$

Components of N_{inputs} = as in Equation 1 and $N_{\text{food products}}$ = amount of N in grain or meat or both.

The N use efficiency was calculated as the fraction of N retained in food products considering all inputs:

$$NUE\% = \frac{N_{\text{food products}}}{\sum N_{\text{inputs}}} \times 100 \quad (2-6)$$

where $NUE\%$ = N use efficiency expressed in percentage while N in food products and N inputs are the same items as in Equation (2-5).

Analyses of variance for *NBAL*, *NSURP* and *NUE%* were conducted with a general linear model in the Infostat software (Di Renzo et al., 2017). In the model, rotation was considered a fixed effect while the block effect nested in year was considered random. A significance level of $P \leq 0.05$ was defined and all data was tested for normality and variance homogeneity.

2.4 Results

2.4.1 Rice yield

Yields were highest in the RI-SOY and in the first rice of the RI-PAST rotations and lowest in the second rice of the RI-PAST and RI-CONT rotations (**Figure 2-1a**). **Table A-5** gives the results of the model assessment using Equations (2-1) – (2-3). The model simulated rice yields well for all rotations over the entire period, with a small average underestimation of $-409 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (**Figure 2-1a**). Analysis within rotations also showed a high association between simulated and observed yields for the first rice of the RI-PAST rotation, the RI-CONT rotation, and the RI-SOY Rotation depending on the RI-SOY rotation sequence (rice-soybean or soybean-rice, respectively). In both cases, the model under or overestimated rice yield on average by -397 and $558 \text{ kg ha}^{-1} \text{ yr}^{-1}$, respectively. In contrast, the simulation for the second rice of the RI-PAST rotation was poor with an average

yield underestimation of $-2937 \text{ kg ha}^{-1} \text{ yr}^{-1}$. This was associated with an average difference in total N uptake of around 25 kg N ha^{-1} compared with the average N uptake of the first rice in the RI-PAST rotation. Based on that N uptake difference we added the mentioned amount as fertilizer and assuming 50% of fertilizer recovery and repeated the model run.

2.4.2 Soybean yield

Yields tended to be lower and more variable during the first years of the experiment because of dry summers and lack of supplementary irrigation, which started in 2015. As with the rice, soybean yield was predicted well by the DNDC model (**Figure 2-1b**). On average, predicted values slightly underestimated yield ($-195 \text{ kg ha}^{-1} \text{ yr}^{-1}$) but efficiency indices values indicated good agreement. Within the RI-SOY rotation, soybean yield predictions were off by -290 and $118 \text{ kg ha}^{-1} \text{ yr}^{-1}$ for the rice-soybean and soybean-rice sequences, respectively, with indices values confirming a good simulation (**Table A-5**).

2.4.3 Pasture yield

The greatest pasture production was during the second year and the least during the last cycle which considered only half a year. Dry matter production during the first and third year were similar. Pasture dry matter production in the RI-PAST rotation was well predicted by the model for the yearly average and the pasture growing season. However, this changed when analysing each growing season separately (**Figure 2-1c**). Except for model prediction of second-year pastures, the simulations showed only intermediate values for the goodness of fit index values (**Table A-5**). *t*-tests did not detect differences between predicted vs. observed values except for the third-year pastures ($P = 0.04$).

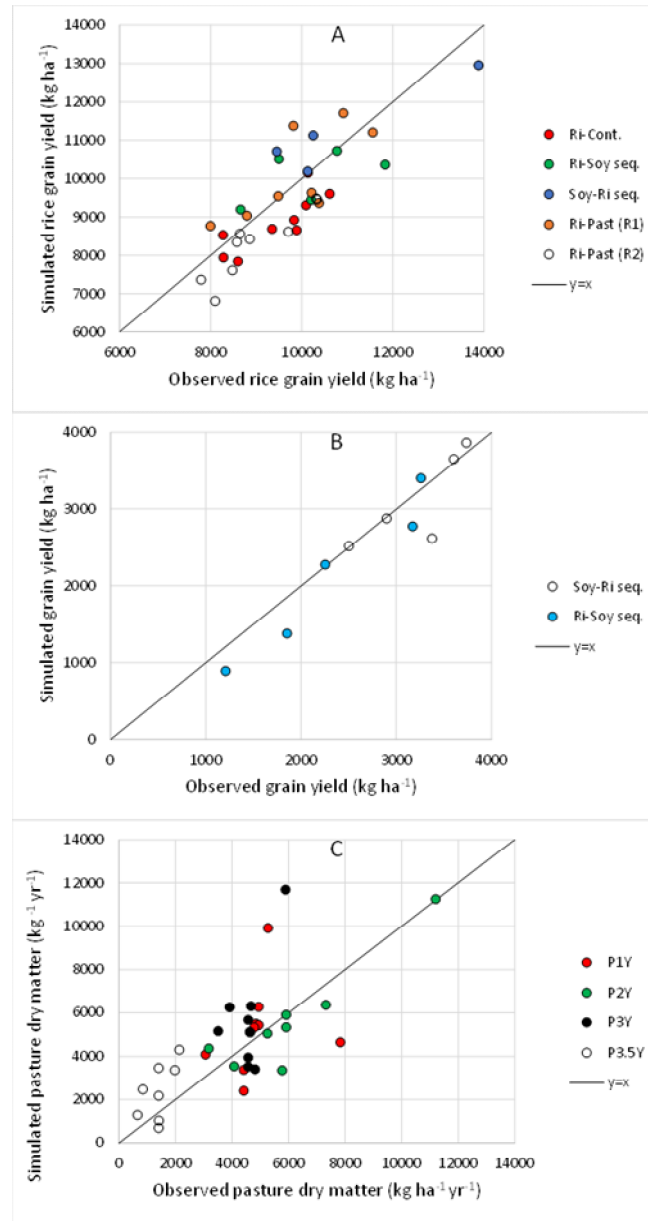


Figure 2-1 Comparison of simulated and observed grain yield for rice (A), soybean (B) and pasture (C) dry matter yield from 2012 to 2021.

2.4.4 Emissions of N₂O

Measured N₂O-N emissions were small in the flooded and dry soil stages (0.54 ± 2.6 and 0.37 ± 2.8 g N₂O-N ha⁻¹ day⁻¹, respectively). During the pasture phase of the RI-PAST and for rice in RI-CONT rotation, measured and simulated N₂O-N values were zero in both growing seasons. Simulated N₂O emissions underestimated measured records during the first rice of the RI-PAST rotation (0.54 and 0.13 g N₂O-N ha⁻¹ day⁻¹ for 2019-20 and 2020-21 respectively) and overestimated for soybean in the RI-SOY rotation (0.5 g N₂O-N ha⁻¹ day⁻¹).

According to the calculated model accuracy indices values, predicted N₂O emissions showed low to intermediate similarity to the observed values (**Table A-5**). On average, the range of predicted and measured N₂O-N emission values was very narrow, and the values were low (0.45 and 0.48 g ha⁻¹ day⁻¹, respectively). For all rotations, t-tests were conducted between predicted and observed values and no significant differences were detected.

2.4.5 Model sensitivity analysis

Simulated NH₃ volatilization and rice yield were sensitive to variations in all the soil parameters tested (**Figure 2-2**). Ammonia volatilization increased with N addition (by 25% on average across the rotations) and to a smaller extent with soil organic C (13%) and soil pH (7%). The effect of pH was greater during the dry stage of rice (around 35 days from seeding to flooding), increasing by 29% with higher soil pH values. Conversely, volatilization decreased with increasing soil clay content (-10%). The effects of crop parameters on NH₃ volatilization ranged from -6 to 8%. In all rotations, rice yields increased slightly with N addition and soil organic C content and decreased with increasing soil clay content and pH. Crop parameters also had small effects on rice yield. In summary, the sensitivity of NH₃ volatilization to variations in soil parameters was greater than for crop parameters, and rice yield was little affected by any of the tested parameters.

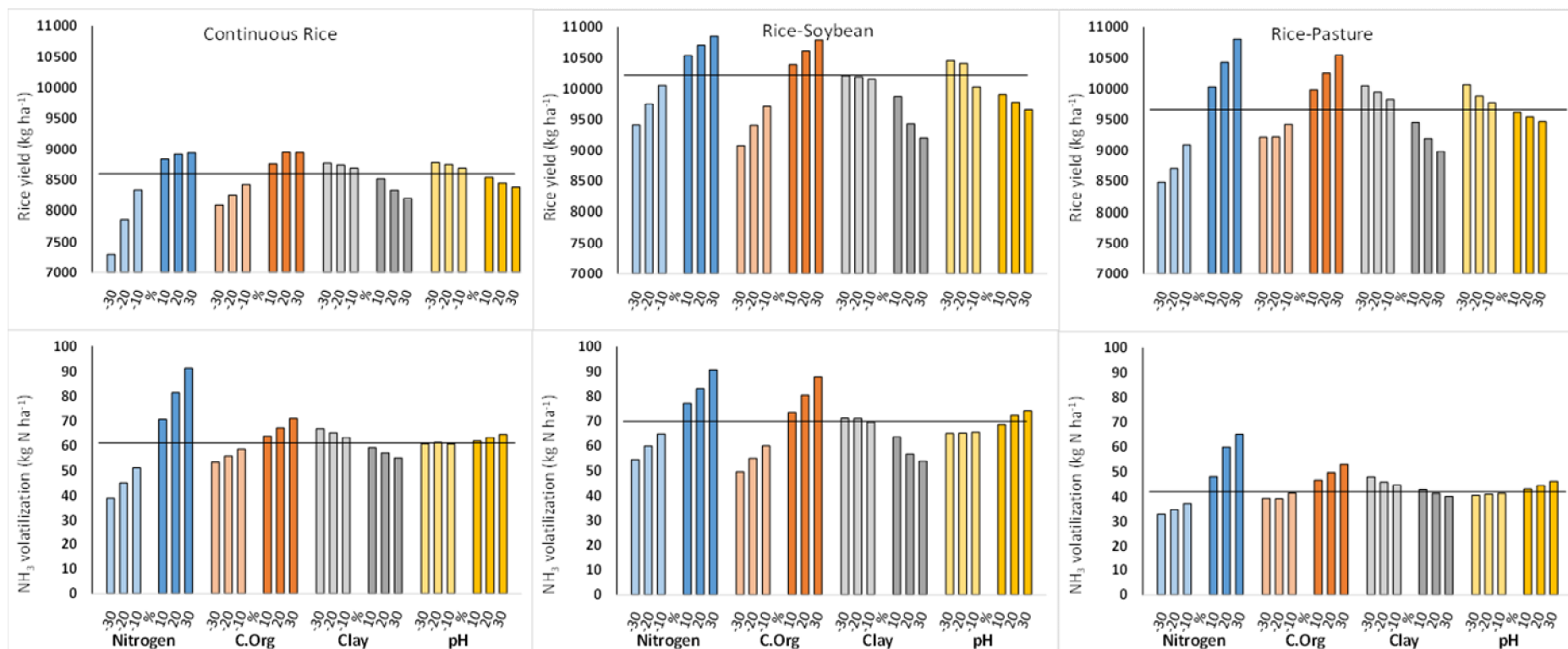


Figure 2-2 Sensitivity of modelled NH₃ volatilization and rice grain yield to variations in the N fertilization rate and soil organic C content, clay content and pH. The parameter values were varied individually by the indicated values. The horizontal lines are the results with the standard parameter values.

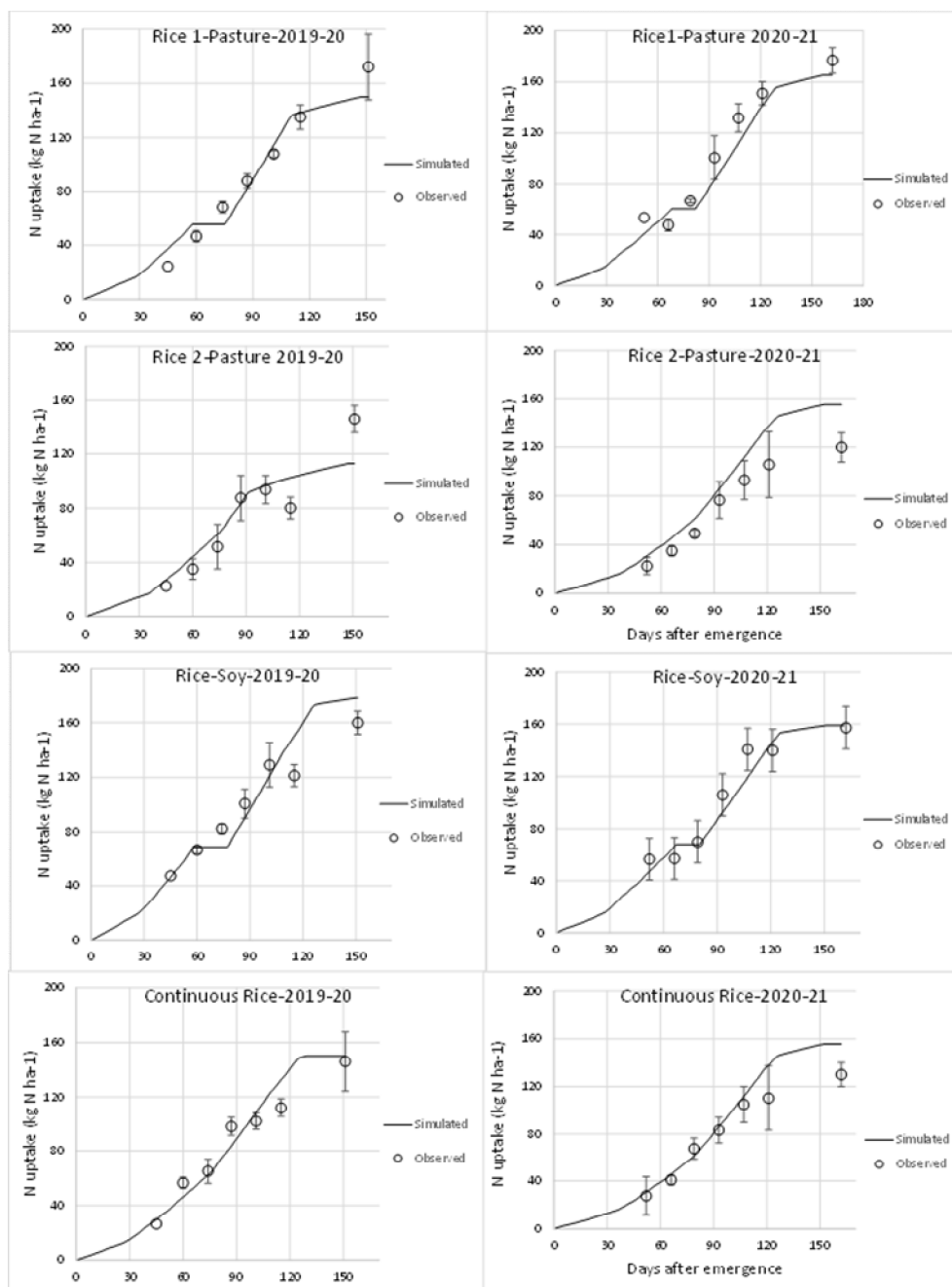


Figure 2-3 Comparison of model simulated and observed rice N uptake (kg N ha^{-1}) in rice-pasture (first and second rice), rice-soybean and continuous rice in the 2019-20 and 2020-21 growing season. The error bars for observed values of each figure indicate the standard error among three replications.

2.4.6 Validation of the DNDC model for simulating rice N uptake and $\text{NH}_4\text{-N}$ concentration in flooded soil

In both 2019–20 and 2020–21 growing seasons, total rice N uptake was greater in the first rice of the RI-PAST (172 and 177 kg N ha⁻¹) and the RI-SOY (160 and 157 kg N ha⁻¹) systems compared with the second rice of RI-PAST (146 and 120 kg N ha⁻¹) and RI-CONT rotations (146 and 130 kg N ha⁻¹), following the same trends as observed for rice yield. The results of the evaluation index values for cumulative N uptake are shown in **Table A-5**. These values indicate that the DNDC model simulated the cumulative rice N uptake in the different rotation systems very well (**Figure 2-3**), with a low deviation between predicted and simulated N uptake values. The model could also satisfactorily simulate the amount of N kept in strow and rice grain in the different rotations and years (**Figure 2-4**).

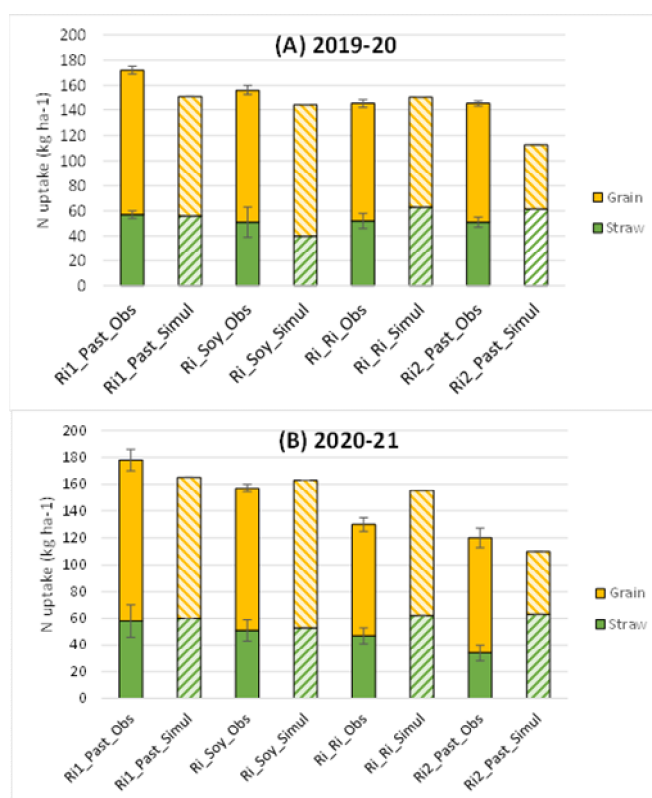


Figure 2-4 Comparison of model simulated and observed rice grain and straw N uptake (kg N ha⁻¹) at physiological maturity in the rice-pasture (first and second rice), rice-soybean and continuous rice in 2019-20 (A) and 2020-21 (B) growing season. The error bars in observed values of each figure indicate the standard error among three replications.

Ammonium in the soil solution during the flooded stage averaged low values (6.3 ± 2.6 and 9.2 ± 5.4 kg NH₄-N ha⁻¹) for 2019–20 and 2020–21, respectively, which decreased from the beginning of the measurements up to the end of it. No differences were detected among rotations within the year. Index values for NH₄-N concentration in the soil solution were *ME* = 1.38 and 0.68; *IA* = 0.48 and 0.54; *MEF* = -0.27 and 0.05, for the first and second rice growing season, respectively. Although observed and predicted average values for NH₄-N in soil solution were very close (10 vs. 6.8 kg ha⁻¹ and 9.8 vs. 8 kg ha⁻¹ in the predicted vs. observed for 2019-20 and 2020-21, respectively), index values of model accuracy showed intermediate results (**Table A-5**). A paired *t*-test did not show differences between the predicted and observed values for this variable in the RI-CONT and the first rice of the RI-PAST rotation in both years, but differences were detected for the RI-SOY rotation in 2019-20 (*P* = 0.02) and 2020-21 (*P* = 0.04). But the trend in simulated and observed soil NH₄-N values was close, corresponding to the small range of observed values for this variable (**Figure 2-5**).

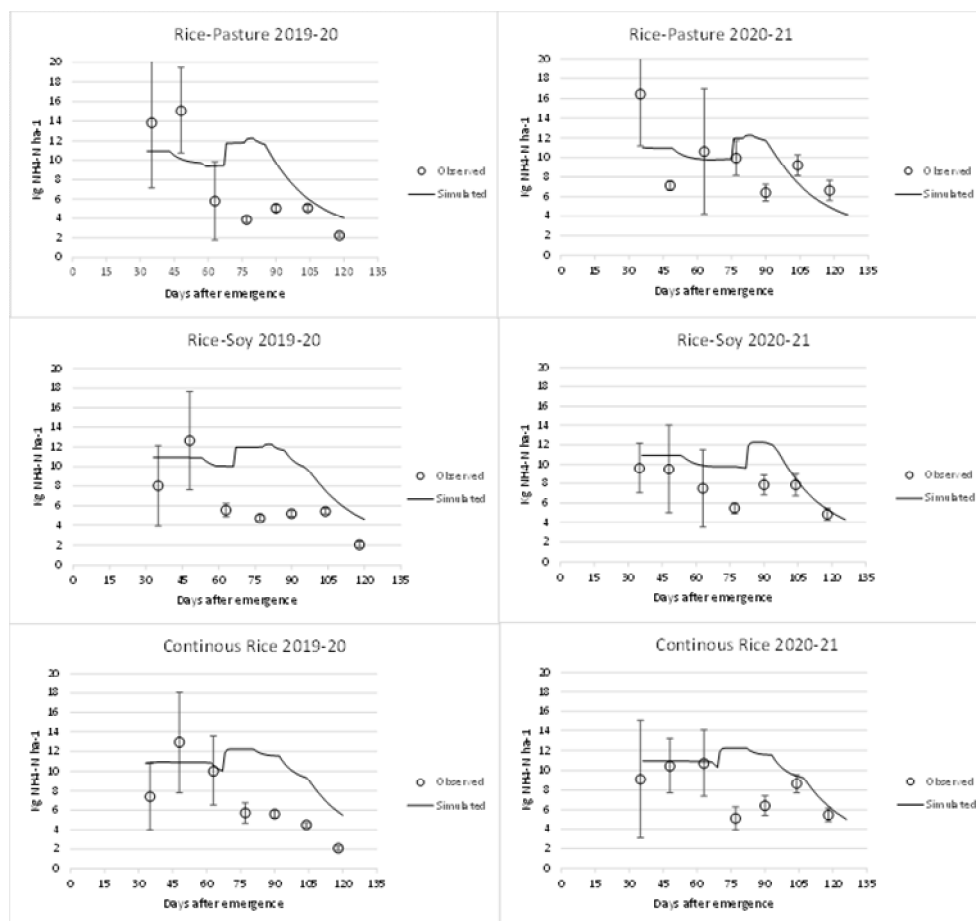


Figure 2-5 Comparison of model simulated and observed soil $\text{NH}_4\text{-N}$ during the flooded rice stage in rice-pasture, rice-soybean and continuous rice in 2019-20 and 2020-21 growing season. The error bars in observed values of each figure indicate the standard error among three replications.

2.4.7 Nitrogen budgeting of rice rotations

Based on the results presented above, we conclude that the model simulations are well corroborated by the observed data of crop/pasture response as well as selected N indicators in the plant-soil system. Therefore, we were confident in using the model to predict unmeasured N loss components.

Total N inputs were different, accounting for 209, 139 and 51 $\text{kg ha}^{-1} \text{yr}^{-1}$ in RI-CONT, RI-SOY and RI-PAST respectively. The main N source was fertilizer for RI-CONT (70%), but it was biological N fixation (BNF, 66%) for RI-SOY. The BNF in RI-CONT (57 $\text{kg N ha}^{-1} \text{yr}^{-1}$) comes from the Egyptian clover used as cover pasture crop, and in RI-SOY from soybean (136 $\text{kg N ha}^{-1} \text{yr}^{-1}$) and Egyptian clover used as cover crop before rice seeding (47 $\text{kg N ha}^{-1} \text{yr}^{-1}$). The RI-PAST rotation had both the lowest inputs of N fertilizer (26 $\text{kg N ha}^{-1} \text{yr}^{-1}$) and BNF (17

kg N ha⁻¹ yr⁻¹). Although N fertilizer inputs for RI-SOY (42 kg N ha⁻¹ yr⁻¹) and RI-PAST might seem low, considering the length of the study (9 years) and the average number of rice crops in each rotation phase (4.5 in RI-SOY and 3.3 in RI-PAST rotations), 84 and 71 kg N ha⁻¹ were applied in each rice modelled growing season, respectively.

