Alternativas de intensificación sostenible de sistemas agrícola-ganaderos basadas en rotaciones y estrategias ganaderas contrastantes

Fabiana Julia Pereyra Goday

Doctorado en Ciencias Agrarias

Julio, 2024

Alternativas de intensificación sostenible de sistemas agrícola-ganaderos basadas en rotaciones y estrategias ganaderas contrastantes

Fabiana Julia Pereyra Goday

Doctorado en Ciencias Agrarias

Julio, 2024

Tesis aprobada por el tribunal integrado por Ing. Agr. (PhD) Valentín Picasso, Ing. Agr. (PhD) Guillermo Siri-Prieto y Ing. Agr. (PhD) Fabio Montossi, el 9 de octubre de 2024. Autor: Ing. Agr. (MSc) Fabiana Julia Pereyra Goday. Directora: Ing. Agr. (PhD) Jordana Rivero. Codirector: Ing. Agr. (PhD) Walter Ayala.

Agradecimientos

A la Facultad de Agronomía, por permitirme desarrollar mis estudios de Doctorado.

A INIA por financiar este proyecto a través de una beca de doctorado (2019-2023) y ANII, por financiar este proyecto a través de una beca de movilidad (MOV_CA_2021_1_171482).

A mi tutora, Jordana Rivero, y cotutor, Walter Ayala, por la paciencia y el apoyo durante estos años.

A Pablo Rovira, responsable técnico del experimento de largo plazo Palo a Pique, por los consejos y aporte técnico a lo largo de estos años de trabajo.

A los investigadores del instituto Rothamsted Research, especialmente Asma Jebari y Graham McAuliffe, por recibirme en North Wyke y permitirme formar parte de su equipo de trabajo.

A Michael Lee, Taro Takahashi, Jesús Castillo y José Terra, por sus consejos y aportes a este trabajo.

Al personal de INIA Treinta y Tres, especialmente a los equipos de Pasturas, Ganadería y Sustentabilidad, por la calidad de los datos obtenidos y la buena disposición siempre.

A mi familia, por el apoyo incondicional.

Tabla de contenido

Página

Resumen

La intensificación sostenible (IS) de los sistemas de producción busca el incremento de los rendimientos sin comprometer la salud de los ecosistemas. Como alternativa de IS se han propuesto los sistemas integrados agrícola-ganaderos. Las rotaciones de cultivo-pastura son parte de estos sistemas e implican una fase de pastura incluida en la secuencia de cultivos. Las principales razones para incluir pasturas en los sistemas de cultivo son la baja productividad de los pastizales naturales y el aumento del rendimiento de los cultivos después de una fase de pastura. El objetivo fue evaluar de forma multidimensional alternativas de intensificación de sistemas agrícolas ganaderos para el período desde mayo 2019 a abril 2022, utilizando para el análisis la información obtenida en el experimento de largo plazo Palo a Pique, de INIA Treinta y Tres. Los sistemas evaluados fueron RA (cultivo continuo), RC (dos años igual a RA, dos años de pasturas), RL (dos años igual a RA, cuatro años de pasturas) y RF (pastura continua con festuca). La producción de peso vivo (PV) fue mayor en RA y RC (426 y 418 kg PV/ha) que en RL (369 kg PV/ha) y RF (310 kg PV/ha). La producción de grano fue 10 %, 16 % y 9 % menor en soja, trigo y sorgo en RA. Las estimaciones de emisiones de gases de efecto invernadero (GEI) fueron 11,3, 11,8, 11,8 y 16,4 kg CO2-eq/kg de PV producido para RA, RC, RL y RF. Los promedios de emisiones de los cultivos fueron 1,23, 0,53 y 0,52 kg CO2 eq/kg para soja, trigo y avena. La NUE (eficiencia de uso de nitrógeno) del sistema fue de 43,4 %, 28,1 %, 29,3 % y 5,5 % para CC, SR, LR y FR. El balance de N en el sistema fue de 4,9 ± 1,52, 41 ± 3,1, 29 ± 2,2 y 64 ± 3,5 para RA, RC, RL y RF. En promedio el margen bruto fue 285, 200, 181 y 197 US$/ha para RL, RC, RA y RF. Los sistemas agrícola-ganaderos con una fase de pastura (corta o larga) en la rotación permiten mejorar diversos indicadores de sostenibilidad.