The RI-PAST rotation had the smallest N outputs (**Figure 2-6**). Nitrogen retained in grain crops accounted for 72, 58 and 51% of total N outputs for RI-SOY, RI-CONT and RI-PAST, respectively, while gaseous N losses represented 30, 42 and 34% of N outputs, respectively. For the three rotations, NH₃ volatilization was the main N loss process (39 kg NH₃-N ha⁻¹ yr⁻¹). Both N₂O emission (4 kg N ha⁻¹ yr⁻¹) and N leaching (2 kg N ha⁻¹ yr⁻¹) were negligible by comparison. Volatilization of NH₃ was mainly associated with the rice phase of each rotation (93% on average). Cumulative NH₃ volatilization losses were different among rotations being 62, 41 and 17 kg NH₃-N ha⁻¹ yr⁻¹ for RI-CONT, RI-SOY and RI-PAST, respectively. The greatest amount of N volatilised was found during the rice phase in the RI-SOY rotation (74 kg NH₃-N ha⁻¹) followed by RI-CONT and RI-PAST (61 and 44 kg NH₃-N ha⁻¹, respectively), all statistically different. In the absence of N fertilizer addition, simulated volatilization was on average 18, 9 and 15 kg N-NH₃ ha⁻¹ for the same rotations.

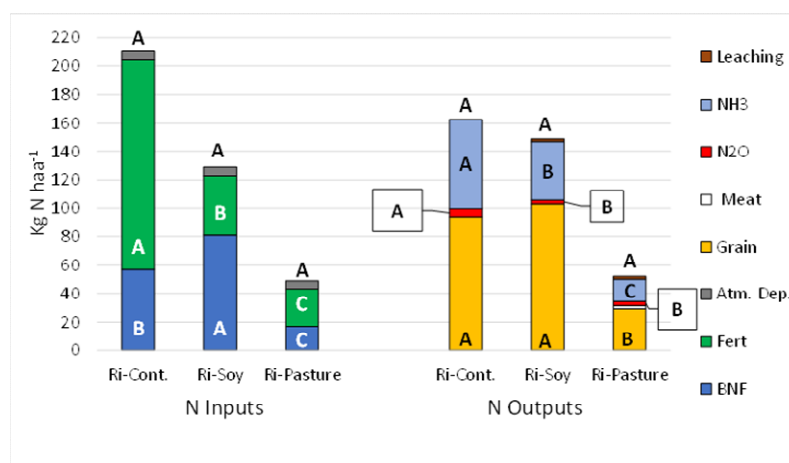


Figure 2-6 Composition of N inputs and outputs and average N amounts for rice-pasture, rice-soybean and continuous rice in 2019-20 and 2020-21 growing season. Capital letters in the bars correspond to the mean comparison (Fisher 5%) among rotations for each component following.

Differences between N inputs and outputs generated differences in N balances, N surpluses and NUE (**Table 2-3**). *NBAL* ranged from +45 kg N ha⁻¹ yr⁻¹ in RI-CONT to -23 kg N ha⁻¹ yr⁻¹ in RI-SOY, with RI-PAST having an intermediate value (-6 kg N ha⁻¹ yr⁻¹). Because of the high fertilizer N input in RI-CONT, which is similar to the total output of this rotation, the positive *NBAL* values are close to the BNF and atmospheric deposition inputs (**Table 2-3**). In contrast, *NBAL* was negative in RI-SOY even though one of the rotation components fixed N. The amount of N derived from the atmosphere to the soybean was defined in 75% of the total N uptake, explaining partly that negative N balance. The RI-PAST rotation reached a very tight N balance of -6 kg N ha⁻¹ yr⁻¹. In the RI-CONT rotation, the amount of N not retained in grain from all the N inputs (*NSURP*) was 115 kg N ha⁻¹ yr⁻¹. This value represented around 79% of the N added as fertilizer. Both, RI-SOY and RI-PAST showed low *NSURP* values (**Table 2-3**). The *NUE* % was higher in RI-SOY (84%) compared with RI-CONT (48%). The *NUE* % in RI-PAST was 53%, similar to RI-SOY. Higher *NUE*% values were associated with less positive *NBAL*.

Table 2-3 Calculated N balance, N surplus and % N use efficiency (defined in Equations (2-4) (2-5) (2-6), for the three rotations from 2012 to 2020.

Rotation	N balance	N surplus	NUE
	kg N ha ⁻¹ yr ⁻¹		%
RI-PAST	-6	13	53
RI-SOY	-23	25	84
RI-CONT	45	115	48
LSD ^a	22	16	18

^a least significant difference

2.5 Discussion

2.5.1 Testing the DNDC

The DNDC model simulated rice yields in the different rotations very well, with a slight average underestimation of 4.5%. This is similar to simulations with DNDC in other high yielding rice systems (Jiang et al., 2021; Zhang et al., 2022). The simulated yields in the different rotations followed the same trends as reported regionally (Ribas et al., 2021) and at national level in Uruguay (García et al., 2009; Méndez, 1993). By contrast, previous modelling studies for the Uruguayan rice-pasture system using the DSSAT model did not correctly simulate yields, largely due to poor simulation of soil and fertilizer N interactions during the flooded stage (Pravia, 2009).

The only situation where the simulated and observed rice yields disagreed greatly was for the second rice of the RI-PAST rotation, where yields were under-predicted by 9%. Compared with the first rice of the RI-PAST rotation, DNDC predicted soil in the second rice had 23% greater dissolved organic carbon (DOC) production, 57% less DOC consumption by soil microbes, 17% greater heterotrophic respiration and 60% more N assimilation by microbes (data not presented). This indicated the sensitivity of DNDC to crop residue decomposition, with the above changes in C decomposition parameters indicating less mineralised N for the rice and therefore lower rice yields. However, the under-predicted yield in the second rice (-9 %) compared with the first rice (- 4 %) of RI-PAST could also be due to errors in simulating decomposition of the Ryegrass cover crop between the rice crops. The use of herbicides at the beginning of the “chemical” fallow generates good soil N conditions for the growth of the next crop through mineralization/immobilization turnover. The DNDC model does not include this “chemical” fallow in its setting options, but the alternative of superficial crop-end-tillage (without soil perturbation) seemed to be suitable to generate that effect when a legume pasture cover crop or mixed pastures (legume + graminea) were simulated, but not in the case of Ryegrass, especially when a narrow window between the cover crop ending and the beginning of the rice crop was considered.

Regarding the other crops, yield simulations showed a slight underestimation (7%) of average soybean yield, similar to reports by Zhang et al. (2018). For pasture production, dry matter yield was overestimated by 11%. This difference could be explained by the selective grazing of more palatable forage by lambs, tending to degrade the pasture, and which is not accounted for in the model simulations (Cuchillo-Hilario et al., 2017; Rutter, 2006). As expected, the relationship between the predicted and observed N uptake data followed the same trend as the yields. Thus, when observed rice yields showed some degree of deviation from the predicted values, the same occurred for N uptake. However, only the N uptake in the RI-SOY rotation showed a slight underestimation in the predicted N uptake values, and in all cases predicted values followed the observed trend. This gives confidence in the DNDC plant growth simulations.

Although the modelled and measured data did not detect N₂O emissions during the pasture phase of the RI-PAST and RI-CONT rotations, N₂O dynamics in the first rice of the RI-PAST and in the soybean crop of the RI-SOY rotation showed a low to intermediate match between observed and simulated values. However, the small magnitude of average N₂O emission during the rice phase (0.25 kg ha⁻¹) was well predicted, and was similar to other studies in the Uruguayan rice system (Illarze et al., 2018; Irisarri et al., 2012; Tarlera et al., 2006). The main gaseous N loss in the simulations – and by inference in the field, though this was not measured – was from NH₃ volatilization. Continuous observation of NH₃ volatilization is expensive, and it is difficult to accurately capture the low flux of NH₃ volatilization outside fertilization periods (Fillery & Vlek, 1986; Wang et al., 2018). So reliable simulation models are critical. The DNDC model was mainly developed to estimate GHG emissions, and has been widely tested for simulating N₂O and CH₄ in rice systems but less so for NH₃ volatilization (Hu et al., 2011; Nurulhuda et al., 2018). However when used the model represented NH₃ emission trends (Li, 2000; Zhao et al., 2020). In our study, the average rate of volatilization simulated during the rice phase (57 kg N-NH₃ ha⁻¹) was comparable to measured rates in well-fertilized high-yielding rice systems (Liu et al., 2015). Simulated volatilization in the RI-PAST rotation (44 kg N-NH₃ ha⁻¹) was comparable to that measured by Shang et al. (2014) in a rice-legume pasture system with a similar N fertilizer dose (80 kg ha⁻¹). On average,

the simulated volatilization rate for all the rotations in the absence of N fertilizer addition was 15 kg N ha⁻¹, which is in the range of several studies (Liu et al., 2018; 2015; Shang et al., 2014). The difference match the typical figure of 35% of applied N fertilizer volatilized as NH₃ (Liu et al., 2018; Wang et al., 2021). The good match between our predicted and published gaseous N losses gives us confidence to use the predicted N losses to calculate N balances and N efficiency parameters.

2.5.2 Sensitivity of NH₃ volatilization and rice yield to model parameter values

We found that NH₃ volatilization was sensitive to N fertilizer additions and the soil organic C content and, to lesser extents, to soil clay content and pH. Volatilization losses increased in the order RI-SOY > RI-CONT > RI-PAST. This matched the trend in soil NH₄-N concentration at the beginning of each rice crop, which averaged 65, 42 and 28 kg NH₄-N ha⁻¹ (0–20 cm depth) for RI-SOY, RI-CONT and RI-PAST, respectively. These differences were consistent with N fertilizer additions and legume BNF. The effect of soil organic C content on volatilization was similar that of N fertilization rate and was presumably due to N mineralization from organic matter.

The decrease in NH₃ volatilization with soil clay content was presumably because of greater NH₄⁺ retention on soil surfaces, lowering the concentrations of NH₄⁺ in the soil solution and gaseous NH₃ in equilibrium with it (Sommer et al., 2001). The relatively modest effect of the initial soil pH on NH₃ volatilization is explained as follows. The first two N fertilizer applications were applied to dry soil at the rice seeding and tillering stages, but the main NH₃ losses occur after the third fertilizer application at flowering stage when the soil is flooded. The effects of the initial soil pH are small when the soil is flooded because of (a) the moderating effect of the biogeochemical changes following soil flooding, which cause the pH of acid and alkali soils to converge on near neutral, and (b) the dominant effect of the alkalinity released in urea hydrolysis in the floodwater ($\text{CO}(\text{NH}_2)_2 + 3\text{H}_2\text{O} = 2\text{NH}_4^+ + \text{HCO}_3^- + \text{OH}^-$), the floodwater pH being only weakly buffered, independent of the soil pH. Hence the floodwater pH increased from near neutral to pH 9.5 immediately after the third N fertilization.

Rice yield was also more sensitive to N fertilizer addition and soil organic C than to soil clay content and pH, with some differences among rotations. The yield response to N fertilizer was consistent with previous reports for rice in similar systems (Fabini et al., 2021; Macedo et al., 2022).

We made the sensitivity analysis because we lacked measurements of NH_3 volatilization with which to test the model simulations. The finding that modelled volatilization was incentive to the values of the soil parameters, other than the rate of $\text{NH}_4\text{-N}$ addition, gives us confidence that the treatment of volatilization and the associated model coefficients are realistic. We have measurements of all the other components of the system N balance, so this gives us confidence in the overall calculations.

2.5.3 Nitrogen balances of rice-pasture rotations

By combining measured and modelled data, we developed complete N balances for the three rice-pasture rotations considering all N inputs and outputs, with apparently coherent results. As in our earlier evaluation of the rice-livestock system in Uruguay at the national level (Castillo et al., 2021), the RI-PAST system had *NBAL* values around neutrality: $-6 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ in this study and $+2 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ at the national scale. The slightly negative balance was basically explained by greater N removal in animal tissues, with higher stocking rates than at the national scale. The forage base of the experiment comprises sown pastures compared to almost 100% of natural pastures at the national scale. Although there were differences in N inputs and outputs between the studies, the relations between the main inputs and outputs considering the whole rotation were similar. Ratios of N rice grain / N fertilizer, and rice N gas losses / total N inputs were 1.19 vs. 1.23 and 0.30 vs. 0.21 when comparing this study with the former. The rice sector of Uruguay is now facing increasing even the national rice yield and identifies the increase of the N fertilizer as the key to fulfilling this objective. As mentioned in the sensitivity analysis, it is expected a significant increase in the rice yield but leading to generating a more negative *NBAL*. The effect of applying $+22 \text{ kg N ha}^{-1}$ over the usual N dose, will increase the output of N in grain and volatilized in -13 and $-23 \text{ kg ha}^{-1} \text{ yr}^{-1}$, turning the current estimated *NBAL* from -6 to $-20 \text{ kg N ha}^{-1} \text{ yr}^{-1}$.

In spite of the high rice yield and high cropping frequency, the *NBAL* of the RI-CONT system showed more positive values (+45 kg N ha yr⁻¹). This was due to both, more N added as fertilizer (+147 kg N ha yr⁻¹) and from BNF by the legume cover crop (+57 kg N ha yr⁻¹). In this rotation, N fertilization was based on a target rice yield of 10 Mg ha⁻¹ which requires a higher N fertilizer input. However, if the cover crop was more productive, the *NBAL* would have been even more positive due to a greater input from biologically fixed N. That indicates that a reduction of the N fertilizer rate could be proposed to avoid highly positive N balances and correspondingly high *NSURP* values. When discussing the sensitivity analysis was observed that a reduction in the NH₃ volatilization, thus, a reduction in N losses could be attained easily reducing the N fertilizer amount without significative reductions in rice yield. That also will reduce the *NBAL* and *NSURP* of that rotation improving in terms of N performance indices. In the last two years of the long-term experiment, floating macrophyte (*Lemna minor* L.) blooms have been observed during the rice phase of the RI-CONT rotation after receiving higher N inputs (209 kg N ha⁻¹ yr⁻¹) (Alvaro Roel, pers. Communication, see Image A-1 in appendices). Although this phenomenon is not common in the Uruguayan rice system, it is in aquatic environments worldwide with high nutrient load (Goopy & Murray, 2003; Kiage & Walker, 2009). Although this plant is associated with eutrophication processes, it has been proposed for phytoremediation of water bodies due to its exponential growth and ability to sequester N, P and heavy metals (Azeez, 2021; Liu et al., 2017), as well as to reduce NH₃ volatilization from rice field floodwater (Li et al., 2009; Sun et al., 2019).

By contrast, *NBAL* in the RI-SOY rotation was negative (-24 kg N ha yr⁻¹). This was despite inputs from BNF in both cover crop pasture and soybean. For this study, we set N derived from the atmosphere (Ndfa) to the soybean at 75%, based on literature for medium to low fertility soils in Brazil (Alves et al., 2003; Hungria et al., 2006). This compares with Ndfa of 60% from fertile soils in Argentina (Collino et al., 2015) or 50–60% from systems elsewhere (Ciampitti & Salvagiotti, 2018). However, with Ndfa of 75%, the *NBAL* of the soybean was just slightly positive (+ 7 kg N ha⁻¹ yr⁻¹) and the negative *NBAL* of the whole RI-SOY rotation was explained by the negative value during the rice phase. Keeping all other

factors constant, only a theoretical value of 100% of N_{dfa} could generate a neutral $NBAL$ ($-1 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) in RI-SOY. This is important because soybean is an increasingly important element of the rice-pasture rotation in Uruguay and the region (Oficina de Estadísticas Agropecuarias, 2018; Ribas et al., 2021), and farmers favour a continuous rice-soybean rotation due to higher economic margins explained by easier management of the rotation, especially of weeds and reduced labour requirements, as well as higher rice yields. The latter is often associated by farmers with a contribution of soybean N to the rice crop due to positive $NBAL$ and the soil N mineralization of soybean residues with low C to N ratio. But, as in maize-soybean rotations in the US, the positive effect on rice yield seems to be more related to less N immobilization when rice is grown after soybean than after rice or another crop with high C to N ratio, such as maize (Green & Blackmer, 1995). Therefore, the negative $NBAL$ in our results is concerning for this rotation type in the long term. These results agree with previous results obtained from this experimental platform (Macedo et al., 2021), where 19 % less N was found in the particulate soil organic matter in RI-SOY compared with RI-PAST, even at early stages of the long-term experiment. Soil N depletion due to negative balances has been reported in pure rice-soybean rotations (Benintende et al., 2008; Hall et al., 2019; Nishida et al., 2013). In our study, $NBAL$ in RI-SOY was negative even when the rotation included Egyptian clover as a cover-crop. If that cover crop was not considered in the rotation, the $NBAL$ would be even more negative ($-48 \text{ kg N ha}^{-1} \text{ yr}^{-1}$). To avoid this, a better N nutrition from BNF has to be ensured (Collino et al., 2015), particularly by maintaining N fixing pasture species in the rotation (Grahmann et al., 2020). Results from a 6 year study (Landriscini et al., 2019) showed that positive $NBAL$ was achieved in almost all the years and soybean-cover crops treatments. The explanation for the difference compared with our results was the N yields of the used cover crops (118 kg N ha^{-1}), which at least doubled our results. If increasing the rice yield rely only in the addition of more N as fertilizer, no changes in the negative $NBAL$ must be expected based on the above sensitivity analysis. An increase of around $30 \text{ kg N fertilizer ha}^{-1}$ which apparently turns the $NBAL$ of this rotation slightly positive (from -23 to $+7 \text{ kg ha}^{-1} \text{ yr}^{-1}$), will generate an extra N output of around 7 and 19 kg ha^{-1} in grain yield and N volatilization, maintaining the negative $NBAL$.

Associated with the highest positive *NBAL* in RI-CONT was the highest *NSURP*. The EU Nitrogen Expert Panel (2015), defined a threshold *NSURP* of 80 kg N ha⁻¹ yr⁻¹ to avoid environmental damage. In our results, *NSURP* of RI-CONT was 115 kg N ha yr⁻¹ on average, exceeding that threshold considerably. As mentioned for the *NBAL*, considering the amount of N fixed by the pasture cover crop, we think this rotation could reduce the amount of N added as fertilizer by about 30-40 kg N ha⁻¹, without causing yield reductions, and thereby bringing *NSURP* values below the mentioned threshold and moving the *NBAL* value closer to that in RI-PAST.

A clear trend in our results was that, in all three rotations, the higher the *NBAL* and *NSURP* values were, the lower was the resulting *NUE*. A well-managed system should have *NUE* between 50 and 90% (Task Force on Reactive Nitrogen, 2011). As in Castillo et al. (2021), *NUE* of the RI-PAST rotation was in this range. However, the higher N inputs in RI-CONT caused a lower *NUE* of 48%. The opposite was observed in the RI-SOY which reached the highest *NUE* value, close to the upper threshold. Efficiency values around or above this threshold potentially indicate soil N mining and the negative *NBAL* values observed for this rotation.

2.6 Conclusions

The DNDC model successfully simulated crop responses and system N dynamics over the 9 years of the rice rotational systems tested. Though we lacked measurements of NH₃ volatilization, which was the main N loss process, our modelled values agreed with literature values for equivalent systems in all three rotations, giving us confidence that our modelled NH₃ volatilization was realistic. After nine growing seasons of the more N-intensive rotations (RI-CONT and RI-SOY), strongly positive N balance and high surplus N values were apparent in RI-CONT, and negative N balance in RI-SOY. These effects were associated with floodwater macrophyte blooms in the last 2 years of RI-CONT and depletion of soil N in RI-SOY. The DNDC model as parameterised here is suitable for exploring how to optimise N management in these systems, understanding past trends and anticipating future challenges.

2.7 Authorship

Conceptualization: JC, SMH, GJDK. Data acquisition, curation, interpretation, analysis, modelling and drafting of the manuscript: JC. Writing – review & editing: JC, SMH, GJDK, MJR. General discussion and review: JAT, WA, AR, PI, GF. Additional field support: GF. JT; long-term experiment manager. JT, AR: long-term experiment funding acquisition. SMH and GJDK supervision.

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3 WHOLE CHAIN NITROGEN ANALYSIS FOR THE RICE-LIVESTOCK SYSTEM IN URUGUAY*

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3.1 Abstract

Over many decades there has been a global trend away from mixed farming and integrated crop-livestock systems to more-intensive single commodity systems. This has distorted local and global nutrient balances, resulting in environmental pollution as well as soil nutrient depletion. Future food systems should include integrated crop-livestock systems with tight nutrient budgets. For nitrogen (N), detailed understanding of processes, fluxes – including of gaseous forms – and budgets at a component level is needed to design productive systems with high N use efficiency (NUE) across the full nutrient chain. In Uruguay, a unique rice-livestock system has been practiced for over 50 years, attaining a high production level for rice (mean grain yields > 8 Mg ha⁻¹) and an average level for livestock (120 kg liveweight gain ha⁻¹ y⁻¹). The aim of this study was to quantify the components of the N balance and NUE of this system, so as to understand its long-term sustainability, and draw conclusions for other regions. Analysis of country-level statistics for each component over the last 16 years shows tight N balances of + 3.49, + 2.20 and + 2.22 kg N ha⁻¹ yr⁻¹ for rice, livestock and the

whole system, respectively. Based on average values of N retained in edible food products, NUE values were 65.7, 13.2 and 23.1% for rice, livestock and the whole system, respectively. While NUE of livestock was unchanged over the period, NUE of the rice component decreased due to increasing fertilizer use. Further gains in N efficiency are possible by better integrating the system components. Actions to increase system level NUE include raising pasture and livestock productivity and controlling the increasing use of N fertilisers in rice. Tightly integrated crop-livestock systems can play a significant role in re-shaping global agriculture towards meeting food security, environmental and socioeconomic sustainability targets.

3.2 Introduction

The independent Scientific Panel on Responsible Plant Nutrition has recently outlined a food system approach to plant nutrition in which multiple socioeconomic, environmental and health objectives must be achieved (Scientific Panel on Responsible Plant Nutrition, 2020). The fate of nitrogen (N) in crop and livestock production is central to this because of its global impacts on sustainable food production. Nitrogen use efficiency (NUE) – here defined as the ratio of N outputs over N inputs – tends to decline as countries intensify their agriculture through increasing fertiliser use. Over time, as fertiliser management technologies, practices and regulations improve, NUE often rises again. However, the process of overcoming N surpluses is slow and remains a challenge in many parts of the world with intensive agriculture, which has both large economic and environmental consequences (Zhang et al., 2015a; Zhang et al., 2015b). Large regional differences in N budgets and NUE exist, aggravated by transnational nutrient transfers due to the separation of crop and livestock farming, as well as increasing global trade of crops and livestock products (Grote et al., 2008; Uwizye et al., 2020). Many high-income countries outsource much of the environmental burden of their food production to other countries, who must nonetheless ensure that their agricultural systems are both competitive and sustainable (Sun et al., 2020). Uruguay is a prime example of a country producing mostly crops and livestock for export.

On average only 16–20% of N inputs from fertiliser and other sources reach consumed products, with up to 80% lost to the environment in different forms along the production to consumption chain (Sutton et al., 2013; Zhang et al., 2020). In crop production, volatilisation, denitrification and leaching are the major N loss pathways, while volatilization from excreta and leaching are the main pathways in livestock systems (Cameron et al., 2013; Oenema, 2006). Large variations have been found among countries in the full-chain NUE as well as within food supply and consumption chains within a country, depending on food consumption, the structure of the agricultural sector and how tightly crop and livestock production are integrated. In Europe, for example, full-chain NUE ranges from about 10% in Ireland to 40% in Italy (Erisman et al., 2018).