Palabras clave: sostenibilidad, rotaciones cultivo-pastura, análisis de ciclo de vida, eficiencia de uso de nitrógeno

Sustainable intensification alternatives for agricultural-livestock systems based on rotations and contrasting livestock strategies

Summary

Sustainable intensification (SI) of production systems aims to increase yields without compromising the health of ecosystems. Integrated crop-livestock systems have been proposed as an SI alternative. Crop-pasture rotations are part of these systems and involve a pasture phase included in the crop sequence. The main reasons for including pastures in cropping systems are the low productivity of natural pastures and the increased crop yields after a pasture phase. The objective was to evaluate multidimensionally alternatives for intensifying crop-livestock systems for the period from May 2019 to April 2022, using information obtained from the long-term Palo a Pique experiment by INIA Treinta y Tres. The evaluated systems were RA (continuous cropping), RC (two years of RA, two years of pastures), RL (two years of RA, four years of pastures) and RF (continuous pasture with fescue). Liveweight (LW) production was higher in RA and RC (426 and 418 kg LW/ha) than in RL (369 kg LW/ha) and RF (310 kg LW/ha). Grain production was 10%, 16% and 9% lower in soybean, wheat and sorghum in RA. The greenhouse gas (GHG) emission estimates were 11.3, 11.8, 11.8 and 16.4 kg CO2 eq/kg of LW produced for RA, RC, RL and RF. The average emissions of crops were 1.23, 0.53 and 0.52 kg CO2 eq/kg for soybean, wheat and oat. The NUE (nutrient use efficiency) of the system was 43.4%, 28.1%, 29.3% and 5.5% for RA, RC, RL and RF. The N balance at the system level was 4.9±1.52, 41±3.1, 29±2.2 and 64±3.5 for RA, RC, RL and RF. On average, the gross margin was 285, 200, 181 and 197 US$/ha for RL, RC, RA and RF. Crop-livestock systems with a pasture phase (short or long) in the rotation can improve various sustainability indicators.

Keywords: sustainability, pasture crop rotations, life cycle assessment, nitrogen use efficiency

1. Introducción

1.1. Contexto mundial y regional

De acuerdo con lo reportado por United Nations (2022), se estima que durante la próxima década la población mundial crecerá a una tasa anual de 0,9 % y la demanda mundial por productos de origen agrícola se incrementará un 1,1 % anual (Organisation for economic co-operation and development/Food and Agriculture Organisation, 2022), esta última impulsada en mayor proporción por cambios en la demografía más que por aumentos en el consumo per cápita. En este contexto, asegurar el suministro de alimentos se plantea como una oportunidad para los países productores, al tiempo que surgen desafíos asociados con incrementar la productividad (Eisler et al., 2014). A esto se suma un escenario de variabilidad en los precios tanto de insumos como de productos (Organisation for economic co-operation and development/Food and Agriculture Organisation, 2022), así como también la incidencia de la recuperación pospandemia COVID-19 y los actuales conflictos bélicos entre Rusia y Ucrania (Rawtani et al., 2022).

Según lo reportado por Uruguay XXI (2022), durante 2022, carne, soja, lácteos y arroz representaron el 47 % de las exportaciones en dólares americanos. Estos productos tuvieron un incremento en el total exportado de un 30 % en los últimos diez años, en US$ totales. Para el mismo período, se observaron cambios en la composición del stock ganadero a faena, con mayor proporción de faena de machos y hembras jóvenes (Dirección Estadísticas Agropecuarias-Ministerio de Ganadería, Agricultura y Pesca, 2022), sumado a un incremento en el área de pasturas sembradas de 17% para el período 2013-2021, lo cual evidencia un proceso de intensificación de la producción ganadera. Sin embargo, escenarios de volatilidad de precios en el mundo, variabilidad en la demanda, sumado a la ocurrencia de fenómenos climáticos extremos, agregan mayor incertidumbre al proceso productivo (Ran et al., 2013).

1.2. Intensificación sostenible de los sistemas de producción

La intensificación sostenible (IS) de los sistemas de producción se define como un enfoque que busca el incremento de los rendimientos sin comprometer la salud de los ecosistemas o aumentar el área agrícola (Campanhola y Pandey, 2019); es decir, producir más sin incorporar nuevas tierras, conservar los recursos utilizados y reducir impactos negativos sobre el ambiente (Ajibade et al., 2023). La IS se considera un componente clave para lograr la seguridad alimentaria y la reducción de los impactos ambientales en la agricultura (Semmartin et al., 2023), enfocándose en un uso más eficiente de los recursos, uso de insumos alternativos o rediseño de los sistemas de producción (Haughey et al., 2023).

A través de la IS se busca alcanzar el agregado de valor ambiental a los productos de la región, reducir la dependencia actual de la agricultura de insumos externos y recursos no renovables, aumentar la eficiencia productiva a fin de reducir el impacto ambiental e incrementar la rentabilidad de la producción, lo cual genera condiciones favorables de vida en el campo, y propender a la resiliencia y adaptabilidad de los sistemas agropecuarios frente a los cambios climáticos globales y de otra índole (Arístide et al., 2020).