Global market drivers over many decades have caused a separation and concentration of crop and livestock farming, resulting in spatially disconnected nutrient cycles. Livestock is generally seen as having low nutrient use efficiency, high waste and large greenhouse gas (GHG) emissions (Erisman et al., 2018; Uwizeye et al., 2020). However, more sustainable livestock production is possible through pasture-based systems, re-integration of crop and livestock farming, and improved nutrient use efficiency in all parts of the system (Eisler et al., 2014). In such a more-circular farming and food system, livestock are primarily used for what they are good at: converting by-products from the food system and forage resources into valuable food and manure (Van Zanten et al., 2019). An example of such a circular crop-livestock system is the rice-livestock system in Uruguay, which has been in operation for more than five decades (Kanter et al., 2016; Pittelkow et al., 2016). This system has had continuously increasing productivity with low to moderate N fertiliser inputs. There are good records of yields and N budgets at local and national scales, providing an opportunity to understand the changing N balances and use efficiencies in the system over time. We know of no equivalent datasets for other crop-livestock systems globally.

The objectives of this study were to analyse N balances and use efficiencies in the rice-livestock system of Uruguay as a whole and in its components, with a view to drawing generic conclusions relevant to similar crop-livestock systems in other regions. We build on the studies of Pittelkow et al. (2016), which dealt with the rice component of the system, and Kanter et al. (2016), which dealt with the

livestock component. We hypothesised that the system has been following the trajectory outlined above, with initially high NUE but low yields under very low N inputs, and declining NUE and increasing N surpluses as yields increased with increasing N inputs, reaching the current status at which improvements in NUE will require fine-tuning of the system's components at the level of the full nutrient chain. We were also concerned with the system N balance as this reflects the extent to which soil reserves may be mined for nutrients over time, as well as losses from the system, which may be large even though the overall inputs and outputs are in balance. We discuss major performance indicators and entry points for future improvements. Many of our findings are relevant for integrated crop-livestock systems in other parts of the world.

3.3 Materials and Methods

3.3.1 The study area

The Uruguayan rice-livestock system is carried out in 8 of the 19 counties of the country, divided into 3 producer regions. Historically, livestock and rice have been among the top five export products, contributing up to 40% to the agricultural GDP of Uruguay (Simoes & Hidalgo, 2011). The sectors are strongly integrated. Although in general different farmers manage the rice and the livestock, they are bound through agreements on the land that the rice farmer will rent each year. A conceptual scheme of the system is given in **Figure 3-1** with general statistical information in **Table 3-1**. In a given year, on average a total of 8.59 Mha is under livestock-pasture and rice, of which 0.68 Mha is part of the actual rice-pasture rotation, including 0.17 Mha (25%) planted with rice. Nearly 60% of the rice is planted following natural or improved pastures, and the remaining area is planted over stubble of a preceding rice crop. In both cases the land is grazed by livestock before the rice is sown. Further details are provided in Appendix **B.1**.

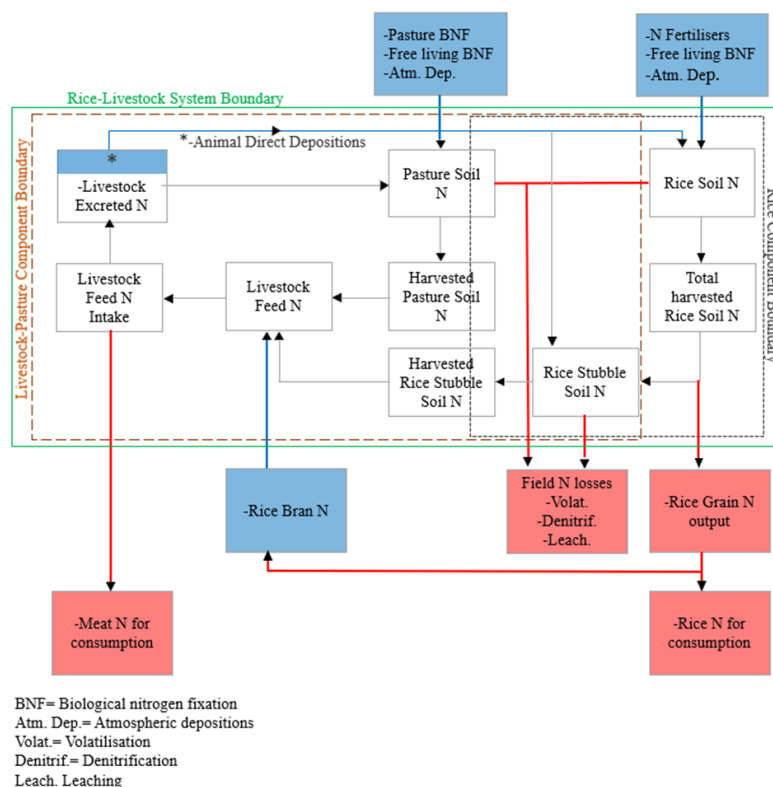


Figure 3-1 Conceptual scheme of the rice-pasture-livestock system of Uruguay. Blue boxes and arrows indicate N inputs, red are the N outputs, and grey arrows show the internal N fluxes.

Table 3-1 Components of the rice-livestock system of Uruguay. Values are averages and ranges of the 2004/05 to 2019/20 growing seasons.

Parameter	Mean	Range	Units
Area of rice harvested annually	0.17	0.14 – 0.19	Mha
Area of natural pasture	7.56	7.35 – 7.88	Mha
Area of improved pasture	0.86	0.71 – 0.99	Mha
Area of pasture linked to rice*	0.68	0.56 – 0.76	Mha
Annual seeded past. renewable rate	16.3	14.2 - 18.7	%
Rice : rotation ratio	1:4.0	1:2.5 – 1:4.5	-
Improved pastures duration	3.8	3.0 – 4.5	years
Beef livestock numbers	6.26	5.96 - 6.66	heads × 10 ⁶
Sheep livestock numbers	5.54	4.24 - 7.21	heads × 10 ⁶
Stock density (bovine + ovine)	0.86	0.81 – 0.89	heads ha ⁻¹

*Estimated area of natural plus improved pastures directly linked to the rice area based on García et al. (2009)

3.3.2 Scope of the study and data sources

We analysed data of rice, forage and livestock production at a national level from 2004 to 2020. We took data for the entire country separated by component, production system and food production from the Agricultural and Livestock Ministry (MGAP), the Agricultural Statistics Department (DIEA), the National Institute of Meat (INAC), the National Institute for Agricultural Research (INIA) and other literature (**Table B-1**). For the rice component, consistent data from the rice milling industry are available at a farm scale for the whole country, while data for livestock and pasture production are only available at a county scale, which means that the statistics footprint area of the latter exceeds the typical 4:1 pasture: rice area ratio in (**Table 3-1**). Hence, actual total pasture area for the livestock calculations was on average 8.42 Mha, of which around 0.68 Mha was directly associated with rice (based on García et al., 2009). Despite that disparity in the scale of the data sources used, we assume that the means and general variation derived from the much larger livestock are representative for the livestock area that is directly part of the rice-pasture rotation. For example, calculated values for cattle stocking rate ($0.79 \text{ livestock units ha}^{-1}$) and the percentage of temporary pastureland (6.3%) were similar to reported values for the main rice-livestock region of the country ($0.76 \text{ livestock units ha}^{-1}$ and 8% respectively; Simeone et al., 2008). For the computation of the rice-livestock system, land dedicated to forest production, horticulture and other crops in each county was not considered, but coverage of these uses was on average no greater than 1.5% of the total area of the counties with rice-livestock systems.

3.3.3 Rice

Data from rice farmers, the rice growers' association, the rice milling industry and governmental agencies are published annually. Information on rice area and yield at county and regional scale were obtained for 2004 to 2020 (DIEA, 2005; 2020). This information covers basically all the rice farmers in the country. Data on N inputs from mineral fertiliser were sourced from annual reports of the rice milling industry summarized for the National Institute of Agricultural Research (INIA, 2020). Detailed records of fertiliser type and doses are collected by the agronomic technical teams of each of the six main rice mills. The rice area serviced by these companies covers 85-90% of the total rice area. The rice varieties grown are

largely from the rice breeding program of INIA, and crop parameters, such as grain N concentration (g N kg^{-1} grain), were taken from internal records of INIA (Deambrosi et al., 2019). Because nearly 75% of the exported or internally consumed rice is processed white rice (OEC, 2020), virtually all the bran obtained after milling (except paddy, cargo, parboiled and seed rice batches) is returned to the livestock component as animal feed. Based on Bodie et al. (2019), we assumed 11% of bran in the whole grain and 16% of protein in the bran.

3.3.4 Pasture

The forage base includes natural grassland (89.5% of area), semi-natural pastureland (4.2%) and temporary pastureland (6.3%), based on Allen et al. (2011) classification, all without N fertilisation. While natural grassland largely contains native grass species, semi-natural and temporary pasture have a mix of improved legumes (*Trifolium spp.* and/or *Lotus spp.*) and grasses (*Festuca spp.* or *Lolium spp.*), differing in productivity and duration. When legumes are included in the forage mix, at least one session of phosphorus fertilization is frequently made (Appendix B.1). We used annual evaluation of the natural grasslands productivity based on remote sensing data of 16 years for the main ecological regions of the country (Asuaga et al., 2019). Similarly, information for the semi-natural pastureland component was taken from Martínez (2011) over 10 years. Productivity of the temporary pastureland component was estimated based on a 7-year pasture database from a local long-term experiment which rotates rice and pastures at INIA facilities (unpublished). Henceforth, we will refer to the semi-natural and temporary pastures as improved pastures. We estimate annual herbage production by regression analyses (Appendix B.2, Figure B-1). Nitrogen availability of the different forages was estimated based on annual dry matter production, botanical composition and N content of each plant species (Carámbula, 2003), and forage utilisation and livestock N intake calculations followed Crempien (1983) (Table B-1).

3.3.5 Livestock

Livestock data were obtained for the period 2004 to 2020 (DICOSE, 2004; INAC, 2020; SNIG, 2020). This includes monthly reports of cattle and sheep sent to the abattoir, with animal category, live weight and kill-out percentage at a county and

regional level. This information was used to calculate the meat production and N accumulated in the animals. Data from DICOSE includes all livestock categories at each farm of each county, including self-consumption on an annual basis. For this study, apart from the livestock sold to the abattoir, we included the farm self-consumption data in the meat production and N animal retention calculations. For sheep production, we also considered an average value of wool production per animal (kg per animal) in the N balance. We used international published values of N content of meat (beef and sheep meat) and wool. Wool was included together with bovine and ovine meat production under the equivalent meat concept (FAO, 2018). The amount of N recycled by livestock was defined as a function of the animal type, pasture botanical composition and herbage dry matter mass, forage utilisation (pastures or rice straw), and animal NUE (Appendix **B.2**, **Figure B-1**).

3.3.6 Data analysis

We calculated a simple N balance based on N inputs minus N outputs, and NUE from outputs relative to inputs using a common N budgeting approach (Watson & Atkinson, 1999). All N inputs and outputs were evaluated within the rice-livestock system boundary and for each component separately (details in Appendix **B.1**, **B.1.1**). For the rice component, N inputs were fertilisers, atmospheric deposition, biological fixation, and the direct animal deposition (faeces plus urine from the 6 months of grazing prior to land preparation for rice or to un-grazed fallow); outputs were rice grain, volatilisation, denitrification and leaching. For the livestock component, N inputs were atmospheric deposition, biological fixation and rice bran as animal feed; outputs were as for the rice and animal products. Nitrogen inputs from biological fixation in pastures and rice bran in the livestock component were calculated over 0.86 Mha and 0.17 Mha respectively and assigned to the total grazing livestock area (8.42 Mha). We did not have data on changes in the soil N pools so inferred these from the balance for each component; for the whole system they are considered as internal processes and not part of the balance. For the NUE of the livestock, we considered N in rice bran returned as livestock feed as an N input. National laws prohibit use of animal by-products as feed. At a system level, animal direct deposition was considered to be internally recycled.

We then evaluated the NUE trajectory against defined low and high thresholds with a desirable N output in food products. This allows us to link the results to potential N losses. For the rice sector, we defined the upper and lower NUE thresholds as < 90 and $> 50\%$ (EU Nitrogen Expert Panel, 2015; Task Force on Reactive Nitrogen, 2011), and a minimum target N output of $80 \text{ kg grain N ha}^{-1} \text{ yr}^{-1}$ which corresponds to the current average rice grain yield (8 Mg ha^{-1} at 13% moisture content and 1.15% N by dry weight) of similar productive systems in South America (Singh et al., 2017). Finally, we defined $80 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ as a maximum surplus (EU Nitrogen Expert Panel, 2015) for the rice component. The term surplus is related to the potential N losses and was calculated as N inputs minus N in food products. Corresponding livestock thresholds were defined. The high and low NUE limits were defined as < 25 and $> 10\%$ based on Gerber et al. (2014). A target output of $3 \text{ kg ha}^{-1} \text{ yr}^{-1}$ as animal N in edible food from extensive grazing systems was defined based on Oenema et al. (2016). A similar value can be derived from Kanter et al. (2016) as a potential value for the Uruguayan livestock sector. Because Uruguay is characterized as a net commodity product exporter with very low food imports, no feed imports and low product industrialisation, the boundary for all the analyses was defined as the farm gate.

3.3.7 Partial sensitivity analysis

We do not have enough data on uncertainties of all the input and output variables to make a full sensitivity analysis, but we illustrated the range of uncertainties by computing a partial sensitivity analysis as follows: for the rice component a) maintaining constant N outputs in grain, b) increasing the values of N volatilised from 20 to 40% of the N applied, denitrified from near 1 to 3 kg N ha^{-1} , and leached from 2 to 5 kg N ha^{-1} during the rice season based on data from irrigated rice systems with fertiliser inputs $> 180 \text{ kg N ha}^{-1}$ (Jian-she et al., 2011; Xu et al., 2012); for the livestock component, a) considering the same sources of losses and the amount of N fixed by pastures which were at least doubled up to 12, 2 and 10 kg N ha^{-1} for volatilisation, denitrification and leaching respectively, and b) 78 kg N ha^{-1} for biological fixation (11 kg N ha^{-1} considering 8.42 Mha), based on intensive livestock systems without fertiliser N addition according to the Uruguayan example (FAO, 2018b; Sagggar et al., 2013).

3.4 Results

3.4.1 Nitrogen balance

Here we present the N balance of the entire rice-livestock system and its components averaged over the 16 year period from 2004 to 2020. On average, total N inputs to rice were $101.1 \text{ kg ha}^{-1} \text{ yr}^{-1}$ of which 66.7, 25.9 and $8.6 \text{ kg ha}^{-1} \text{ yr}^{-1}$ were fertiliser inputs, N transferred via animal direct deposition from the livestock-pasture component to the rice component, and biological N fixation plus atmospheric deposition, respectively (**Figure 3-2 a**). For the livestock-pasture component, N inputs were from atmospheric deposition ($6.2 \text{ kg ha}^{-1} \text{ yr}^{-1}$), legume pasture biological N fixation ($6.8 \text{ kg ha}^{-1} \text{ yr}^{-1}$) and the rice bran used for livestock nutrition ($0.4 \text{ kg ha}^{-1} \text{ yr}^{-1}$) (**Figure 3-2 b**) totalling on average $13.4 \text{ kg ha}^{-1} \text{ yr}^{-1}$. The amount of inputs from fixed N by pastures and rice bran appears to be quite low once it was assigned to the total livestock area (8.42 Mha). Nevertheless, the real values for these inputs were $46 \text{ kg ha}^{-1} \text{ yr}^{-1}$ for N fixed (over 0.86 Mha) and $19 \text{ kg ha}^{-1} \text{ yr}^{-1}$ of bran availability (over 0.17 Mha). Combining both components, annual average N input to the whole system was around $14.6 \text{ kg ha}^{-1} \text{ yr}^{-1}$ (**Figure 3-2 c**).

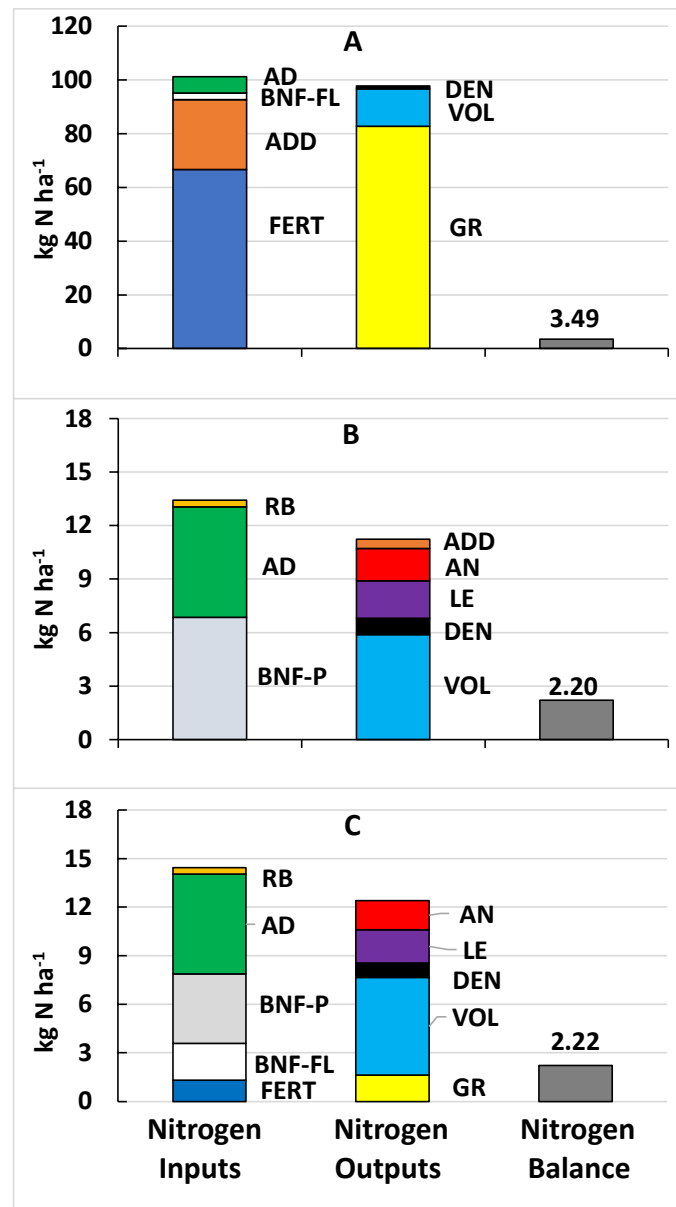


Figure 3-2 Simple N balance of the rice-pasture-livestock system: (A) the rice component, (B) the pasture-livestock component, (C) the whole system. Values are averaged over 2004/5 to 2019/20. Inputs: FERT = fertilisers, ADD = animal deposition, BNF-FL = biological N fixation by free living microorganisms, AD = atmospheric deposition, BNF-P = biological fixation by pastures, RB = rice bran. Outputs: GR = N rice grain, VOL = volatilization, DEN = denitrification, LE = leaching, AN = N whole animal.

Total rice N output was on average 97.6 kg ha⁻¹ yr⁻¹, mainly due to the large removal of N in rice grain (82.8 kg ha⁻¹ yr⁻¹) and N volatilisation plus denitrification (14.9 kg ha⁻¹ yr⁻¹). Outputs of the livestock-pasture component were much smaller: 11.2 kg ha⁻¹ yr⁻¹, of which N associated with losses (8.9 kg ha⁻¹ yr⁻¹) was greater than that in animal products (1.8 kg ha⁻¹ yr⁻¹). The N transferred via animal deposition to rice was small (0.5 kg N ha⁻¹ yr⁻¹ on 8.42 Mha). Outputs in crop and

animal products plus losses in the entire system were around $12.4 \text{ kg ha}^{-1} \text{ N yr}^{-1}$ on an annual base.

Overall, the average rice-livestock system N balance for the whole country was slightly positive during the study period ($+2.22 \text{ kg ha}^{-1} \text{ yr}^{-1}$). That value was closely related to the positive pasture-livestock component balance result of $+2.20 \text{ kg ha}^{-1} \text{ yr}^{-1}$ over 8.42 M ha, but slightly increased by the positive rice component balance ($+3.49 \text{ kg ha}^{-1} \text{ yr}^{-1}$) in a smaller area (0.17 M ha).

3.4.2 Full Chain-NUE and N surplus

3.4.2.1 Rice Component

On average, the full chain-NUE (N recovered in the rice grain relative to total annual N inputs) was 65.7%. When the NUE was plotted throughout the studied period, 81% of the records (except for the years 2004, 2005 and 2009) achieved optimal NUE values between the defined thresholds and exceeded the target for a desirable N output in rice edible food ($80 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) (**Figure 3-3 a**). The years when the output target was not met corresponded with relatively high rainfall (INIA, 2021) prior to sowing or at the heading stage, which caused lower yields due to delayed seeding and low solar radiation. Despite the good NUE values attained, increase in N fertiliser addition (Section 3.4.1) shifted NUE towards the lower threshold ($\leq 50\%$) in the later years (**Figure 3-3 a**). This is due to reduced N transfer into grain and N losses to the environment. Thus, there was a negative correlation ($r = -0.96$) between NUE and N surplus values so that when NUE was maximum, the N surplus was minimum and vice versa (**Figure 3-4 a**).

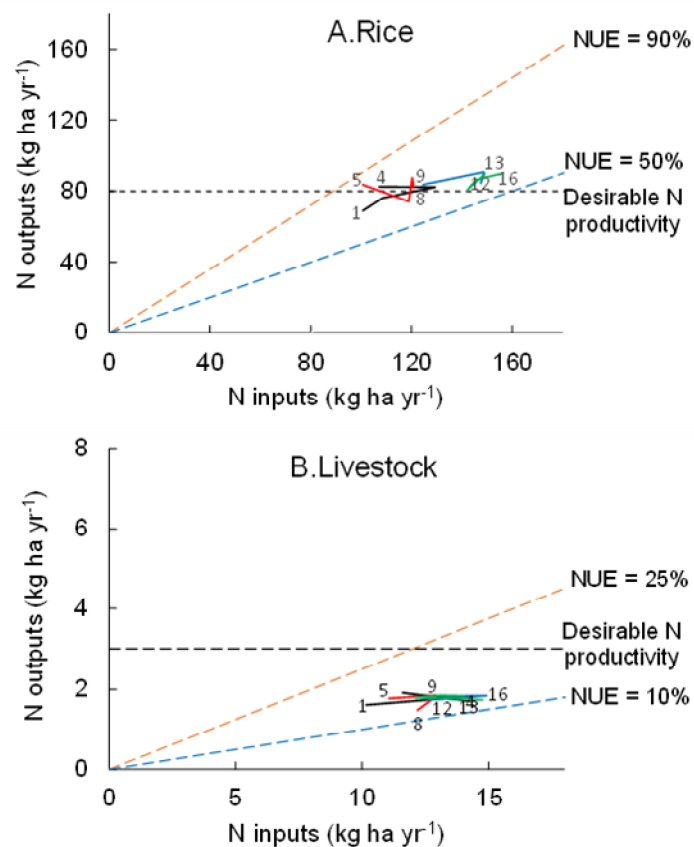


Figure 3-3 Changes in N outputs versus inputs from the 2004/5 to the 2019/20 growing seasons: (A) rice component, (B) livestock component. Solid black, red, blue and green lines indicate changes from the 1st to 4th, 5th to 8th, 9th to 12th and 13th to 16th years of the series, respectively. Dashed orange and blue lines indicate NUE (= outputs / inputs × 100) of 90 and 50%, respectively for rice and 25 and 10%, respectively for livestock. Dashed black lines indicate the expected N output for a desirable level of production.

3.4.2.2 Livestock Component

The expected NUE of the livestock component is low due to a much smaller N output in animal edible products in this extensively managed, grazing-based livestock system. On average, livestock NUE was 13.2% and annual values were between the defined upper and lower NUE boundaries, but below the desirable minimum N productivity output of 3 kg ha⁻¹ yr⁻¹. Similar to the rice component, NUE tended to decrease in later years of the study, due to plateauing N outputs in meat and a slight increase of N inputs in the last 4 years (**Figure 3-3 b**) due to a greater area of improved pastures with N fixing legumes. Nonetheless, the changes in NUE and N surplus were smaller than in the rice component and were steady over the study period (**Figure 3-4 a**).

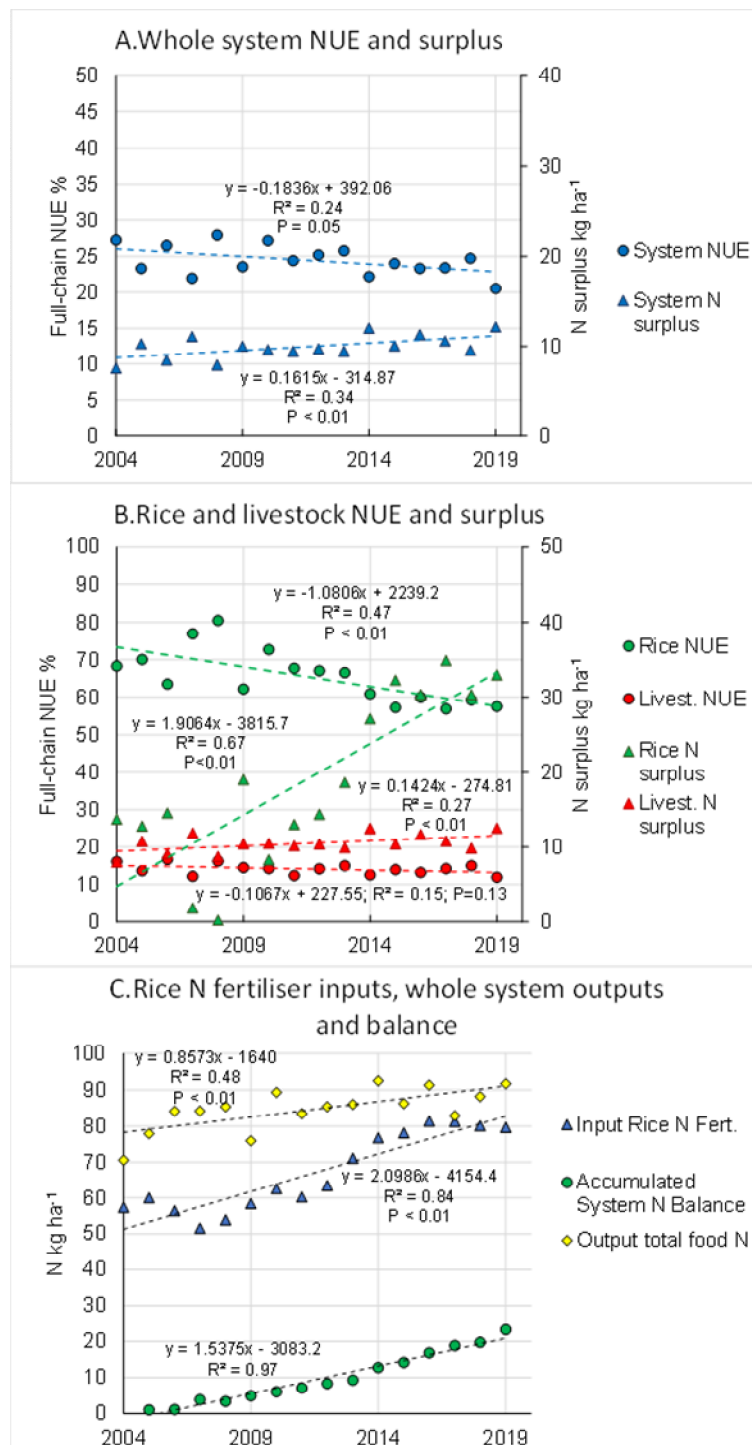


Figure 3-4 Changes from 2004/5 to 2019/20 in NUE (= outputs / inputs × 100) and N surplus (= inputs – outputs) in (A.) the whole system and (B) the rice and livestock components; (C) N inputs, outputs and balance in the whole system.

3.4.2.3 Rice-livestock system

Over 16 years, the mean value of N retained in both food products for the entire system was 23.1%. As shown for each component, system NUE decreased over time (-11.4%) (**Figure 3-4 a**), close to what was observed for the livestock

component (-11.7%) but smaller than the fall in rice NUE (-22.0%) (**Figure 3-4 b**). The N surplus at a system level remained almost unchanged at low levels (9.9 kg ha⁻¹ yr⁻¹ on average) although this value was mainly influenced by the low N surplus in the livestock component (10.5 kg ha⁻¹ yr⁻¹) compared with the rice component, which experienced an increase of 28.6 kg in the N surplus over the period. A positive balance in the rice component due to an upward trend in N fertiliser added (+31.5 kg N ha⁻¹ from 2004 to 2020), greater than the increase in yield and N output in rice grain (+12.8 kg ha⁻¹), explained increasing N surplus as well as the change in the N accumulated balance from -1.29 kg ha⁻¹ to +38.6 kg ha⁻¹ (**Figure 3-4 c**).

3.5 Discussion

The Uruguayan rice-livestock system is highly integrated within regions and production cycles, combining one of the world's most productive rice systems (yields > 8 Mg ha⁻¹) with grazing-based average livestock productivity for meat. The low average N fertiliser inputs (67 kg ha⁻¹ yr⁻¹), accounting for 66% of total rice N inputs and 81% of grain N, might mean negative soil N balances and mining of soil N over time. However, we found that substantial amounts of N are contributed by animal deposition, accounting for 26% and 31% of total rice inputs and grain removal, respectively. Similarly, in other mixed farming systems, over 20% of the N for crop production comes from livestock (Liu et al., 2010).

At the beginning of the period studied here, some years were close to the high NUE threshold ($\geq 90\%$), which could raise concerns about sustaining soil health. However, the system has been in operation for 50 years with no signs of land degradation or declining yields. Country-average rice yields have increased from < 5 Mg ha⁻¹ in the early 1990s to about 8 Mg ha⁻¹ in recent years, with some farmers reaching 10 Mg ha⁻¹ or more (Tseng et al., 2020). It has also been shown that soil organic carbon is maintained under rice rotations that include improved pastures (Deambrosi, 2009). However, the inclusion of improved pastures in the rotation, or a higher crop intensity vs pasture length, could alter the amounts of nutrients recycled by livestock, and could also change the current near-neutral N balance (+3.49 kg ha⁻¹ yr⁻¹). Overall, and strongly influenced by the livestock sector, the entire system has maintained an adequate balance between N inputs

and outputs, resulting in very low N surpluses and no noticeable deterioration of soil health.

Assuming the proposed enlarged values of losses mentioned for the partial sensitivity analysis, N balance would reach $-2 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ for the rice system and less than $-10 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ for the livestock and whole system respectively. That means that, even if our calculations under- or over-estimate the losses or N inputs or both, the magnitude of the balance suggests that soil N changes would be less than detection limits. Because the rice and livestock are produced for export, it is important that production can be shown to be environmentally benign. For this, improving records, particularly of N losses from the system, will be important.

Despite an upward trend in rice yield, the recent increase in N fertiliser application to rice (Pittelkow et al., 2016) has decreased the system NUE and shifted the N balance slightly towards surplus. Our results suggest that the Uruguayan rice-livestock system is approaching the end of the first stage of this process with still increasing fertilisation rates. This can be observed in the NUE of the most recent years in our time series, when the relation between inputs and outputs approached the lower threshold of NUE. Despite the recent decrease in NUE, the average for rice during the entire 2004–2020 period was high (65.7%) compared with pure upland crop systems (Jarvis et al., 2011), systems including N fixing crops or pastures (Godinot et al., 2016) or rice systems in Asia (Singh et al., 2017), and close to a desirable future target of 70% (Scientific Panel on Responsible Plant Nutrition, 2020). In addition, this high average NUE is associated with an average N surplus of only around $20 \text{ kg ha}^{-1} \text{ yr}^{-1}$, still far from the defined upper threshold ($80 \text{ kg ha}^{-1} \text{ yr}^{-1}$). This is contrary to findings for other systems where high yields and NUE may still result in relatively high N surpluses (Silva et al., 2021).

Because livestock N inputs and outputs were more constant over the period analysed, NUE and N surplus tended to plateau. The average NUE (13.2%) was in the range of published values internationally (Gerber et al., 2014; Jarvis et al., 2011). At a system level, NUE and surplus values were closer to those for the livestock component because of its greater extent than the rice. Although NUE

comparisons among systems may be confounded by differences in methods or defined boundaries, the NUE of this rice-livestock system (23.1%) is only slightly higher than values reported for other systems (Galloway & Cowling, 2002; Howarth et al., 2002). In integrated systems, NUE values between 35% (Godinot et al., 2015) and 45% (Westhoek et al., 2014) are attained only when the proportion of the crop component greatly exceeds the livestock.

A key question is whether the system's NUE could be improved further whilst preserving the low N surplus. Should efforts be concentrated on the rice component which has already achieved high NUE and capture of N, or on the low-production and low-efficiency livestock? The rice sector is now making efforts to move the national yield average from 8.3 to 10 Mg ha⁻¹ yr⁻¹, which would approach the attainable yield potential. Reaching this ambitious goal would require a further increase in fertiliser-N use by 40–50 kg N ha⁻¹ for recent high-yielding rice varieties, leading to a further decline in NUE and an increase in N surplus. Hence, efforts to close the current rice yield gap must pay close attention to more precise N management with location-specific rates, splitting, timing and forms of N applied.

Kanter et al. (2016) proposed a realistic set of management improvements to the Uruguayan livestock sector to increase beef productivity by 25%. Two of the main interventions suggested are the doubling of the area of improved pastures, including more legumes in the pasture, and a modest increase in supplements fed to livestock. Both would allow an increase in the live-weight gain and a reduction in the age of slaughtering (Rivero et al., 2021). The N footprint would decrease from about 66 to 48 kg N lost per kg live weight per year, resulting in a 30% reduction of N losses from the beef sector (Kanter et al. 2016). This increase of 25% in the animal productivity would improve the NUE of the livestock component from 13.2 to 14.3%. If part of the proposed increase in improved pasture area (+0.5 Mha) were assigned to the area left by the last rice of the rotation (0.1 Mha), an increase of the country's average rice yield and a lower N fertiliser requirement would be expected (Carlos et al., 2020). In addition, the expected greater production of rice bran could offset the extra animal feed supplement suggested by (Kanter et al., 2016).

As regards improvement of soil fertility for pasture production, typical current practice is to seed the pasture over rice stubble after the floodwater is removed, taking advantage of any residual P (Gamarra, 1996), and to add 22-26 kg P ha⁻¹ at the beginning of the second year of the pasture. When pasture is seeded outside the rice rotation, a similar amount of P is added at the time the pasture is sown. However, P fertilisation in natural pastures has not been shown to increase forage productivity, and causes loss of species diversity (Cardozo et al., 2017; Jaurena et al., 2021). Likewise, the main soils in the Uruguayan rice-livestock areas do not benefit from lime applications (Fedrigo et al., 2021). The proposed degree of improvement of pastures is consistent with extensive livestock production under the prevailing warm and dry climate, and a low level of investment compared with more-intensive systems.

Are improvements of the pasture-livestock sector economically viable? Assuming the average length of an improved pasture is 4 years, the average cost of establishment and maintenance are approximately US\$ 45 ha⁻¹ yr⁻¹. On the other hand, using the average animal productivity of this system (Simeone et al., 2008) and considering current meat values (INAC, 2020), an increase of 25% in gross animal production could result in US\$ 50-60 ha⁻¹ yr⁻¹ of extra income, indicating that the proposal could be economically viable. In addition, an improvement in the rice component productivity is expected which would further increase profitability.

How could such changes be brought about? Both system components are strongly vertically integrated through industry, government, research, extension, and farmer associations. However, the key drivers of change must be the livestock farmers. The beef production model developed by Soares de Lima (2009) could be used to guide increasing beef productivity, including doubling the percentage of sown pastures and other management changes. Complexity has been identified as a critical issue limiting integration (Maletti et al., 2014). Asai et al. (2018) summarize various examples of integration between crops and livestock spanning organic crops and vegetables to rice in France, the USA, the Netherlands, and Japan. In all cases, developing strategies that reduce the costs associated with goods transactions and having technical support to maintain farmer networks are mentioned as a priority. In Brazil, a new integration scheme

among crops, livestock and forestry for low carbon agriculture has been recently introduced, aiming to recuperate degraded pastures and mitigate soil degradation (Garrett et al., 2020). For rice, research has recently evaluated the benefits of shifting from monocropping rice to an integrated rice-livestock-soybean rotation in terms of productivity and fertilisers savings (Carlos et al., 2020). Regardless of the system, once separated, return to integration is difficult because the relevant skills and knowledge have often been lost (Martin et al., 2016). Thus, the Uruguayan rice-livestock system could become a reference for similar systems worldwide.

3.6 Conclusions

1. In spite of only modest mineral fertiliser N inputs, rice yields reached high levels and livestock production moderate levels, and the N balance of the whole system remained positive throughout the period analysed. This reflects the tight integration of the rice and livestock components of the system at local and national scales. Nonetheless, N losses from the system were substantial and could be reduced.
2. Changes in NUE from 2004 to 2020 followed the expected trajectory in rice and to a lesser extent in livestock, tending to decrease over time as N inputs increased. As NUE decreased, N surpluses increased in both rice and livestock, leading to greater losses, mainly through volatilization.
3. Actions to increase system level NUE include raising livestock productivity, increased area of improved pastures and better management of the increasing applications of N fertiliser to rice.
4. The Uruguayan rice-livestock system demonstrates how crop and livestock farming can be successfully integrated to achieve multiple productivity and sustainability goals at the level of a whole country. The principles and mechanisms employed in it could also be applied in other countries and different forms of crop-livestock farming.

3.7 Authorship

Conceptualization: JC, SMH, AD, GJDK. Data acquisition, curation, interpretation, analysis, modelling and drafting of the manuscript: JC. Writing –

review & editing: JC, SMH, GJDK, MJR. Critical revision of the manuscript for important intellectual content: AD. Supervision SMH and GJDK.

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4 REGIONAL ANALYSIS OF NITROGEN BALANCE, USE EFFICIENCY AND SURPLUS FOR THE URUGUAYAN RICE-LIVESTOCK SYSTEM*

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4.1 Abstract

The reconnection of crops with livestock systems is proposed as a way of sustainably intensifying food production globally with reduced environmental impacts, particularly those involving nitrogen (N). Detailed understanding of processes governing N fluxes and budgets is needed to design productive and efficient systems. The aim of this study was to investigate regional differences in N balance (NBAL, defined as all N inputs minus outputs), use efficiency (NUE, defined as N outputs / inputs × 100) and surplus (NSURP, defined as all N inputs minus only outputs in food products) in the rice-livestock system of Uruguay over the last 18 years. Three regions across Uruguay are distinguished based on soil fertility and length of pasture rotation. The northern region has high soil fertility and short length of rotation (HFSR); the central region has medium soil fertility and medium length of rotation (MFMR); the eastern region has low fertility and long pasture rotation (LFLR). Results show a very high NUE (90%) for the rice component in all rotations, associated with negative NBALs ranging from -35 kg N ha⁻¹ yr⁻¹ in HFSR to -3 kg N ha⁻¹ yr⁻¹ in LFLR. However, the livestock component,

which overall had a low productivity in animal food products ($< 2 \text{ kg N ha}^{-1} \text{ yr}^{-1}$), had low NUE ($< 10\%$) but positive NBALs in all the rotations, sustaining N supply in the rice component. At the system level, NUE was high (60%) and NBAL was slightly positive in all rotations (from $+ 2.8 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ in HFSR to $+8.5 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ in LFLR). Because of a recent increase in the fertilizer N dose in rice, NSURP was intermediate ($40 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) and should be monitored in the future. Even with the very good N parameters values attained, there is an opportunity to improve the system efficiency by improving the livestock component.

4.2 Introduction

Over the past many decades, production systems in most parts of the world have adapted to growing global food demand and changes in diets by specialization (Lassaletta et al., 2014; Russelle et al., 2007). Specialized systems frequently rely on large amounts of external inputs, of which fertilizers, particularly N, play a key role, causing environmental damage including a major contribution to global greenhouse gas (GHG) emissions (Galloway et al., 2008; Hilimire, 2011). Such negative externalities are due to low N use efficiency (NUE), with on average $< 20\text{--}25\%$ of N inputs reaching the final consumable product (Sutton et al., 2013; Zhang, 2020). In general, pure crop systems have higher NUE than livestock systems, which are associated with high animal waste and GHG emissions (Uwizeye et al., 2020). Specialization has broken a virtuous circle between livestock and crops whereby the forage, fibre and grains for animal feed were provided by cropping whilst nutrients and organic matter were provided to crops from animals (Thorne, 2007; Wolfe, 2011). A return to integrated crop-livestock production systems is increasingly discussed as a way of achieving high production whilst avoiding the negative externalities of specialized systems (Baiyeri et al., 2019; Peterson et al., 2020; Vogel et al., 2021).

There are many variants of integrated crop-livestock systems, from those managed in separate farms but sharing by-products and residues, to those in which crops and animals are in the same farm, sometimes in rotation on the same land (Brewer & Gaudin, 2020; Garrett et al., 2017; Wolfe, 2011). But in all cases, regardless of the degree of integration, the common denominator is the use of

animals for what they are good at: converting fibrous feeds (e.g., forage) and by-products from the food system into high value products and manure (Van Zanten et al., 2019), allowing a circular farming system (Velasco-Muñoz et al., 2021).

The Uruguayan rice-livestock system is an example of a circular farming system, with the whole country rice area integrated in a pasture rotational scheme (García et al., 2009; Lanfranco et al., 2018). The system has been operated for four to six decades depending on the region, with a constant yield increase over time of 90 kg ha⁻¹ yr⁻¹ (Blanco et al., 2010) and with relatively low use of N fertilizers (Tseng et al., 2021). In an earlier study (Castillo et al., 2021), we analyzed the system at a national level, and found complementarity through N transfer from animal deposition to rice, with biological N fixation during the pasture phase, and through N recycling in rice bran to livestock. We found N balances are tight (< 3.5 kg N ha yr⁻¹ in both components and the system), and N surpluses low but increasing, with N use efficiency reaching high values for rice (65%) but much lower for livestock (13%) and the system (23%). National rice yields of 10 Mg ha⁻¹ are now targeted by farmers, potentially requiring more N fertilizer. Over time, this could lead to a decline in NUE and potentially increased N surplus up to undesirable values (Dobermann et al., 2022). At that point, adjustments in fertilizer technology and regulations would be needed. And despite a fairly uniform rice crop management at national level, there are regional differences across the rice-livestock system, mainly in terms of length of pasture rotations and the level of natural soil fertility. These are likely be linked to differences in NUE and N surplus and their progression over time, which needs to be understood to design strategies for improving the overall system.

Our objectives were to assess N balances and NUE and their components in rice-livestock rotations across Uruguay and follow their changes from 2004 to 2022. Based on our national scale assessment (Castillo et al., 2021), we hypothesized that even with relatively small N fertilizer additions to rice, the NBAL has been around neutrality while NUE reached high values in all the rotations over the period investigated. But we expected differences among rotations due to different pasture lengths and management practices. Because of small N outputs in animal products, we hypothesized that the livestock component reached positive and stable NBALs, and medium to low NUE across the period, resulting in positive

NBALs and medium NUE in the whole system. We also explored different production scenarios to identify the more sensitive aspects of NBAL and NUE for improving management practices.

4.3 Material and Methods

4.3.1 Overview

Uruguay is a net agricultural exporting country with more than 80% of the total area (16 M ha) devoted to agriculture (World Bank, 2018). Livestock production is the main economic export activity (Simoes & Hidalgo, 2011). Livestock production is supported by permanent perennial native pastures across the country, of which around 50% is a pure and extensive pasture-livestock system while the rest is integrated into upland crop rotations, forest plantations, or rice (Oficina de Estadísticas Agropecuarias, 2018). While livestock production is carried out across the whole country, upland agriculture is mainly developed in the western region where better soil properties (i.e., fertility, soil water availability and internal drainage) allow a permanent winter-summer crop rotational arrangement, occasionally including pastures and livestock. Rice cultivation, on the other hand, is strictly integrated into a rotation with livestock and pastures for direct grazing, with varying length of the rotation according to the different producer regions.

4.3.2 Cropping system characteristics

The rice-livestock system consists of around 163,000 ha of rice and 570,000 ha of pastures integrated into a stable rotation divided in three regions (**Table 4-1**). The main region is in the east accounting for 70% of the national rice area. The remaining area is distributed between the northern (20%) and the central (10%) regions. The eastern region is characterized by a flat landscape with slopes around 0.1%, medium to low soil fertility and river water sources for flood irrigation. In the northern and central regions, rice is grown on more fertile soils, which includes sloped areas of less than < 5% (nearly 60% in the north and 25% in the central region) and irrigation water is sourced from artificial dams. Despite those particularities, the main differences among the regions are soil fertility and pasture phase length after rice. On average, after 2 or 3 consecutive rice crops (the latter mainly associated with the northern region), 4, 3 or 2 years of perennial

pastures grazed by livestock complete the rotation in the eastern, central and northern regions, respectively (García et al., 2009; Giménez et al., 2011; Lanfranco et al., 2018). Based on the soil fertility and pasture length differences among regions, we will refer to the east region as low to medium soil fertility and long rice-pasture rotation (LFLR), the central as medium soil fertility and medium rice-pasture rotation (MFMR), and the north as high soil fertility and short rice-pasture rotation (HFSR), respectively (**Table 4-1**). Therefore, from here on, the rotation type is synonymous with the respective region. Following rice crops, around 31% of HFSR, 33% of MFMR and 38% of LFLR are mixed pastures including legume species, seeded into the rice stubble. The combination of these factors means that the ratios of rice seeded into (a) rice stubble, (b) improved pastures including legumes or (c) native grassland are 60-17-23 for HFSR, 50-18-32 for MFMR and 35-21-44% for LFLR. But even when considering the differences in rotation length and pre-rice crop each year, few differences in crop management and the amount of fertilizer and agrochemical products added are found. Despite rice and livestock farmers being different actors, the system as a whole is strong and stable, and based on land agreements in which the rice farmers rent land for long periods or on an annual basis. Complementary information about the system integration was presented in Castillo et al. (2021).

4.3.3 Scope of the study and sources of information

We analysed data of rice, forage and livestock production at a regional level from the 2004–05 to 2021–22 growing seasons. Data from the Agricultural and Livestock Ministry (MGAP), the Agricultural Statistics Department (DIEA), the National Institute of Meat (INAC), the National Institute for Agricultural Research (INIA) and the rice milling industry were taken and discriminated by rice production region, system component, production and food system. It must be noted that the original data is available at different scales. For example, while rice data is available from farm to county level, livestock and pasture information is only available at county level. This means that pasture data exceeds the regional pasture/rice area ratio mentioned in 2.2. However, calculated cattle stocking rates for each region (0.79, 0.75 and 0.81 livestock units ha⁻¹) were similar to the 0.76 livestock units ha⁻¹ reported in previous studies of a typical rice-livestock rotation district (Simeone et al., 2008).

Table 4-1 Components of the rice-livestock system of Uruguay at a regional level. HFSR represents the high fertile and short rotation northern region, MFMR the medium fertile and medium rotation central region and LFLR the low to medium fertile and low rotation eastern region. Values are averages and ranges of the 2004–05 to 2021–22 growing seasons.