Se han propuesto como formas de intensificación sostenible la agricultura orgánica, la intensificación ecológica, la agricultura de conservación, así como también los sistemas integrados agrícola-ganaderos (Cortner et al., 2019). En Uruguay, los sistemas mixtos que combinan agricultura con ganadería a través de la rotación de cultivos con pasturas son ampliamente utilizados, lo que se asocia con la utilización de siembra directa (García-Préchac et al., 2004) y con las limitaciones impuestas en los niveles de tolerancia de pérdida de suelos en sistemas agrícolas (MGAP, 2020).

Diversos autores reportan las ventajas de la rotación cultivo-pasturas en la evolución del contenido de carbono en el suelo (Terra et al., 2006) en los rendimientos de cultivo posteriores a la fase de pastura (Diaz Zorita et al., 2002), en la reducción en la incidencia de pestes, enfermedades y malezas

(Martin et al., 2020), en el contenido de biomasa microbiana en el suelo (De Faccio Carvalho et al., 2010), en la eficiencia del uso de nutrientes (Ward et al., 2016; Denardin et al., 2020), en la diversificación de ingresos (Peyraud et al., 2014), así como también en la productividad global del sistema, dado el potencial de producción de carne que estos sistemas tienen sobre pasturas de alta calidad (De Faccio Carvalho et al., 2021).

Teniendo en cuenta el contexto actual de demanda por productos de origen animal y vegetal, en mayor cantidad y calidad (Flachowsky et al., 2017), y la presión que esto genera en los sistemas. Asociado al interés por conocer cómo se llevan adelante los procesos de producción (Xue et al., 2010), sumado a las preocupaciones por los impactos ambientales derivados de la producción de alimentos (Thornton, 2010), resulta crítico analizar la sostenibilidad de los sistemas agrícola ganaderos. Al mismo tiempo, dada la complejidad que estos sistemas presentan en términos de uso de la tierra, uso de insumos y productos obtenidos, resulta relevante contar con nuevos enfoques metodológicos, así como también desarrollar métricas o indicadores que permitan evaluar la sostenibilidad de las diferentes estrategias de intensificación (Paruelo y Sierra, 2023). En este sentido, Garrett et al. (2017) reportó que existe escaso conocimiento sobre comportamiento animal, compensaciones (*trade-offs*) entre indicadores de biodiversidad, emisiones de gases de efecto invernadero (GEI), control de pestes y resultado económico en sistemas integrados agrícola-ganaderos.

Por lo tanto, resulta fundamental analizar de manera multidimensional los sistemas de producción, evitando el foco en una única dimensión. De acuerdo con Lee et al. (2021), un enfoque que involucre el componente productivo, económico, social y ambiental de la sostenibilidad sería más beneficioso que un enfoque exclusivamente ambiental, donde los resultados obtenidos pueden llevar al desarrollo de políticas o análisis inadecuados.

1.3. Indicadores asociados a la evaluación de la sostenibilidad de los sistemas de producción

Arístide et al. (2020) propone una clasificación de las variables claves para la evaluación de la sustentabilidad de los sistemas agropecuarios en las dimensiones ambiental (estado del ambiente e impactos), productiva, social, económica y bienestar humano. Además, sugiere que al momento de seleccionar las variables deben considerarse las compensaciones que puedan existir.

La selección de indicadores adecuados para analizar la sostenibilidad de los sistemas de producción resulta clave, aunque no es una tarea sencilla dado las discrepancias que pueden existir en torno a la representatividad de los indicadores seleccionados, así como también la metodología de cálculo empleada (Mahon et al., 2017). Sin embargo, estos proveen información y permiten tomar decisiones, siendo un reflejo de la presión ejercida sobre los sistemas (Cazzulli y Paruelo, 2023). Conocer los valores para realizar comparaciones entre sistemas puede ser de gran utilidad, aunque también resulta relevante conocer las trayectorias de estos indicadores a lo largo del tiempo. En este sentido, es necesario considerar diferentes indicadores, individuales o agrupados en un índice, dado que la sostenibilidad de un sistema involucra diversas dimensiones o áreas de estudio en su evaluación.