Sites	HFSR		MFMR		LFLR		
Rice livestock system parameter	Mean	Range	Mean	Range	Mean	Range	Units
Area of rice harvested annually	0.33	0.25 – 0.40	0.15	0.10 – 0.23	1.15	1.0 – 1.38	ha x 10 ⁵
Area of natural pasture	0.44	0.41 – 0.53	0.30	0.27 – 0.40	2.90	2.78 – 3.22	ha x 10 ⁵
Area of improved pasture	0.22	0.12 – 0.31	0.15	0.12 – 0.29	1.7	1.12 – 2.21	ha x 10 ⁵
Rice: rotation ratio	1:2	–	1:3	–	1:4	–	–
Stock density (bovine + ovine)	0.75	0.61 – 0.80	0.79	0.73 – 0.9	0.81	0.77 – 0.84	LU ha ⁻¹ *
Main soil properties (0-20 cm)**							
pH	6.3	5.8 – 7.6	5.7	5.1 – 6.0	5.8	5.1 – 6.0	1:1 H ₂ O
Cation Exchange Capacity	33.3	13.1 – 43.9	21.5	8.3 – 43.7	11.8	8.1 – 30.7	cmol _c kg ⁻¹
Organic Carbon	28.1	15.5 – 43.3	21.1	15.3 – 34.4	18.3	9.1 – 39.9	g kg ⁻¹
Total soil N	2.5	1.6 – 4.3	2.3	1.0 – 3.7	1.8	0.7 – 4.1	g kg ⁻¹
Sand	200	130 – 270	260	110 – 560	290	150 – 460	g kg ⁻¹
Silt	340	250 – 460	440	200 – 530	390	300 – 480	g kg ⁻¹
Clay	460	220 – 610	300	180 – 450	320	150 – 560	g kg ⁻¹
Bulk density ***	1.25	–	1.31	–	1.35	–	g cm ⁻³

*Livestock unit. 1 LU= 380 kg animal live weight ha⁻¹. **Average data obtained from soil samplings of 52 experiments conducted over 3 years in the main rice production locations at each region. *** Estimated values using the SPAW software (Saxton & Willey, 2006), based on soil type, percentage of sand, clay and organic carbon.

For this reason, we will use those calculated stocking rates for each specific rotation. Complementary data for calculations for each component were selected from the literature (**Table C-1**).

4.3.4 Rice data

Annual information of rice yield and seeded area were collected from governmental agencies (DIEA, 2005; 2022). Crop management data are presented annually by the rice milling companies and summarized by INIA, covering around 85–90% of the total rice area. Because the rice varieties used are largely from the rice breeding program of INIA, crop parameters and characteristics associated with each variety were taken from internal records of INIA. Nearly 75% of the exported or internally consumed rice is white rice (OEC, 2020), so we assumed that all the bran after milling was returned to the rice-livestock system as animal feed. Detailed data of rice coefficients used for calculations are presented in **Table C-1**. Additionally, soil information for the dominant rice systems in each region were collected from a multi-year-location field trial network of N response conducted by INIA (**Table 4-1** Components of the rice-livestock system of Uruguay at a regional level. HFSR represents the high fertile and short rotation northern region, MFMR the medium fertile and medium rotation central region and LFLR the low to medium fertile and low rotation eastern region. Values are averages and ranges of the 2004–05 to 2021–22 growing seasons. **Table 4-1**).

4.3.5 Pasture data

The country forage base is composed of native grassland, semi-natural pastureland and temporary pastureland, averaging 90, 4 and 6%, respectively, following the Allen et al. (2011) classification. Native grassland comprises native grass species and the other two pasture categories include legumes (*Trifolium spp.* and/or *Lotus spp.*) and grasses (*Lolium spp.* or *Festuca spp.*). None of these pastures is fertilized with N. Henceforth, we will refer to the semi-natural pastureland and temporary pastureland as improved pastures. Natural grasslands forage productivity was estimated based on 16 years of remote sensing data for the main ecological regions of the country (Asuaga et al., 2019) and 10 years of remote sensing data for improved pastures (Martínez, 2011). Additional information on dry matter production and botanical pasture

composition at different pasture stages and years was taken from a long-term experiment on rice - improved pasture rotations at INIA facilities. Coefficients of animal forage utilization and local pasture species N content are presented in **Appendix C**.

4.3.6 Livestock data

We estimated animal meat production (beef and sheep) and the N accumulated in the animal body as follows. We used long-term data of county annual livestock stock composition (DICOSE, 2004; SNIG, 2022), and monthly reports of the livestock category and liveweight of animals received at the abattoir from each county (INAC, 2020). The latter also include records of on farm self-consumption on an annual and county basis. These records were used for the animal meat production and N retention calculations. Additionally, wool production was included in the meat production calculations under the equivalent meat concept (FAO, 2018b). Wool was also included into the N retention calculations considering the country average wool production of 4 kg animal⁻¹ yr⁻¹ (DIEA, 2020) adjusted to dry and clean basis, and a literature N concentration value of 16% (ARC, 1980). We calculated the animal N recycling as a function of the animal species, the botanical pasture composition and production, as well as the forage utilisation efficiency (including rice straw) and animal internal N use efficiency (**Table C-1**).

4.3.7 Modelling of missing N data

Despite having good long-term records for calculating the main N pool fluxes, data on soil N losses are scarce and partial in the country. We have recently parameterized and tested the DeNitrification-DeComposition (DNDC) model for different rice rotations (including the rice-pasture-livestock rotation) on a typical rice soil present in the main producer region (east) of Uruguay (Castillo et al., 2023). Results showed good agreement between simulated and observed crops and pasture yields, cumulative N rice uptake and soil NH₄-N during flooded conditions, as well as acceptable estimates of N₂O emissions during aerobic and anaerobic soil conditions. For this study, we used DNDC to simulate N losses (gaseous NH₃ and N₂O, and NO₃⁻ in leaching and runoff) in rice-pasture + livestock rotations in each region over the study period. Considering all the

rotation phases present in one year, we started the modelling in 2004–05 with a first-year rice and second-year rice or pasture, varying the pasture duration as appropriate for each region. Both natural grassland and improved pasture were simulated. The crop parameters were as in our previous study (Castillo et al., 2023) and the soil data according to the region as in **Table 4-1**. Climatic data were obtained from INIAs' weather stations in each region.

4.3.8 Data analysis

We conducted simple NBAL analyses defined as inputs minus outputs following a mass conservation approach and a full chain NUE analysis defined as the quotient of N in food products relative to all new N inputs for both the component and the system level, and for each rotation. For rice, inputs were N in fertilizers, atmospheric N deposition, biological N fixation (BNF) and animal N deposition (AND) occurring during the six months prior to the crop, and outputs of N in grain, gaseous NH_3 and N_2O , and leached NO_3^- . Nitrogen inputs for the livestock component included N from pasture BNF, atmospheric deposition and rice bran, while outputs were N in animal tissue, gaseous NH_3 and N_2O , and leached NO_3^- . The N in AND is that in faeces and urine from the livestock-pasture component of the six months prior to land preparation or chemical fallow. Rice bran is the main feed input used in commercial farms of the rice-livestock system area, so we assumed all the annual production was returned to the livestock component in the same proportion as regionally produced. These N inputs were not considered when analysing the entire system because they act as intermediate product between components. Mineralization of N and N present in the forage were considered as in constant recycling and not included in the calculations.

Complementary to the NUE analysis, we conducted trajectory NUE assessments across the study period based on a graphical approach (EU Nitrogen Expert Panel, 2015). The resulting values of N outputs in edible food products in relation to inputs were plotted against defined low and high NUE thresholds and a desirable N target in food products. For rice, defined NUE thresholds were < 90 and $> 50\%$ and the crop N target was $80 \text{ kg N in grain ha}^{-1} \text{ yr}^{-1}$ (EU Nitrogen Expert Panel, 2015; Task Force on Reactive Nitrogen, 2011). This crop N target value is in accordance with the average rice yield of the period (8.1 Mg ha^{-1} , 130 g kg^{-1} humidity) and with the high yielding rice pasture systems of South America

in general (Singh et al., 2017). While NUE values higher than the upper threshold are associated with soil N mining, NUE values below the lower threshold are associated with low crop N recovery and potentially large N losses to the environment. The latter is covered by the N surplus (NSURP) concept and defined as the difference between all N inputs and outputs in food products, and a maximum admissible value of $80 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ is suggested to avoid environmental risks. For the livestock component, NUE thresholds were < 25 and > 10 based on Gerber et al. (2014), and the defined target N in food products was $3 \text{ kg N ha}^{-1} \text{ yr}^{-1}$. This is reasonable for extensive grazing systems (Oenema et al., 2016) and similar to the values stated by Kanter et al. (2016) as attainable values for Uruguayan extensive conditions. We set the system boundary as the farm gate given the negligible food import and low product industrialization that typify Uruguay as a net commodity exporter.

For all the assessed parameters, rotations were compared using multiple t tests with a significance level of 5%. The Satterthwaite procedure was used if variances were not homogenous. Adjusted regressions were analysed using auxiliary variables to test the equality mean effect of the different groups and to test the homogeneity regression slope. Analyses were conducted using InfoStat (Di Renzo et al., 2017).

4.3.9 Uncertainties and scenarios analysis

We analysed data for the average situation of the rice and livestock components and for the whole system. The pasture component has greatest variability, which in turn influences livestock production (forage offer) and the rice component (N recycling), giving uncertainty to our estimations. For example, a survey of different rice-livestock systems in Uruguay (Simeone et al., 2008) has shown that the percentage of improved pastures considered in those systems ranged from 8 to 84% of the total pasture grazing area. The animal productivity of those scenarios ranged from 54 to 355 kg live weigh $\text{ha}^{-1} \text{ yr}^{-1}$ ($148 \text{ kg ha}^{-1} \text{ yr}^{-1}$ on average). This means that calculations for this study with 34% of improved pastures (**Table 4-1**) could be under or overestimated when different percentages of improved pastures are considered. Another source of uncertainty is the amount of N applied to rice. The Uruguayan rice sector is trying to shift the current national

average yield (see 4.4.1) up to 10 Mg ha⁻¹, probably demanding the addition of 40 kg N ha⁻¹ extra as fertilizer.

To assess the effects of these uncertainties, we analysed three scenarios. First, rice-livestock production rotation on regenerated natural grasslands after rice crop (SGR). Second, the same scenario but with 40 kg N ha⁻¹ fertilizer to rice (SGRN). Third, with 80% improved pastures (SIP). For SGR we considered a decrease of N output in animal products by 25% based on the stocking rate of extensive livestock systems (Soares de Lima, 2009). That also decreases the N transferred from the livestock component to the rice. For the SIP scenario we assumed a high meat production of 355 kg live weigh ha⁻¹ yr⁻¹ (Simeone et al., 2008) and an extraction rate (ratio of sold animal weight to total animal weight in stock) of 40% (Soares de Lima, 2009). Further on, we calculated rice yield and N losses with the DNDC model. Increased N in rice bran fed to livestock after higher rice yields were allowed for.

4.4 Results

4.4.1 Rice yield and nitrogen balances

Rice yield reached 8100 ± 727 kg ha⁻¹ yr⁻¹ with yield gain rates from 66 kg ha⁻¹ yr⁻¹ (HFSR) to 110 kg ha⁻¹ yr⁻¹ (MFMR and LFLR) over the period (**Figure 4-1a**). The N fertilization rate also showed increasing trends of 3.6, 2.7 and 2.3 kg N ha⁻¹ yr⁻¹ in HFSR, MFMR and LFLR, respectively (**Figure 4-1b**). The annual increases in N rate were 4.7 (HFSR), 1.2 (MFMR) and 0.9 (LFLR) times the annual increase of N removed in grain yield.

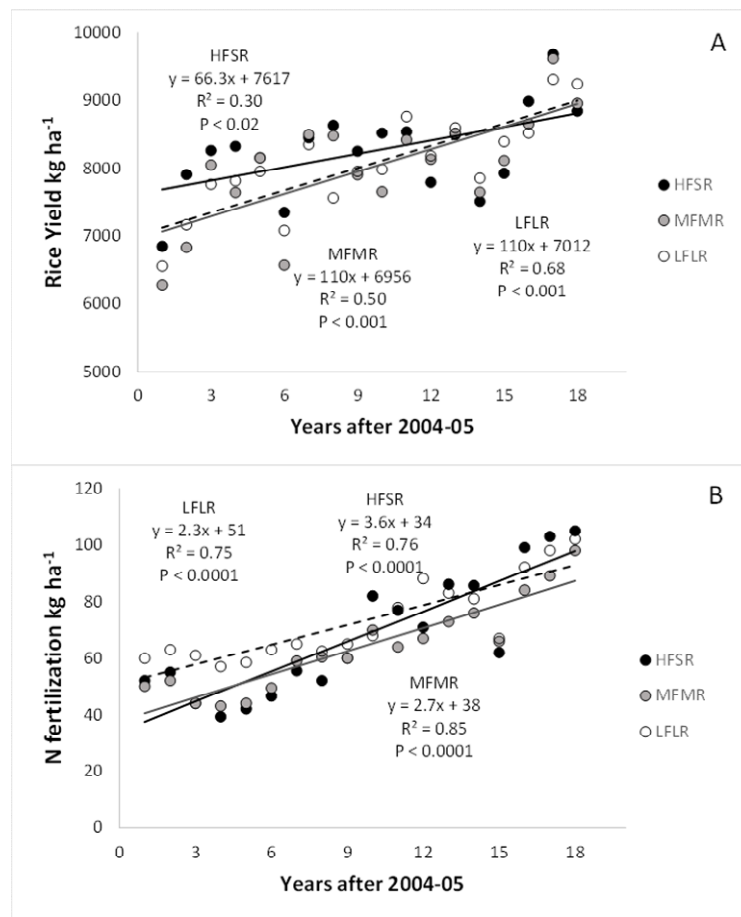


Figure 4-1 Changes from 2004/5 to 2019/20 in NUE (= outputs / inputs $\times$ 100) and N surplus (= inputs – outputs) in (A.) the whole system and (B) the rice and livestock components; (C) N inputs, outputs and balance in the whole system.

Total N input to rice was greater in LFLR than HFSR and MFMR (**Table 4-2**). The main N input to rice was in fertilizers (73, 70 and 68% of the total N inputs for HFSR, MFMR and LFLR, respectively) with smaller contributions from BNF and atmospheric deposition. Differences in total N inputs among rotations were due to AND transferred to rice in LFLR (26.1 kg N ha⁻¹ yr⁻¹) which was 40 and 57% greater than for MFMR and HFSR, respectively. For the pasture-livestock component, total N inputs were greater in HFSR followed by MFMR and LFLR, due to the BNF from pastures. This BNF value is linked to the entire pasture area of each region (native grasslands + improved pastures); but when referring to the specific improved pasture area, on average 46 kg ha⁻¹ yr⁻¹ of N were fixed. Similarly, differences in N input from bran are mainly explained by the total area of rice in each region, e.g., N of rice bran in HFSR was twice the amount in LFLR because the rice rotation ratio is half. On average, atmospheric N deposition was

very similar among rotations averaging 6 kg N ha⁻¹ yr⁻¹. At the rice-livestock system level, total N inputs for each rotation differed in the order HFSR > MFMR > LFLR.

Total N outputs for the rice component were greater in HFSR than MFMR and LFLR. Nitrogen in grain was the main output and reached very similar values among rotations, averaging 84.6 kg N ha⁻¹ yr⁻¹, i.e., around 72% of the total N output. Differences in total N output were associated with N losses, which represented 31, 28 and 24% of total N output in HFSR, MFMR and LFLR, respectively. Volatilization was the main N loss (97, 91 and 87%), followed by denitrification (2, 7 and 8%) and leaching plus runoff (1, 2 and 5%) for HFSR, MFMR and LFLR, respectively.

Average total N output of the livestock-pasture component was 12% of that in the rice component. Nitrogen losses were the main output averaging 7.2 kg N ha⁻¹ yr⁻¹, followed by the N transferred from the livestock to rice (5.1 kg N ha⁻¹ yr⁻¹), and both outputs varied little among rotations. Output animal tissue was only 1.8 kg N ha⁻¹ yr⁻¹. At the system level, total N input and output values were close to each other, resulting in slightly positive balances. However, N system balances differed between rotations over the study period. The system NBAL for HFSR increased from -8.5 in 2004/05–2009/10 to +2.8 in 2010/11–2015/16 and +14.1 kg N ha⁻¹ yr⁻¹ in 2016/17–2021/22. By contrast, system N balances for MFMR and LFLR were always positive but decreased over time from 5.7 to 3.8 to 4.2 kg N ha⁻¹ yr⁻¹ in MFMR and from 9.7 to 8.7 to 7.1 kg N ha⁻¹ yr⁻¹ in LFLR over the same periods. For all rotations, system NBAL was highly correlated with total N inputs of the livestock-pasture component ($r = 0.87, 0.82$ and $0.75, p < 0.001$ for HFSR, MFMR and LFLR, respectively), mainly due to the amount of N fixed during the pasture phase ($r = 0.73, 0.70$ and $0.66, p < 0.01$ for the same rotations). In addition, the system NBAL was strongly associated with N fertilizer inputs in HFSR ($r = 0.86, p < 0.0001$). Further correlations among all the N factors could be found in **Table 4-3**.

Table 4-2 Nitrogen balance of each system component and the entire rice-pasture-livestock system. Values are averaged over 2004/5 to 2021/22 growing seasons. HFSR = high fertile and short rotation (northern region), MFMR = medium fertile and medium

length rotation (central region), and LFLR = low to middle fertile soil and long rotation (eastern region).

Component		Rice-pasture rotation (rice: rotation ratio)					
level	Balance factor	HFSR (1:2)		MFMR (1:3)		LFLR (1:4)	
Rice	<i>Inputs (kg ha⁻¹)</i>						
	Fertilizers	67.6	a	63.8	a	72.9	a
	Animal direct deposition	16.6	b	18.7	b	26.1	a
	Atmospheric deposition	6	a	6.4	a	5.8	a
	BNF free living + symbiotics	2.5	a	2.5	a	2.5	a
	Total N Inputs	92.8	b	91.5	b	107.2	a
	<i>Outputs (kg ha⁻¹)</i>						
	Grain	86.1	a	83.6	a	84.1	a
	Total N losses*	39.4	a	32.6	ab	26.3	b
	Total N outputs	125.5	a	116.2	b	110.4	b
<i>N Balance</i>		-32.7	b	-24.7	b	-3.2	a
Livestock	<i>Inputs (kg ha⁻¹)</i>						
	Pasture BNF	20.2	a	17.7	ab	16.0	b
	Rice bran	10.2	a	6.5	b	4.9	c
	Atmospheric deposition	6	a	6.4	a	5.8	a
	Total N Inputs	36.3	a	30.7	b	26.7	c
	<i>Outputs (kg ha⁻¹)</i>						
	N in animal tissue	1.7	a	1.8	a	1.8	a
	Total N losses*	6.2	b	8.6	a	6.9	b
	Animal direct deposition	5.5	a	4.7	a	5.2	a
	Total N outputs	13.3	a	15.1	a	14.0	a
<i>N Balance</i>		23.1	a	15.6	b	12.7	c
System	Total N Inputs (kg ha ⁻¹)**	49.5	a	41.2	b	37.6	c
	Total N outputs (kg ha ⁻¹)**	46.7	a	36.8	b	29.1	c

<i>N Balance (kg ha⁻¹)</i>	2.8	c	4.4	b	8.5	a
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Means followed by the same letter within rows are not statistical different (p=0.05)

*Total N losses considered NH₃, N₂O, N leached and runoff

**Total N inputs and outputs at a system level did not include the animal direct deposition factor. Presented N inputs and outputs values were adjusted by the proportion of each component (rice and livestock) on an annual base.

4.4.2 Full chain-NUE and N surplus analysis

4.4.2.1 Rice Component

The full-chain NUE for the rice component, defined as the quotient between N in grain relative to the annual total N inputs, was higher in HFSR and MFMR (98 and 94%, respectively) than LFLR (79%) averaged over the 18 years. The NUE trajectory had two stages in HFSR: first where NUE values exceeded the upper threshold (average 115%), and then when NUE was in the target zone (**Figure 4-2 a**). This shift happened because of an increase in the N fertilization rate (50 vs. 86 kg N ha⁻¹ yr⁻¹; **Figure 4-1**) In the second phase, during 2018 only, the NUE exceeded the threshold due to less N fertilizer application. On average, the total N removed in grain was higher than the desirable N target (80 kg N ha⁻¹ yr⁻¹), with just a few records (16%) below that value. For MFMR, 50% of the records were above or around the upper threshold, and the remaining data was in the target zone. Again, the increase in N fertilizer dose explained a constant offset of NUE into the desirable zone ($r = -0.73$, $p < 0.001$; **Figure 4-2 a**).

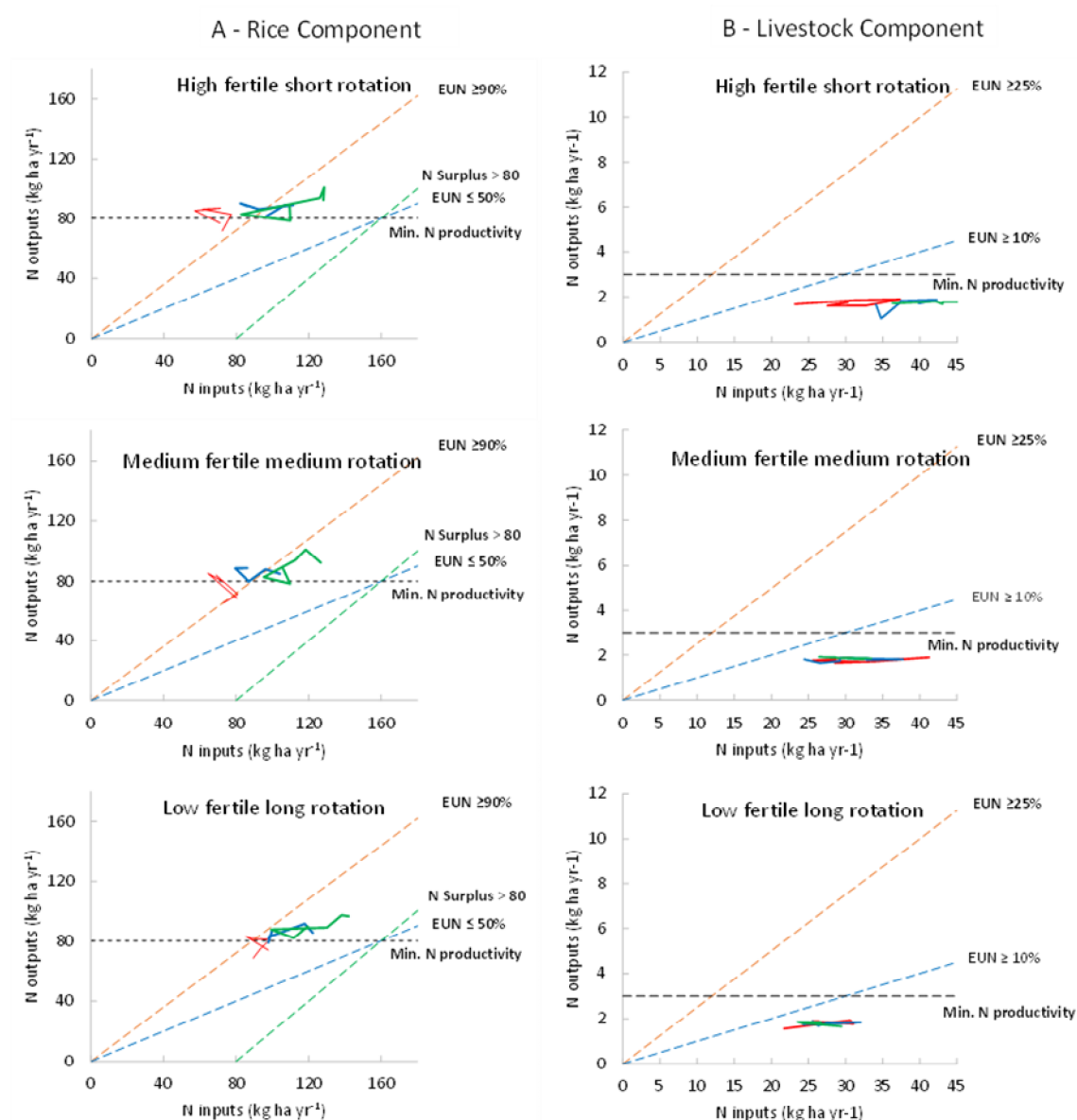


Figure 4-2 Changes in N outputs in food products versus N inputs from the 2004–05 to the 2021–22 growing seasons for the three rotations/regions: (A) rice component, (B) livestock component. Solid red, blue and green lines indicate changes from 2004–05 to 2009–10, 2010–11 to 2015–16 and 2016–17 to 2021–22, respectively. Dashed orange and blue lines indicate NUE (= outputs / inputs $\times$ 100) of 90 and 50%, respectively for rice and 25 and 10%, respectively for livestock. Dashed black lines indicate the expected N output for a desirable level of production and dashed green indicate the maximum admissible N surplus.

Unlike the other rotations, almost all the data (90%) for LFLR attained NUE values in the target zone across the study period, reaching an average amount of N in grain of 84 kg N ha⁻¹ yr⁻¹. Here again, the increase in the N fertilizer dose strongly influenced NUE each year ($r = -0.80$, $p < 0.0001$) shifting values toward the lower

NUE threshold (50%) in the last years of the study. On average, NSURP of LFLR was higher ($23 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) than in MFMR and HFSR (8 and $7 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, respectively), across the study period. However, positive values for NSURP in MFMR and HFSR were observed around the middle of the time series when NUE fell below 100% (**Figure 4-3 a**). At the end of the study period NSURP reached 36, 33 and 46 kg N ha^{-1} for HFSR, MFMR and LFLR, respectively. As expected, NSURP in rice was positively correlated with N fertilizer addition in all regions ($r > 0.90$, $p < 0.001$). A negative correlation between NUE and NSURP was found for all rotations ($r = -0.97$, $p < 0.0001$). Complementary correlations of the selected N factors for the rice component are shown in **Table 4-3**. The decline in NUE across the period differed ($p = 0.005$) between HFSR and LFLR, while MFMR was intermediate. Similarly, the NSURP rate increase differed between HFSR and LFLR, with MFMR intermediate (**Figure 4-3 a**). The different downward trends of NUE and associated upward trend of NSURP matched the different stages of the generalized pathways shown in **Figure 4-3 c**.

4.4.2.2 Livestock Component

Livestock NUE values were much lower than those for rice. For the 18-year period, NUE values were 6.8, 6.0 and 4.8 % for LFLR, MFMR and HFSR, respectively. As mentioned in Section 4.4.1, N output in animal tissue was almost the same for the different rotations, so differences in NUE were associated inversely with the total N inputs, mainly by pasture BNF ($r = -0.80$, $p < 0.0001$) followed by rice N bran ($r = -0.72$, $p < 0.0001$). The remaining N factors correlations for the livestock component are shown in **Table 4-3**. While NUE values for LFLR and MFMR remained flat over time, values for HFSR decreased by around 20% (**Figure 4-2 Figure 4-2b**), explained by an increase in pasture BNF linked to a greater area of improved pastures over time. Both, NUE and N in animal products were below the targets (10% NUE and 3 kg N ha^{-1} animal products, respectively) in all rotations. Records were closer to the lower NUE threshold during the first years in LFLR but more distant in the last years of HFSR. Unlike rice, NUE of the livestock component was not associated with NSURP and both variables remained steady over the study period.

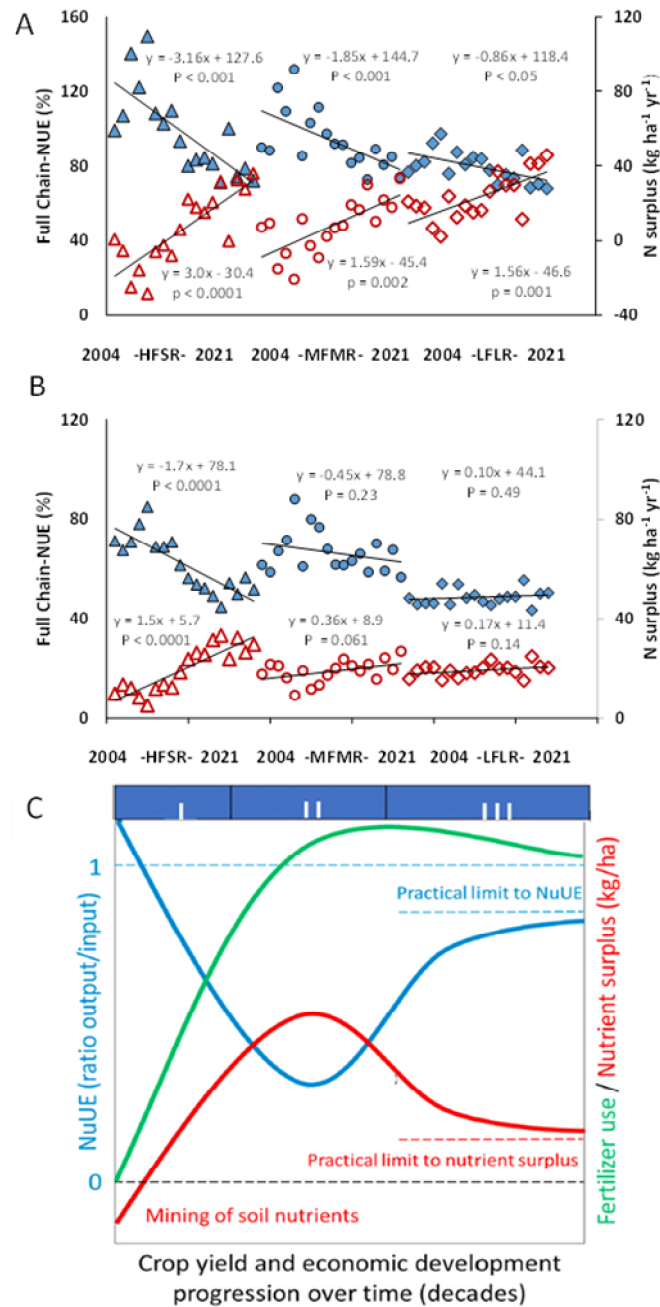


Figure 4-3 (A) Rice and (B) system N surplus and N use efficiency from 2004–05 to 2021–22 growing seasons. Triangles = HFSR (high fertile and short rotation), circles = MFMR (medium fertile and medium length rotation), rhombuses = LFLR (low to medium fertile and long rotation regions). Blue symbols = Fch-NUE and red symbols = N surplus. (C) Generalized development pathway for N use efficiency and N surplus (after Dobermann et al., 2022).

Table 4-3 Pearson correlation matrix for selected nitrogen (N) factors and N balance (NBAL), use efficiency (NUE) and surplus (NSURP) in each rotation. The numbers in the table body represent the correlation coefficient (r), while asterisks denote statistical significance: correspond to **** (<0.0001), *** (<0.001), ** (<0.01) and * (<0.05). Coefficient values without asterisks are not statistically significant.

Rotation	N factor	N factor												
		(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)	(13)
HFSR	Fertilizer (1)	1.00****	-0.36	0.58	0.54	0.86***	-0.86***	0.95***	0.66**	-0.36	0.58*	0.86***	0.7***	0.91***
	Rice N losses (2)		1.00****	-0.16	-0.07	-0.73**	0.45	-0.46***	-0.33	-0.37	-0.38	-0.65***	0.35	-0.38
	Pasture BNF (3)			1.00****	0.30	0.50*	-0.54*	0.57*	0.93****	-0.76***	0.97****	0.74***	-0.82****	0.82****
	Rice Bran (4)				1.00****	0.29	-0.23	0.34	0.20	-0.50*	0.39	0.27	-0.21	0.35
	Rice NBAL (5)					1.00****	-0.92****	0.94****	0.57*	-0.38	0.53*	0.94****	-0.82****	0.85****
	Rice NUE (6)						1.00****	-0.97****	-0.54*	0.34	-0.55*	-0.87****	0.90****	-0.88****
	Rice NSURP (7)							1.00****	0.58*	-0.34	0.57*	0.90****	-0.90****	0.92****
	Livestock NBAL (8)								1.00****	-0.69***	0.90****	0.82****	-0.81****	0.82****
	Livestock NUE (9)									1.00****	-0.81****	-0.54*	0.50*	-0.54*
	Livestock NSURP (10)										1.00****	0.73***	-0.81***	0.82****
	System NBAL (11)											1.00****	-0.91****	0.94****
	System NUE (12)												1.00****	-0.98****
	System NSURP (13)													1.00****

Table 4-4 (Continued)

Rotation	N factor	N factor												
		(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)	(13)
MRMF	Fertilizer (1)	1.00***	0.52*	-0.24	0.63**	0.56*	-0.73***	0.80***	-0.18	0.13	-0.05	0.17	-0.44	0.57*
	Rice N losses (2)		1.00***	-0.03	-0.005	-0.11	-0.65***	0.71***	-0.10	0.07	0.04	-0.18	-0.55*	0.50*
	Pasture BNF (3)			1.00***	-0.17	-0.15	0.15	-0.13	0.90***	-0.87***	0.92***	0.70**	-0.39	0.51*
	Rice Bran (4)				1.00***	0.19	-0.03	0.13	-0.004	0.13	-0.01	0.10	0.24	0.09
	Rice NBAL (5)					1.00***	-0.65**	0.62**	-0.14	-0.10	0.04	0.47*	-0.48*	0.47*
	Rice NUE (6)						1.00***	-0.98***	0.23	0.02	-0.03	-0.17	0.81***	-0.70**
	Rice NSURP (7)							1.00***	-0.18	-0.02	0.06	0.19	-0.77***	0.73***
	Livestock NBAL (8)								1.00***	-0.88***	0.91***	0.80***	-0.30	0.49*
	Livestock NUE (9)									1.00***	-0.92***	-0.84***	0.50*	-0.62**
	Livestock NSURP (10)										1.00***	0.82***	-0.52*	0.69**
	System NBAL (11)											1.00***	-0.55*	0.71***
	System NUE (12)												1.00***	-0.92***
	System NSURP (13)													1.00***

Table 4-5 (Continued)

Rotation	N factor	N factor												
		(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)	(11)	(12)	(13)
LRLF	Fertilizer (1)	1.00****	0.52*	-0.19	0.75**	0.53*	-0.80****	0.92****	-0.38	0.11	-0.14	-0.01	-0.08	0.54*
	Rice N losses (2)		1.00****	-0.35	0.21	-0.37	-0.44	0.45	-0.41	0.37	-0.40	-0.57*	0.06	0.06
	Pasture BNF (3)			1.00****	0.15	0.76***	0.27	-0.22	0.76***	-0.75***	0.92****	0.66**	-0.47*	0.55*
	Rice Bran (4)				1.00****	0.33	-0.27	0.49*	-0.12	-0.03	0.19	0.08	0.24	0.42
	Rice NBAL (5)					1.00****	-0.63**	0.66**	-0.04	-0.26	0.17	0.56*	-0.39	0.57*
	Rice NUE (6)						1.00****	-0.96****	0.39	-0.05	0.20	-0.06	0.44	-0.56*
	Rice NSURP (7)							1.00****	-0.37	0.05	-0.16	0.08	-0.33	0.60**
	Livestock NBAL (8)								1.00****	-0.79***	0.81***	0.80***	-0.48*	0.40
	Livestock NUE (9)									1.00****	-0.88****	-0.80***	0.70**	-0.69**
	Livestock NSURP (10)										1.00****	0.76***	-0.57*	0.68**
	System NBAL (11)											1.00****	-0.63**	0.66**
	System NUE (12)												1.00****	-0.78**
	System NSURP (13)													1.00****

4.4.2.3 Rice-livestock system

The average annual NUE at system level was higher in HFSR and MFMR (62 and 67%, respectively) than LFLR (49%), following the same trend as for the rice component (**Figure 4-3 a** and **Figure 4-3 b**). The annual rate decrease was higher in HFSR (-1.7%, $p < 0.0001$) than MFMR (-0.45 %) and LFLR (0.1%), the latter being basically flat during the study period. The NUE was positively correlated with rice NUE at HFSR ($r = 0.91$, $p < 0.0001$) and MFMR ($r = 0.80$, $p < 0.0001$), and with livestock NUE in LFLR ($r = 0.70$, $p < 0.001$). For HFSR there was also a negative correlation with N fertilizer addition ($r = -0.84$, $p < 0.0001$). For NSURP, the annual increase was higher in HFSR (+ 1.5 kg N ha⁻¹ yr⁻¹) than MFMR (+ 0.36 kg N ha⁻¹ yr⁻¹) and LFLR (+ 0.17 kg N ha⁻¹ yr⁻¹), which were not significantly different. For all rotations, NSURP was positively correlated with rice N fertilizer addition ($r = 0.55$, $p = 0.018$; $r = 0.57$, $p = 0.013$ and $r = 0.91$, $p < 0.0001$ for LFLR, MFMR and HFSR, respectively). As for the rice component, NUE was negatively correlated with NSURP ($r = -0.78$, -0.92 and -0.98 , $p < 0.0001$ for LFLR, MFMR and HFSR, respectively). A complete matrix correlation for all the N factors of the rice-livestock system is shown in **Table 4-3**.

4.4.2.4 Scenario analysis

For all rotations, the SGR scenario generated the most negative NBAL and a higher NUE than the original situation in the rice component (**Figure 4-4**). That was due to a greater reduction in N inputs (less transference of AND because the absence of improved pastures) than the decrease in N outputs, mainly N in grain and N losses. In the HFSR and MFMR rotations, NUE was shifted beyond the upper NUE threshold (103 and 98%, respectively) but not in the LFLR rotation (85%). Adding more N fertilizer in the SGRN scenario not only increased N inputs as expected but also increased N outputs, mainly due to greater N losses which increased by 52, 60 and 80% over the original values for HFSR, MFMR and LFLR, respectively. This resulted in an even more negative NBAL and all three rotations reached NUE values between 80 and 87%. Total N inputs of the SIP scenario were almost the same as for SGRN but a more substantial N input was obtained from pasture BNF. However, N losses were lower than in SGRN because less N was added as fertilizer. The NBAL for the SIP scenario was the least negative among the three scenarios, increasing on average by 10 kg N ha⁻¹

yr^{-1} for HFSR and LFLR and not changing in MFMR. The resulting NUE was 84, 79 and 70% for HFSR, MFMR and LFLR, respectively. The NSURP was higher in SIP (23, 27 and 43 kg N $\text{ha}^{-1} \text{yr}^{-1}$ for HFSR, MFMR and LFLR, respectively) than in the other scenarios and rotations (all < 26 kg N $\text{ha}^{-1} \text{yr}^{-1}$).

For the livestock component, the fall in N inputs of the SGR scenario (-40% on average) was explained mostly by the absence of N inputs from pasture BNF. Nitrogen outputs also decreased mainly because of the reduced AND transference to rice (-50% on average), followed by a fall in animal N products (-25%). Because the fall in N inputs was higher than the decrease in N outputs, NBAL and NSURP decreased in all rotations and NUE increased, reaching values above the lower NUE livestock threshold (10%) for MFMR and LFLR, and close to it for HFSR (8.5%). Increased N fertilization in the SGRN scenario only affected the input from rice bran which increased the NBAL compared with SGR but was still smaller than in the original situation. As expected, there were greater changes in the SIP scenario due to a substantial increase of N inputs from pasture BNF. However, N outputs from animal N products and N losses also increased, resulting in increases in NBAL and NUE (41, 38 and 43% for NBAL and 14, 11 and 15% for NUE in HFSR, MFMR and LFLR, respectively).

At a system level and for all rotations, NBAL was negative in SGR and SGRN and positive in SIP. By contrast, NUE was higher in SGR and SGRN and lower in SIP, in the latter being even below the original situation. The NUE values in the SGR scenario were 32, 30 and 44% higher than the original situation while it decreased by 15, 10 and 14% at SIP for HFSR, MFMR and LFLR, respectively.

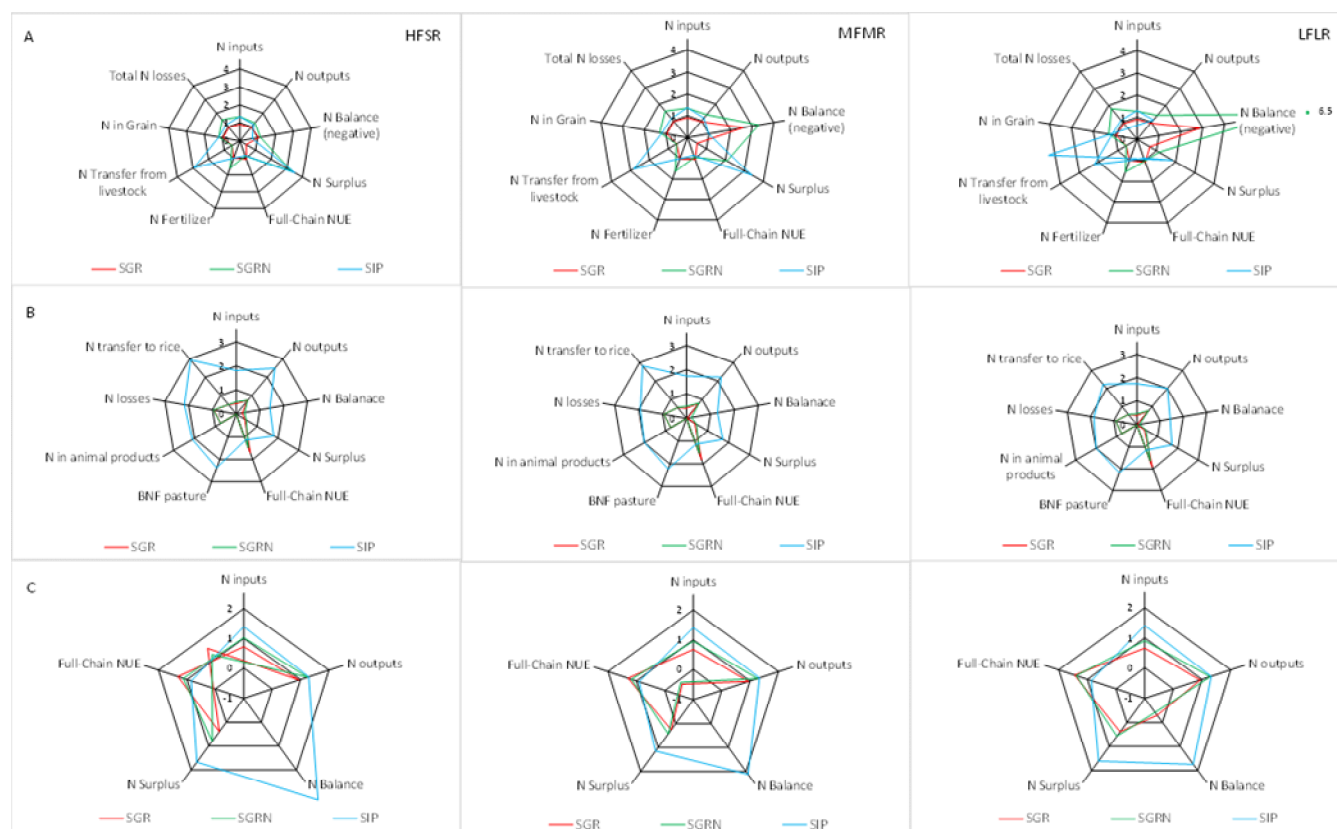


Figure 4-4 Relative changes of N balances, inputs and outputs, N surplus and N use efficiency for three simulated scenarios. Red line represents a rice rotation with grassland (SGR), green line a rice rotation with grassland but adding extra N to the rice crop (SGRN) and the blue line a rice rotation with improved pasture on 80% of the total forage area (SIP). (A) Rice component, (B) livestock component and (C) entire system. HFSR = high fertile and short rotation, MFMR = medium fertile and medium length rotation and LFLR = low to middle fertile soil and long rotation.

4.5 Discussion

4.5.1 Nitrogen balance

We have found regional differences in the N balance of the rice and livestock component as well as the whole system. The negative N balances in the rice in the more-fertile HFSR and MFMR regions differed from the slightly positive balances we found at the country level ($3.5 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) in our earlier study (Castillo et al., 2021). But country level estimates relied on literature data and some N outputs, particularly N volatilization, might have been underestimated. The HFSR and MFMR regions have greater N losses because of the higher natural soil fertility and a proportionally greater N transfer from the livestock component to rice. The N balance is highly sensitive to inputs and losses, as shown by the far more negative values in HFSR and MFMR rotations when N fertilizer use was much lower during the first five years (-57 and $-30 \text{ kg N ha}^{-1} \text{ yr}^{-1}$, respectively). Therefore, the greater precision of this regional analysis is important for correctly understanding the system.

It is likely that the accumulated NBAL before our study period was highly negative because of much lower or no N fertilizer use after the introduction of the rice component. Linking the negative NBAL with the inferred initial N stock based on the soil parameter data, we estimate an average depletion of 10, 8 and 1% of the total N (0.20 m depth) for HFLR, MFMR and LFLR, respectively. Such mining of soil N is typical of the agriculture of developing countries at the early stages of intensification, but this can be partially reversed with increased N fertilizer doses over time, in turn leading to increased losses and environmental hazards in the long term (Quemada et al., 2020).

How it is possible that the Uruguayan rice system has been in operation for more than 50 years with consistently high yield levels if the annual NBAL has been negative? The answer is linked to the livestock component. The contributions of N fixed by pastures and N returned in rice bran exceed the relatively low N outputs from the system. The main output was the N lost ($7.2 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) of which 75% was by volatilization, which is similar to previous reports for Uruguay (FAO, 2018b; Perdomo et al., 2009). As a result, all rotations reached positive NBALs for the livestock component which, when integrated, resulted also in positive

NBALs for the whole system (**Table 4-2**). Therefore, the livestock component plays a key role in supporting the rice component by offsetting its negative NBAL. Such complementarity between components has been reported in other systems. For example, García-Préchac et al. (2004), showed that during 46 years of upland crop-pasture rotation in a long term experiment, soil organic C was depleted during the upland crop phase, but recovered in the pasture phase. In each crop-pasture cycle, soil C rose to near the initial C level. Similar results were reported by Macedo et al. (2021), indicating that the presence of pastures for animal grazing in rice rotations is the key to maintain soil organic C and total N in the rotation. Additionally, Carlos et al. (2020) mentioned that integrated rice-livestock systems also contribute to increase the biochemical soil processes improving the cycling and availability of several nutrients in addition to N.

In the following sections we discuss how the different N balances in each component and the entire system are related to their N use efficiencies and N surpluses, and how the simulated scenarios can inform future improvements of the system.

4.5.2 Full-chain N use efficiency and N surplus

In general, the less positive the NBAL the higher was the NUE, reaching values greater than the upper threshold (90%), indicating soil N mining. For HFSR (98%) and MFMR (94%), this is mainly explained by low N fertilizer use during the early years. Some studies in European countries (Erisman et al., 2018; EU Nitrogen Expert Panel, 2015) have shown a trajectory opposite to this, with the NUE moving from very low values towards the desirable target after reducing N inputs and improving N recovery by the crops. In our study, the shift to the target NUE zone of HFSR and MFMR regions was associated with higher N fertilizer rates. In the LFLR region, which had a slightly negative average NBAL, NUE was in the desirable zone for the whole period. The high yield reached by rice in all years and rotations ensured the minimum N target in grain ($80 \text{ kg N ha}^{-1} \text{ yr}^{-1}$), indicating a high contribution of indigenous soil N, at least for the HFSR and MFMR rotations. However, the trend of increasing N fertilizer rates across the three regions resulted in NUE in rice of 75, 80 and 70% for HFSR, MFMR and LFLR, respectively, over the last three years of the series, which is very close to the

desirable 70% NUE value for crop systems (Scientific Panel on Responsible Plant Nutrition, 2020).

By contrast to the rice, the positive NBAL in the livestock component corresponded to a very low and steady NUE (6% on average), much below the defined thresholds ($25\% > \text{NUE} > 10\%$; Gerber et al., 2014), but similar to reports from extensive livestock systems, which ranged from 4 to 7% (Gameiro et al., 2019). In addition, the N amount captured in animal food products ($1.8 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) was also low compared with the target ($3 \text{ kg N ha}^{-1} \text{ yr}^{-1}$). In our previous study (Castillo et al., 2021), the livestock NUE was within the thresholds because the pasture area data exceeded the typical rice to pasture ratio. That meant firstly that the percentage of improved pasture area was lower than in this study, and with it, the amount of N fixed by improved legume pastures; and secondly, the amount of rice bran per hectare was lower because of the greater total pasture area considered. Therefore, lower amounts of both N inputs explained the higher livestock NUE of the previous study. But even though a “too low” NUE is associated with inefficient resource use and could be linked to high N losses to the environment, our study shows how a low-efficiency component (livestock) helps the other system component (rice) reach a very high NUE record. When combined, the entire system reached a high average NUE (62, 67 and 49% for HFSR, MFMR and LFLR, respectively, for the entire period). These values are higher than reported for other mixed systems, which were around 35% (Godinot et al., 2015) and 45% (Westhoek et al., 2014). However, in the last third of the time span analysed here, system NUE values decreased considerably (51, 63 and 48%) due to greater N fertilizer use.

As shown for the rice component, the observed increase in N fertilizer use positively affects the NSURP and decreases the NUE of the component and the complete system in all the rotations. Given that there is no evidence of a stabilization in the applied N dose, it is possible that NSURP will continue to increase and NUE decrease. Given the trajectory of nutrient use efficiency in the theoretical scheme in **Figure 4-3 c** (after Dobermann et al., 2022), a similar development could occur in the Uruguayan rice-livestock system. It seems that the three rotations were at different parts of Stages I and II in **Figure 4-3 c** based on the slope of the adjusted regression for NUE and NSURP. While HFLR seems

to be in the left upper and bottom zone of Stage I for NUE and NSURP, respectively, MFMR and LFLR are likely to be in the first and approaching the middle zone of Stage II. Accordingly, the increase in the N fertilizer rate for those regions was $HFLR > MFMR > LFLR$ while NSURP in the last years of the series was greater in $LFLR > MFMR = HFLR$. The system-level analysis followed the same trends for the rice component. Because the three rotations were apparently in different stages and zones within the NUE development pathway scheme, management changes should consider the initial situation to shift the current scenario to Stage III, trying to avoid Stage II as much as possible. Such an analysis could help to identify the best management practices to be adopted in each region.

4.5.3 Scenario analysis

In some areas, improved pastures provide biologically fixed N to the system, compensating for N exported in grains which exceeds that added as fertilizer (Pittelkow et al., 2016; Tseng et al., 2021). However, the area where rice rotates is often dominated by native grassland with no or very low inclusion of improved pasture species, which increases the importance of N contributed to the rice crop from livestock depositions. Removal of the improved pasture in the SGR scenario caused a greater decrease of N inputs (- 9%) than N outputs (-5%) in the rice component, mainly due to less N transferred to rice as animal direct deposition, especially in HFLR (-50%) because of a shorter pasture phase. As a result, NBAL was even more negative and NUE more positive, especially in LFLR. However, the results of this simulation might be incomplete because a continued negative NBAL over time would reduce crop yields in the future, as mentioned by Stoorvogel et al. (1993). In that case, less N removed in grain will decrease NUE and the NBAL will be less negative.

When the NUE indicates N mining, a strategy of N replenishment is recommended (Quemada et al., 2020). Our simulations with increased N application rates (SGRN scenario) showed that after the N fertilizer was increased by 60%, the NUE was improved, shifting the efficiency values from mining into the desirable zone, which also increased the rice yield. However, NBAL and NSURP reached the minimum and maximum values of the scenarios,

respectively, associated with a significant increase of 63% in N losses, indicating that a strategy of N replenishment through N fertilizer addition is not a good alternative. Finally, the SIP scenario maintained a similar amount of N input to SGRN but replaced the fertilizer N with BNF. This allowed a higher rice yield than in the original situation (17% on average), low increase of N losses (15% on average) compared with the SGRN scenario and reduced the negative NBALs for all rotations. The only negative trend was the increase of NSURP, which was similar to SGRN, but N losses did not increase as observed for that scenario.

For the livestock component, the scenario analysis showed a similar trend as for rice but included major changes in SGR and SIP. For SGR, the removal of improved pastures decreased N inputs (40% on average), and with that reduction all other parameters related to the NBAL decreased. However, the greatest change was the increase of NUE (+ 72, + 82 and + 91% for HFLR, MFMR and LFLR, respectively) which shifted NUE values into or around the targeted efficiency zone ($25\% < \text{NUE animal systems} > 10\%$). This means that if pastures are improved through the inclusion of legumes, an increase in animal productivity brings the NUE within the desirable zone. That was precisely what happened in the SIP scenario, where a greater percentage of improved pastures (80%) increased meat productivity by 100%. But because of the higher N inputs from biological fixation, NUE values were just 14% higher on average, reaching values below the lower threshold (10%). In those cases, alternative management towards increasing the animal productivity must be applied but avoiding risks associated with very high stocking rates (Lezama & Paruelo, 2022). However, there is still an opportunity to improve animal productivity because the stocking rate of the SIP scenario (1.4 livestock units of 380 kg live weight ha^{-1}) is still far below the standard of improved pastures (Rovira et al., 2020).

In summary, the scenario exploring a stable and greater use of improved legume pastures seems to give benefits of other components in terms of productivity and N budget for each component and the system. This is close to the proposal of Kanter et al. (2016) and Soares de Lima (2009) who identified improved practices to increase the livestock productivity, and indirectly the crop component. But we also believe that there is still room for improved integrated management of the rice-livestock system which must be defined to enhance the main aspects of the

circular economy approach, leading the system into an intensified and sustainable scenario.

4.6 Conclusions

The Uruguayan rice-livestock system is highly efficient and productive, with relatively low N fertilizer inputs and low N surpluses across the regions and rotations. In all the regions and rotations, this system is sustainable in terms of N balance because of the complementarity of the livestock and rice components. This could be challenged if either or both components were to intensify without considering the whole system. For this, a good quantification of the trajectory of all the N parameters and the use of modelling tools can help to design future strategies. Improvements in livestock productivity and efficiency could be achieved by adjusting pasture lengths in regions with shorter pasture rotations and increasing the proportion of legume pastures. This could also contribute to greater rice yields without more N fertilizer use. Fine tuning the system could also help to reduce greenhouse gas emissions and other costs associated with fertilizers use.

4.7 Authorship

Conceptualization, visualization, data acquisition, curation, interpretation, analysis, modelling and drafting of the manuscript: JC. Writing – review & editing: JC, SMH, GJDK, MJR. Supervision SMH and GJDK.

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5 DISCUSSION

Uruguay is a primary food producer country that exports around 80 % of its meat production and 90% of its rice. Around one million hectare of the country is used for a permanent rice-livestock rotation, which has been conducted for over five decades. Despite the economic importance of both food products in the national economy, few studies have assessed in detail the productivity and efficiency of the individual components at regional or national scales. Past studies have focused on one or other of the components without considering interactions in the system as a whole. To the best of our knowledge, this is the first time the rice-livestock system of Uruguay has been assessed at regional and national scales in terms of nitrogen (N) use efficiency and its components. In addressing objective 1 it was possible to reach a good simulation for the crop response and the main N losses of different rice rotations, critical in N balance calculations. Additionally, the fulfilment of objectives 2, 3 and 4 allowed to investigate N trajectories not only for N balances but also for NUE and N surplus.

5.1 Consistency of the results across scales

Comparison among systems requires uniform assessment procedures – e.g. calculation methods and system boundaries – and parameters – e.g. for N inputs: biological nitrogen fixation (BNF), N in feed or atmospheric N deposition are decisive in the calculated NUE values (Erisman et al., 2018; Ladha et al., 2020). In this study, the rice-livestock-pasture system of Uruguay was assessed at three different scales and time periods.

The calculated NUE values differed for a) the country level with a broad rice-to-pasture ratio and over 16 years, b) for the long-term experiment and over 10 years period, and c) for the regional level within the country and 18 years studies (NUE of 23, 53 and 59%, respectively), they were as expected. The lowest NUE value at the country-level assessment was strongly influenced by the defined rice:pasture ratio (1:50) for that study, which clearly exceeds the typical value (1:4). That meant that the system NUE was closer to the livestock than the rice NUE value (13 vs. 66%). However, when the rice: pasture ratio was similar among

studies, NUE values also were (53 vs. 59% for the other two cases). Therefore, the differences found were due more to a scale issue when combining the rice and livestock components than to differences in the N input values used for the calculations. Independent of the study level and with a slight decrease over time because of increasing use of N fertilizers in rice, the average system NUE values were very high compared with the literature, even when the livestock component is proportionally dominant. Reported NUE values up to 45–50% for mixed systems were only attained when the crop component (higher NUE) exceeded the area of the livestock component (lower NUE) in the system (Godinot et al., 2015; Westhoek et al., 2014). The average system NUE values calculated for this work (between 50–60%) are within the range proposed by the EU Nitrogen Expert Panel (2015) for mixed systems (40–60%), which are associated with a balanced system N fertilization. All the NUE results obtained in this study were directly associated with objectives 3 and 4 in terms of understanding NUE in the rice-livestock system of Uruguay. The latter interpretation about the balanced N fertilization is in accordance with the N balance results of this study which ranged from - 6 to + 5 kg N ha⁻¹ yr⁻¹ when analysed over a longer term. Deambrosi (2009) and Macedo et al. (2021), found that soil organic C and N were maintained in the long-term when a stable rice-pasture rotation including livestock was conducted, which is aligned with the close to neutral N balance values of this study.

The high NUE values and tight N balances at different scales of the Uruguayan rice-livestock system are accompanied by low N surplus values being another outcome of the fulfilment of objectives 3 and 4. For the three study levels, the N surplus was on average < 20 kg N ha⁻¹ yr⁻¹, interpreted as very low (< 60 kg N ha⁻¹ yr⁻¹) based on the tentative scheme proposed by the EU Nitrogen Expert Panel (2015) for mixed systems. But the low average N surplus value when considering the long term, hides an increasing trend in recent years almost doubling the N surplus. Therefore, the results are consistent with the generalized development pathway proposed by Dobermann et al. (2022), where NUE tends to decrease and N surplus increase over time. This is a critical point to consider when intensifying a system in a sustainable manner. Thus, it seems logical to identify the stages within the system towards that objective, avoiding extreme NUE (too

high or low) and high N surplus, potentially polluting the environment. The defined objectives and their fulfilment enabled the comparison of the aforementioned N parameters.

5.2 Combining system intensification with optimum N budget records

Reconnection of crops and livestock and inclusion of legume cover crops are increasingly mentioned in the literature for maintaining soil quality, recycling nutrients and minimizing environmental damage (De Faccio Carvalho et al., 2021; Peyraud et al., 2014; Taifouris & Martin, 2021) while maintaining or even increasing productive and economic margins (Ryschawy et al., 2012; Vogel et al., 2021). Such factors have enabled the Uruguayan rice-livestock system to reach very high rice productivity with continuously increasing yield despite relatively low N inputs from fertilizers (Pittelkow et al., 2016). Nevertheless, in recent years farmers have started to replacement pastures and livestock with soybean to diversify their incomes and benefitting rice yields when rotated with improved legume pastures. Despite the preliminary experiences being promising in terms of economic profit and rice yield after soybean, the longer-term effects could be negative. For example, soil N depletion has been reported when a stable rice-soybean rotation was maintained over time (Nishida, 2016) and after 5 years a rice-soybean rotation had 15% less N in the labile fraction of the soil organic matter in comparison to a rice-pasture rotation (Macedo et al., 2021). In relation to objectives 1 and 2 and after the assessment of different rotations, this study showed that despite reaching a higher system NUE, the rice-soybean rotation had the more negative N balance ($-23 \text{ kg N ha}^{-1} \text{ yr}^{-1}$) even when introducing a legume pasture cover crop between the soybean and rice crop. That rotation could benefit from longer duration pastures and inclusion of livestock in the long term. The results obtained linked to objective 1 after the DNDC model parametrization would appear crucial for a fine N tuning in those intensified scenarios.

Could the current rice-livestock rotation be intensified and at the same time reach higher NUE values, maintaining a tight N balance and low N surplus? Rice yields

may continue to increase adding more fertilizer N without adjusting the livestock-pasture component, and even more so with a greater percentage of native grassland over improved pastures. But a continuous increase in N fertilizer use at the current rate shift the NUE and N Surplus to ever less-sustainable values, as reported for several world systems (Dobermann et al., 2022; EU Nitrogen Expert Panel, 2015). An increase in N fertilizer inputs could generate proportionally greater N outputs because of more N removal in grain and higher N losses (Lin et al., 2007; Yu et al., 2013). That was found in this study when simulating a scenario of increasing N dose rate, resulting in a more negative N balance. Therefore, we believe that the Uruguayan rice-livestock system is ready to define new objectives in terms of continued sustainable intensification of the system. It could be possible to further integrate the system components, helping to reach higher N standards than reported here. On the one hand, it could be possible to avoid negative aspects associated with high N fertilizer load (excessive N losses and N Surplus). On the other hand, it could be possible to lift some limitations, like the low animal N retention in food products due to a low meat production. Accordingly, increased animal productivity has been proposed through better livestock management in Uruguay through a greater percentage of improved pastures, a slight increase in animal concentrated feed and better allowance for the age at the first pregnancy (Jaurena et al., 2021; Kanter et al., 2016).

A promising strategy would be to move from a scheme where animals and pastures are simply part of the rotation to a scheme where animals and pastures are managed and used deliberately in favour of the rice crop as another tradable N source. The application of this strategy can also provide positive feedback to the livestock component. Related practices are: a) to ensure the presence of legume pastures (N fixation) at least in some stages of the rotation, reducing rotating rice with grassland of low productivity, therefore low stocking rate and N recycling; b) inclusion of productive pastures (or cover-crops) for legume N fixation or for animal use between the summer tillage and the beginning of the rice crop, grazed at higher stocking rates to facilitate nutrient recycling; c) include pastures between the first and second rice which increase the forage volume (pasture + rice straw) to be used by animals at higher stocking rates. Additionally,

research is needed to determine stocking rates to ensure a defined amount of nutrients is recycled prior to the rice crop. To facilitate animal management and improve the distribution of manure, technologies like “virtual fencing” could be useful. All the proposals, including an increase of forage and animal stock, will increase the livestock component’s performance. Finally, complementary management strategies for the crop component like the wider use of improved N fertilizer technologies (e.g., coating, inhibitors, placement) (Fitts et al., 2014; Meng et al., 2020), and the use of materials that help to minimizing N losses, such as zeolites or rice by-products like husk (Oenema & Tamminga, 2005), should be considered.

5.3 Towards a national rice-livestock-other crops rotation N observatory

This study investigated past and current N budgets and opportunities for N management interventions. Once new N management strategies are identified and implemented, a system for monitoring annual N budgets should be put in place. The rice and livestock components both have historical and current records that are almost unique globally. This includes total amounts of fertilizer applied by rice farmers, the percentage of improved pastures of the total forage area, the animal tracking system which accounts for their movements throughout the entire animal life (from immediately after birth up to the arrival in the abattoir), and the possibility of remote sensing to estimate the forage productivity (Baethgen et al., 2016; Freeman & Orejas, 2015). Based on this data combined with GIS technology, it would be possible to develop a national rice-livestock-other crops rotation observatory for N. For each farm it would be possible to know exactly the amount of N removed in animal and grain beyond the farm gate boundary. For the same farm it would also be possible to know the amount of N input from fertilizers and based on remote sensing the percentage of improved pastures for BNF estimations. Where more efforts are needed is in estimating the systems internal fluxes including N losses. Related to objective 1, although the DNDC model was calibrated for several variables of the rice system in this study, key variables like the NH_3 volatilization (the main N loss in the rice system) must be validated against real volatilization data not measured in the country until now. If a good agreement is found, the model could be used to fine-tune N fertilizer

management, potentially linking to satellite measurements to verify detected hot spots of NH_3 emission (Cao et al., 2022). Selected representative sites distributed across the country could be established to monitor soil parameters.

The nutrient budgeting approach developed in this thesis for N could be applied to other nutrients with the aim of anticipating other potential nutrient issues instead of reacting once they appear. For example, soil phosphorous (P) contents are generally low under natural conditions across Uruguay (Hernández & Meurer, 1998), P is added to rice based on results of response-trials, often at rates which exceed grain removal, generating positive P balances in the long term. Since around 2010 objective information for P management has been available (Hernández et al., 2013) and soil testing has shown medium to high concentrations. Therefore, from 2010 up to now, P fertilization has been slightly reduced, following a replenishing approach of P removed in the preceding growing season, so maintaining the high soil P levels generated by positive P balances for decades.

However, the opposite has been observed for potassium (K). From the beginning of rice production in Uruguay to about 10 years ago, K was not considered in the rice fertilization programs. Then K deficiency symptoms appeared and yield responses to K fertilization at the field level were reported. After several decades of accumulated negative balances, K fertilizers started to be added and at present are used in almost all rice areas. Independent of that, rice nutrition in Uruguay has always been dealt as a matter of two factors: the crop nutrient response and/or requirement and the addition of fertilizers, with less attention to other components and processes outside that bi-directional relationship.

Several studies have stressed reduced requirements of fertilizer for crops when integrated into a lowland rotation with cattle (Carlos et al., 2020; De Oliveira Denardin et al., 2020), showing the importance other effects outside the crop component. Currently, other nutrients like sulphur (S) are being considered but focusing only on the rice component without considering other external aspects. For example, the S concentration in rain ranges from 0.26 to 1.0 mg S l⁻¹ in parts of Uruguay (Zunckel et al., 2003). Considering the annual average rain precipitation of about 1300 mm yr⁻¹ (INIA, 2021), and the average rice yield of this

study ($8.1 \text{ Mg ha}^{-1} \text{ yr}^{-1}$), the wet atmospheric deposition of S is close to or even higher than that removed in grain (1 g kg^{-1} ; Dobermann & Fairhurst, 2000). Such examples indicate the importance of developing nutritional strategies that consider the whole system.

Returning to N, the system is now facing new challenges in terms of continuously increasing rice yields largely through increasing N fertilizer applications. Based on this study, the system is now at a point where maintaining the current N strategy would mean starting to lose the current high NUE, low surplus and neutral balances. The Uruguayan rice-livestock, which has always been well integrated, should explore a re-boosting among component to continue intensifying in a sustainable way.

5.4 References

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6 CONCLUSIONS

This work has provided an understanding of the general stability of the rice-livestock system of Uruguay over time, and the very high NUE, low N surplus and neutral N balances that have been achieved. During the study period, the livestock component sustained the rice component in terms of N, allowing very high rice productivity with relatively low N fertilizer use. Adjustments of the livestock component in rotation with rice must be made if a continued increase in rice yield is to be achieved without greatly increasing N fertilizer use. For this, greater contribution of N from BNF in legume pastures and higher stocking rates to recycle more N from the livestock component should be considered. Improvements in the livestock component would allow better economic margins, so offering an alternative to continuous rice-soybean agriculture which is currently expanding. Because these systems present subtle negative balances, their long-term sustainability also depends on including pastures and livestock in the rotation.

Simulation tools such as the DNDC model show good agreement with measured data on complex rotational systems, allowing a good simulation of the N processes in the system. Therefore, it should be useful for identifying potential problems in advance and making tight management of the N budget of the different systems. That tight adjusted N management for different situations is key since, as shown in this study, a similar N management in rice practised across different situations generates different results in terms of NUE, balance and surplus over time. However, further works aimed to validate missing information such as NH_3 volatilization losses are necessary to precisely adjust N in the system because of its importance. Considering all the data capabilities and availability, it should be possible to develop a national platform to monitor the system regularly in terms of N and other nutrients.

APPENDICES

Appendix A Supplementary Materials for Chapter 2

A.1 Additional details of the experimental site

A.1.1 Long-term experiment characteristics and components

The rice-pasture-other crops long-term experiment (LTE) is located in the east of Uruguay at the National Institute of Agricultural Research (INIA) at Paso de la Laguna research station. Established in 2012, it is situated on a paddock of 13 ha, occupying an effective area of 7.2 ha. It is composed of 60 plots, grouped in 3 experimental blocks. Each experimental unit is 20 m wide and 60 m long with an effective experimental area of 1150 m². It was set up to allow all management operations with commercial machinery and to include animal production as on commercial farms. It is composed of 6 rotations which vary in crop intensity and pasture length, all related to the business-as-usual production system in Uruguay, which is a rice - 3.5 years pasture rotation (RI-PAST). The other rotations are I: continuous rice (RI-CONT), II: rice-soybean-rice-sorghum (RI-SOY-RI-SOR), III: rice-short/pasture (RI-shoPAST), IV: rice-soybean-medium length pasture (RI-SOY-medPAST) and V), and continuous rice-soybean (RI-SOY). The experiment is laid out in a synchronic randomized complete block design with all the phases present at any time (Patterson, 1964) (see **Error! Reference source not found.**). Main soil properties at the beginning of the LTE are presented below (**Table A-1**).

Table A-1 Soil physical and chemical properties at the beginning of the long-term experiment (2012) in two soil depths (0-15 and 15-30 cm).

Property	Depth (cm)	
	0–15	15–30
Clay, g kg ⁻¹	29.7	31.6
Sand, g kg ⁻¹	19.1	19.4
Silt, g kg ⁻¹	51.2	49.0
pH (1:1 H ₂ O)	5.6	6.5
Organic carbon, g kg ⁻¹	14.2	7.2
Total nitrogen, g kg ⁻¹	1.4	0.8
Extractable N-NO ₃ , mg kg ⁻¹	6.2	5.1
Extractable P (citric acid), mg kg ⁻¹	10.3	2.8
Cation exchange capacity, cmol _c kg ⁻¹	12.6	14.3
Exchangeable Ca, cmol _c kg ⁻¹	7.0	8.9
Exchangeable Mg, cmol _c kg ⁻¹	2.8	3.7
Exchangeable K, cmol _c kg ⁻¹	0.3	0.2
Exchangeable Na, cmol _c kg ⁻¹	0.4	0.8

A.1.2 Agronomic and crop management of the long-term experiment

Prior to the start of the experiment, the land was tilled twice (disc harrow) and levelled by passing twice with a land planer as part of the preparation of the surface rice irrigation system. After this initial tillage, all experimental treatments have been maintained under no-till until today. Rice (*Oriza sativa* L.) is seeded between mid-October and mid-November, permanently flooded around 35 days after seeding up to 20 days after the heading stage (50%) and harvested between the end of March and mid-April. Soybean (*Glycine max* (L) Merr.) was seeded between mid- November and mid-December and harvested up to the middle of May. Since 2015, a surface supplementary irrigation was applied up to three times during critical crop stages, using the same irrigation system used for rice. Both, permanent pastures of the RI-PAST rotation and annual pastures used as cover crops in the three evaluated rotations were broadcast seeded immediately before or after the cash crops harvest.

Fertilizer was applied annually in cash crops and pastures but with different criteria. The rice crop in the RI-PAST rotation was fertilised based on local recommendations for nitrogen (Castillo et al., 2015), phosphorus (Hernández et al., 2013) and potassium (Deambrosi et al., 2015), following a sufficiency level

approach based on soil analysis. Phosphorus and potassium fertilisations for the RI-SOY and RI-CONT rotations were calculated to replenish the nutrient amount removed in the previous growing season with the grain, while nitrogen fertilisation of RI-CONT was determined for a target rice yield of 10000 kg ha⁻¹. Pasture cover crops were not fertilized, and pasture crops of the RI-PAST rotation were fertilised with N, P and K at establishment and re-fertilised with phosphorous at the second pasture year based on local blanket recommendations. A detailed description of the fertilization practice and applied nutrient amounts can be found in **Table A-2**.

Around thirty days before the summer cash crops were established, a systemic herbicide was applied to the pasture cover-crops of the three rotations or to the final pasture of the RI-PAST rotation, establishing a chemical fallow up to the seeding date of the next crop.

Table A-2 Average fertilisation applied to the rice-pasture (Ri-Past), rice-soybean (Ri-Soy) and continuous rice (Ri-Cont.) between 2012-13 to 2020-21.

Rotation	Crop	Fertilisation approach	N	P ₂ O ₅	K ₂ O
----kg ha ⁻¹ crop ⁻¹ ----					
Ri-Past	Rice 1	Sufficiency level	74	15	28
	Rice 2	Sufficiency level	98	28	52
	Pasture 1 st year	Blanket *	23	45	23
	Pasture 2 nd year	Blanket	--	35	--
Ri-Soy	Rice	Replacement/Sufficiency level**	89 **	48	65
	Soybean	Replacement	4 ***	68	59
Ri-Cont.	Rice	Replacement/Target yield****	147 ****	71	43

*Fertilisation based on historical response dose trials developed by the national research system

**Nitrogen (N) fertilisation of rice in the rice-soybean rotation was based on a sufficiency level approach

***Nitrogen fertilisation to soybean in the rice-soybean rotation was due to the use of a binary (N-P) fertiliser when P₂O₅ was added

****Nitrogen fertilisation of rice in the continuous rice rotation was based on a 10 Mg ha⁻¹ rice yield target, assuming an average N input of biological fixation by the legume cover crop of 40 kg N ha⁻¹ yr⁻¹, a N recovery efficiency of 50% for N fertiliser inputs, and 7 kg N ha⁻¹ yr⁻¹ from atmospheric deposition.

A.1.3 Livestock management and main animal N considerations

Pasture in RI-PAST rotation was grazed with sheep averaging 8-10 times a year for each plot being from 7 to 10 days in each grazing period. Because this short period meat production was estimated and not measured. Ovine production was estimated as a function of measured forage (kg DM ha⁻¹ yr⁻¹) and a grazing efficiency of 60% (Crempien, 1983) multiplied by meat N concentration (g kg⁻¹). For wool consideration, 2 kg animal yr⁻¹ was assumed. An amount of 15 kg DM animal day⁻¹ was assumed to produce 1 kg meat (**Table A-3**).

A.1.4 Nitrogen fixation calculations considered in the N balance

Biological N fixation (BNF) was calculated based on the dry matter (DM) annual production. Pastures were measured seasonally each year and the botanical composition was recorded. The BNF was estimated multiplying each aboveground legume pasture species DM production (Mg ha⁻¹ yr⁻¹) by a fixation coefficient (kg N Mg DM⁻¹) and including an estimation of the belowground N in nodules, stolon, or roots (**Table A-3**). In the case of soybean, BNF was estimated based on measurements in 2019-20 and 2020-21 growing seasons of the harvest index at R6 stage. We assumed that from the total amount of plant N at R6, 75% was derived from atmospheric N (Alves et al., 2003), and to consider N in roots and nodules, we multiplied that number by 1.5 following Herridge et al. (2008). Literature values and factors considered for calculations are presented in **Table A-3**.

Table A-3 Summary of data used for N balances calculations and DNDC model parametrization, and data sources. Parameter	Range	Source	Reference
Pasture Biological N Fixation	30 kg N Mg ⁻¹ <i>white clover</i> aboveground dry matter (in mixture with grasses) 21 kg N Mg ⁻¹ <i>Lotus spp.</i> aboveground dry matter 28 kg N Mg ⁻¹ <i>Trifolium alexandrinum</i> L. 1.6 = Coefficient for whole-plant N ₂ fixation (f_{yield})	Literature	FAO, 2018 Iannucci et al., 1996 Labandera et al., 1988 Peoples and Craswell, 1992
N in animal products	Ovine meat: 2.27%	Literature	ARC, 1980
	Wool: 16.5%		
Botanical composition (average 9 years)	9% white clover, 10% lotus spp., 56% tall fescue, 7% natural grasses, 18% dry dead pasture	INIA LTE records	Unpublished, used in this study

Rice N content and harvest index (HI)	47% Rice HI (9 year average) 1.16% Rice grain N 0.65% Rice straw N	INIA Rice – Pasture Long term Experiment	Unpublished INIA Deambrosi et al., 2019
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Table A-4 (Continued)

Parameter	Range	Source	Reference
Soybean N content and harvest index (HI)	32% Soybean HI (2 year average) HI=Grain DM/ total DM at R6 stage 5.8% Soybean Grain N 2.6% Soybean Straw N (2 year average of stems, leaf and pods)	INIA Rice – Pasture Long term Experiment Literature	Unpublished INIA Hintz & Albrecht, 1994 Salvagiotti et al., 2009

Table A-5 Accuracy of the model predictions of rice, soybean and pasture yields (kg ha⁻¹), N uptake (kg N ha⁻¹), soil N-NH₄ solution (kg ha⁻¹) and N-N₂O emission (g ha⁻¹ d⁻¹) in the three rice-pasture rotations in different periods of the long-term experiment. The rotations are defined in **Error! Reference source not found.**. The model goodness-of-fit indices are defined in Equations (2-1) (2-3).

Rotation	Crop/Sequence	Index	2012–21			2019–20		2020–21		2019–20	2020–21
			Rice	Soybean	Pasture	N uptake	Soil N-NH ₄	N uptake	Soil N-NH ₄	N-N ₂ O emission	N-N ₂ O emission
RI-PAST	1 st Rice	ME	-206.89			-1.07	1.23	-4.85	0.09	-0.27	-0.06
		IA	0.82			0.97	0.39	0.96	0.23	0.06	0.41
		MEF	0.95			0.94	-0.08	0.92	0.01	-0.03	0.36
	2 nd Rice	ME	-389.21			0.76		1.71			
		IA	0.72			0.89		0.95			
		MEF	0.20			0.83		0.92			
	Pasture 2013/14 and 2018/19 *	ME			136.50					-	-
		IA			0.19					-	-
		MEF			-2.24					-	-
	Pasture 2014/15 and 2019/20 **	ME			-186.06						
		IA			0.91						
		MEF			0.79						
		ME			552.67						
		IA			0.35						

Pasture 2015/16 and 2020/21 ***	MEF	-13.68				
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Table A-6 (Continued)

Rotation	Crop/Sequence	Index	2012–21			2019–20		2020–21		2019–20	2020–21
			Rice	Soybean	Pasture	N uptake	Soil N-NH ₄	N uptake	Soil N-NH ₄	N-N ₂ O emission	N-N ₂ O emission
	Pasture 2016 and 2021 ****	ME			420.80						
		IA			0.45						
		MEF			-7.75						
		ME	-198.27	-145.49		1.23	1.65	-1.75	0.91		
		IA	0.58	0.91		0.94	0.51	0.96	0.58		

RI-SOY ¶	Rice-Soy sequence	MEF	0.46	0.76		0.84	-0.88	0.92	-1.18		
		ME	279.1	-59.19						0.25	-
	Soy-Rice sequence	IA	0.77	0.78						0.02	-
		MEF	0.41	0.44						-0.19	-
RI-CONT	Rice	ME	-236.27			0.54	1.27	-4.49	1.04	-	-
		IA	0.78			0.96	0.56	0.96	0.40	-	-
		MEF	0.45			0.92	-0.20	0.92	-1.22	-	-
Averages		ME	-204.55	-97.52	242.91	0.37	1.38	-2.34	0.68	0.01	-0.06
		IA	0.79	0.92	0.71	0.95	0.48	0.96	0.54	0.01	0.41
		MEF	0.50	0.81	0.27	0.89	-0.27	0.93	0.05	-0.11	0.36

*, **, ***, **** and , ¶ see # and ## in **Error! Reference source not found.** for conceptual framework.

Image A 1 Mid-season photos of plots from the continuous rice rotation (left side – R1-Az) and the rice-pasture rotation (right side – R4-Az1) of the long-term experiment in January 2022 after panicle initiation stage. Duckweed (*Lemna Minor* L.) was present in the continuous rice rotation (left side) in 2019-20 and 2020-21. Credits: Dr. Álvaro Roel.



A.2 References appendix A

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Appendix B Supplementary Materials for Chapter 3

B.1 The spatial integration among rice and livestock components

Rice crops usually grow for around five months from October-November to March-April ("A" in Error! Reference source not found.). On average, nearly 60% is seeded over different types of pastures used by cattle, and the reminder area is seeded over rice stubble of a first rice crop (see rotation percentage in Error! Reference source not found.). Management of the pasture area prior to the rice seeding is following two strategies. The first one, and the one recommended by research (Gamarra, 1996) and preferred by farmers, consists in an anticipated land preparation in summer for the next rice season (December-March, "B" in Error! Reference source not found.). The second one prepares the land later, between August to October ("C" in Error! Reference source not found.). In both cases, land preparation includes the generation of a chemical fallow, two or three ploughings and one or two soil surface leveling. In the first case, around 50% of the prepared summer land is seeded with an annual Graminea pasture for animal grazing. On the other half of the area, a spontaneous regeneration of forage will occur and, depending on the type and botanically composition of those previous pastures, will define the potential animal use for grazing ("D" in Error! Reference source not found.). Typically, the summer land preparation includes a minimum tillage, immediately before the rice seeding ("E" in Error! Reference source not found.). If a second rice crop is seeded over the first rice from the previous year, a common practice after the first rice harvest is grazing the stubble up to August-September ("F" in Error! Reference source not found.), followed by the land preparation for the second rice crop ("G" in Error! Reference source not found.).

Different pasture types precede the rice crop each year: natural, improved, or annual Graminea pastures ("H" in Error! Reference source not found.). The first ones are composed by native spp. and is the staple forage base of the entire country. Improved pastures are a mix of Graminea spp. (*Lolium multiflorum* or *Festuca Arundinacea*) and forage legumes (*Trifolium Repens* and *Lotus corniculatus*). Annual Graminea seeded after the summer land preparation is a pure pasture of *Lolium multiflorum*. Typically, improved pastures are seeded after the rice harvest, over the stubble by plane, and on average reaches a length between 3 and 4 years ("I" in Error! Reference source not found.). When the forage mix includes legume pastures ("H" and "I" in the Error! Reference source not found.), phosphorus (P) fertilisation is typically made at least once

in the forage length. In “H”, P is applied at the pasture sowing date and in pastures seeded over rice stubble (“I”) is applied at the beginning of the second year because the first year the forage pasture use residual P from the rice crop of the previous year. In both cases P dose is around 50-60 kg ha⁻¹ P₂O₅. After summer land preparation, annual *Lolium multiflorum* is seeded at broadcast without nitrogen (N) addition and will be present up to the end of August when the chemical fallow followed by a minimum tillage prior the rice crop will be held (“D” in Error! Reference source not found.). All the areas where a rice crop will be sown (pastures and rice stubble) will be grazed by cattle up to around 1 month prior to the seeding rice date. Then the livestock will be removed to another area of the rice rotation.

B.1.1 The integration among rice and livestock farmers

It is often said that rice and livestock components are strongly integrated. This integration is based in the good relationship among the rice and livestock farmers, which mainly are different actors. Basically, rice production is conducted by rice farmers renting the land to the livestock farmers (DIEA, 2003). When a livestock farmer gives allows a rice farmer to develop a rotation, various resources are shared, e.g. the rice farmer could provide water to animal drinking basically in summer. Frequently, the livestock farmer provides with cattle to the ricer farmer to manage the rice stubble for the next rice crop growing season. The rice milling of the rice farmer, ensures the money income related to the land rental to the livestock farmer, independently of the final grain yield or economically result reached by the rice farmer. In addition, the rice farmer frequently executes labors in livestock farmer land, of the specific rice land and livestock surface as road maintenance, pasture seeding and fertilization among others.

B.2 Forage production and nitrogen transference from livestock to rice component

Forage production (kg dry matter ha⁻¹ yr⁻¹) of the natural pasture component was estimated averaging data of remote sensing from 2001 to 2016 of the main agro-ecological regions where the rice rotation take place (Asuaga et al., 2019). Semi-natural (Martínez, 2011) and temporary pasture productivity (INIA, unpublished data) for the whole period were estimated by regression analyses of multi-year dry matter production surveys (10 and 7 years respectively) among each component and the natural pasture production (**Figure B-1**).

Nitrogen recycled by livestock was estimated based on estimated available forage (**Figure B-1**), and literature values for forage chemical composition, animal forage

utilization and animal nitrogen use efficiency (**Table B-1**). Stocking rate was not considered in the recycling calculation because we assume that the available forage and the present livestock units are in equilibrium. Animal direct deposition (ADD), defined as the amount of N recycled from the livestock to the rice component, was considered over 6 months of grazing (“D, F and H” in Error! Reference source not found.). The N content of dung depositions (faeces plus urine) has been measured for up to 90 days in crop-livestock system in the south of Brazil (Carpinelli et al., 2020) and cow urine has been shown to contribute to a better nutrient management in integrated system as well as increasing the rate of organic matter decomposition (Pradhan et al., 2018). And although it can be expected that the animal dejections would have a very uneven distribution (Carpinelli et al., 2020), we expected that the tillage and land levelling prior to crop establishment would increase the uniformity of the N distribution from ADD.

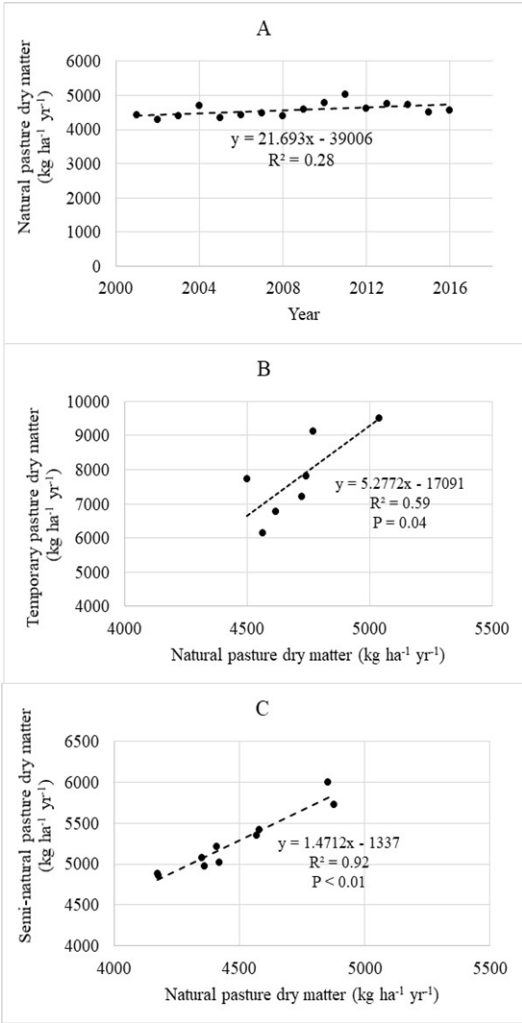


Figure B-1 Forage dry matter production (kg ha⁻¹ yr⁻¹) of the three pasture components of the rice-livestock system. (A) natural pasture production across selected years, (B) regression among average natural pasture production of the rice-livestock system and pasture production of temporary pastures of the INIA’s long term experiment (rice-

pasture-other crops) and (C) regression among contiguous fields of natural and semi-natural pastures Schematic representation of the rice-livestock system of Uruguay and the annually average percentage of type of rotation seeded with rice each year. Vertical red dotted line indicates a typical rice growing season.

B.3 Data and coefficients used in calculations.

Table B-1 Summary of data used for N balances calculations and data sources.

N Flux	Parameter	Range	Source	Reference
N Inputs	Fertilizers	45-85 kg N ha ⁻¹	INIA, rice mills	INIA, 2020
	Biological N Fixation	2.5 kg ha ⁻¹ free-living microorganisms	Literature	FAO, 2018
		30 kg N Mg ⁻¹ <i>White Clover</i> aboveground dry matter (in mixture with grasses)	Literature	Labandera et al., 1988
		21 kg N Mg ⁻¹ <i>Lotus spp.</i> aboveground dry matter 1.6 = Coefficient for whole-plant N ₂ fixation (f _{yield})		Peoples and Craswell, 1992 FAO, 2018
	Atmospheric Deposition	3.7-9.0 kg N ha ⁻¹ based on rain precipitation* *1394 mm ha ⁻¹ yr ⁻¹ (SD=265 mm)	Literature Weather stations	Zunckel et al., 2003 Casartelli et al., 2008 INIA, 2021

Table B-2 (Continued)

N Flux	Parameter	Range	Source	Reference
N Outputs	Primary food production	Rice: 5.2-8.9 Mg ha ⁻¹	Governmental statistics	DIEA, 2020, 2005
		Meat: 60-78 kg ha ⁻¹	Governmental statistics, literature, calculated in this work	Aguirre, 2018 INAC, 2020
	% N in food products	Bovine: 2.43%	Literature	ARC, 1980
		Ovine: 2.27%	Literature	Biel et al., 2019 FAO, 2018
		Rice grain: 1.15%	Literature	Deambrosi et al., 2019
		Wool: 16.5%	Literature	ARC, 1980

Table B-3 (Continued)

N Flux	Parameter	Range	Source	Reference
N Outputs	Gas N losses	5.9 kg N ha ⁻¹ volatilisation from livestock	Literature	FAO, 2018 Martínez et al., 2010
		0.9 kg N ha ⁻¹ denitrification	Literature	Tarlera et al., 2006 FAO, 2018
		21% of total N applied to rice	Literature	Castillo, 2018
	N leaching	2.1 kg ha ⁻¹	Literature	FAO, 2018
N recycling	N by-products (Rice Bran)	19 kg N ha ⁻¹ produced at rice component area (0.17 Mha) 0.4 kg N ha ⁻¹ once assigned to all livestock area (8.39 Mha)	Calculated in this work	
	Animal direct deposition to rice	14-30 kg N ha ⁻¹	Calculated in this work	

Table B-4 (Continued)

N Flux	Parameter	Range	Source	Reference
N recycling	Forage dry matter yield (aboveground)	4.5 Mg ha ⁻¹ yr ⁻¹ Natural grassland 5.3 Mg ha ⁻¹ yr ⁻¹ Semi-natural pastureland 7.5 Mg ha ⁻¹ yr ⁻¹ Temporal pasture 5.7 Mg ha ⁻¹ yr ⁻¹ Rice straw	Literature Calculated in this work	Asuaga et al., 2019; Deambrosi et al., 2019; Martínez, 2011
N recycling	Forage botanical composition (average 3 years)	85% natural grasses spp, 15% dead pasture. Natural grassland 8% wite clover, 9.4% lotus spp., 64% natural grasses, 18% dry dead pasture semi-natural pastureland 8% wite clover, 9.4% lotus spp., 56% tall fescue, 7% natural grasses, 19% dry dead pasture Temporal pasture	INIA Rice – Pasture Long term Experiment	Unpublished
	Forage N content (aboveground dry matter)	3.9% White clover 4.2% Lotus spp. 3.2% Tall Fescue 1.5% Natural pasture 0.5% Dead pasture 0.6% Rice straw	Literature	Deambrosi et al., 2019 Carámbula, 2003 Ayala and Carámbula, 1994

Table B-5 (Continued)

N Flux	Parameter	Range	Source	Reference
	Forage utilisation	55% Natural grassland 60% Semi-natural pastureland 65% Temporal pasture 35% Rice straw*	Literature *Expert judgment	Crempien, 1983
Internal efficiencies	Animal NUE	15-20% Bovine 20% Ovine	Literature	Watson and Atkinson, 1999

B.4 References appendix B

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Appendix C Supplementary Materials for Chapter 4

Table C-1 Summary of data used for N balances calculations and DNDC model parametrization, and data sources.

N Flux	Parameter	Range	Source	Reference
N Inputs	Fertilizers	40-105 kg N ha ⁻¹ Northern Region (HFSR) 43-98 kg N ha ⁻¹ Central Region (MFMR) 57-102 kg N ha ⁻¹ Eastern Region (LFLR)	INIA, rice mills	INIA, 2020
	Biological N Fixation	2.5 kg ha ⁻¹ free-living microorganisms	Literature	FAO, 2018
		30 kg N Mg ⁻¹ <i>White Clover</i> aboveground dry matter (in mixture with grasses) 21 kg N Mg ⁻¹ <i>Lotus spp.</i> aboveground dry matter 1.6 = Coefficient for whole-plant N ₂ fixation (f _{yield})	Literature	Labandera et al., 1988 Peoples and Craswell, 1992 FAO, 2018 Iannucci et al., 1996

Table C-2 (Continued)

N Flux	Parameter	Range	Source	Reference
N Inputs	Atmospheric Deposition	3.7-9.0 kg N ha ⁻¹ based on rain precipitation* *1394 mm ha ⁻¹ yr ⁻¹ (SD=265 mm)	Literature Weather stations	Zunckel et al., 2003 Casartelli et al., 2008 INIA, 2021
N Outputs	Primary food production (rice or meat entered to the mill or abattoir)	Rice: 6.2-8.7 Mg ha ⁻¹ Northern Region (HFSR) Rice: 5.6-8.6 Mg ha ⁻¹ Central Region (MFMR) Rice: 5.9-8.4 Mg ha ⁻¹ Eastern Region (LFLR)	Governmental statistics	DIEA, 2005; 2022
		Meat: 47-60 kg ha ⁻¹ Northern Region (HFSR) Meat: 64-78 kg ha ⁻¹ Central Region (MFMR) Meat: 69-77 kg ha ⁻¹ Eastern Region (LFLR)	Governmental statistics, literature, calculated in this work	Aguirre, 2018 INAC, 2020

Table C-3 (Continued)

N Flux	Parameter	Range	Source	Reference
N Outputs	N in food products (%)	Bovine: 2.43%	Literature	ARC, 1980
		Ovine: 2.27%	Literature	Biel et al., 2019 FAO, 2018
		Rice grain: 1.15%	Literature regional trial multiple site/years	Deambrosi et al., 2019
		Wool: 16.5%	Literature	ARC, 1980
N recycling	N by-products (Rice Bran)	19 kg N ha ⁻¹ produced at rice component area (0.17 Mha) 0.4 kg N ha ⁻¹ once assigned to all livestock area (8.39 Mha)	Calculated in this work	
	Animal direct deposition to rice	14-30 kg N ha ⁻¹	Calculated in this work	

Table C-4 (Continued)

N Flux	Parameter	Range	Source	Reference
N recycling	Forage dry matter yield (aboveground)	4.5 Mg ha ⁻¹ yr ⁻¹ Native grassland 5.3 Mg ha ⁻¹ yr ⁻¹ Semi-natural pastureland 7.5 Mg ha ⁻¹ yr ⁻¹ Temporal pasture 5.7 Mg ha ⁻¹ yr ⁻¹ Rice straw	Literature and Calculated in this work	Asuaga et al., 2019; Deambrosi et al., 2019; Martínez, 2011
	Forage botanical composition (average 3 years)	85% natural grasses spp, 15% dead pasture. Natural grassland 8% wite clover, 9.4% lotus spp., 64% natural grasses, 18% dry dead pasture. Semi-natural pastureland 8% wite clover, 9.4% lotus spp., 56% tall fescue, 7% natural grasses, 19% dry dead pasture. Temporal pasture	INIA Rice – Pasture Long term Experiment	Unpublished
	Forage N content (aboveground dry matter)	3.9% White clover 4.2% Lotus spp. 3.2% Tall Fescue 1.5% Natural pasture 0.65% Rice straw	Literature	Deambrosi et al., 2019 Carámbula, 2003 Ayala & Carámbula, 1994

Table C-5 (Continued)

N Flux	Parameter	Range	Source	Reference
N recycling	Forage utilisation	55% Natural grassland 60% Semi-natural pastureland 65% Temporal pasture 35% Rice straw*	Literature *Expert judgment	Crempien, 1983
Internal efficiencies	Animal NUE	18% Bovine 20% Ovine	Literature	Watson & Atkinson, 1999

C.1 References appendix C

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