1.4. Experimentos de largo plazo como plataformas de evaluación de la sostenibilidad de los sistemas de producción

Los experimentos de largo plazo (ELP) son cruciales para entender la sustentabilidad de los sistemas de producción, especialmente en un contexto de cambio climático y modificaciones en el uso del suelo (Sanderson et al., 2016), ya que actúan como réplicas de los sistemas comerciales de producción (Scott et al., 2013). Al mismo tiempo, dada la escala espacial y temporal, funcionan como base para la comprensión de procesos complejos, que no es posible entender en experimentos convencionales (Sayre et al., 2012). Los ELP, además, proveen información relevante para el desarrollo de políticas públicas (Poulton, 1996). Sin embargo, este tipo de experimentos presenta desafíos en su funcionamiento, tales como necesidad de superficies amplias, recursos económicos suficientes, equipos multidisciplinarios para abordar los diferentes componentes, así como también dificultad para realizar análisis estadísticos convencionales (Scott et al., 2013).

En Uruguay existen ELP ubicados en diferentes regiones que intentan responder diferentes preguntas de investigación (Terra, 2017), abarcando los diferentes sistemas de producción presentes en el país. Tal es el caso de INIA La Estanzuela (iniciado en 1963), EEMAC (1993) o INIA Palo a Pique (1995), donde se evalúan diferentes rotaciones de cultivos y pasturas; Paso de la Laguna (INIA Treinta y Tres, desde 2012), donde se evalúan diferentes rotaciones arroceras; INIA Las Brujas (2012), que aborda la producción hortícola, y recientes trabajos en INIA Tacuarembó e INIA Treinta y Tres con foco en manejo y productividad de campo natural.

1.5. Objetivos

1.4.1. Objetivo general

El objetivo del presente trabajo fue evaluar de forma multidimensional diferentes alternativas de intensificación de sistemas agrícolas ganaderos basadas en rotaciones y estrategias ganaderas contrastantes para el período que comprende desde mayo 2019 a abril 2022, tomando como base para el análisis la información obtenida en el ELP Palo a Pique, de INIA Treinta y Tres.

1.5.2. Objetivos específicos

• Evaluar la productividad en términos físicos de cada uno de los sistemas de rotaciones cultivo pastura: «Management and Productivity of Key Integrated Crop–Livestock Systems in Uruguay: The Palo a Pique Long-Term Experiment's Third Phase» (Pereyra Goday, F.; Rovira, P.; Ayala, W. y Rivero, M. J.: https:// doi.org/10.3390/agronomy12123023).

• Aplicar la metodología de análisis de ciclo de vida con foco en la huella de carbono para analizar el impacto ambiental de cada uno de los sistemas: «Carbon footprint of mixed farming crop-livestock rotational-based grazing beef systems using long term experimental data» (Pereyra Goday, F.; Jebari, A.; Takahashi, T.; Rovira, P.; Ayala, W.; Lee, M. R. F.; Rivero, M. J. y McAuliffe, G. A.: https://doi.org/10.1007/s13593-024-00977-1).

• Evaluar la eficiencia de uso y balance de nitrógeno de cada uno de los sistemas: «Nitrogen use efficiency of integrated crop-livestock systems at different levels of intensification» (Pereyra Goday, F.; Castillo, J.; Rovira, P.; Ayala, W.; Lee, M. R. F. y Rivero, M. J.; a submitir a *Frontiers in Sustainable Food Systems*).

• Evaluar el resultado económico de cada uno de los sistemas de rotaciones cultivo-pastura.

• Evaluar la contribución a la seguridad alimentaria a través de la producción de proteína consumible humana y energía consumible humana.

1.6. Descripción del experimento

El experimento se ubica en la Unidad Experimental Palo a Pique de INIA Treinta y Tres (33º 16' S, 54° 29' O) y fue instalado en 1995. Los suelos dominantes en el área experimental pertenecen a la Unidad Alférez y consisten en argisoles subéutricos melánicos abrúpticos y planosoles subéutricos melánicos/ócricos clasificados como de clase III por su capacidad de uso y manejo. Estos suelos se clasifican respectivamente como argiudol oxiácuico vértico y argiacuol de acuerdo con USDA-NRCS Soil Taxonomy (United States Department of Agriculture-National Resources Conservation Services, 1996). Presentan baja o moderada fertilidad con 1,5 - 2% de contenido de carbono orgánico en suelo, medido en 20 cm de profundidad (Terra y Préchac, 2002). Las precipitaciones promedio (1995-2019) en el sitio experimental fueron de 1379 ± 58 mm por año. La temperatura del aire para el mismo período fue de 23 ± 0,1°C y 11,3 ± 0,6 C (máxima promedio y mínima promedio, respectivamente).

Actualmente se encuentran funcionando cuatro sistemas agrícola-ganaderos, basados en rotaciones cultivo-pasturas, bajo siembra directa. Cada sistema presenta una estrategia ganadera asociada, así como también un área de campo natural complementaria. La figura 1 presenta una breve descripción de cada sistema.

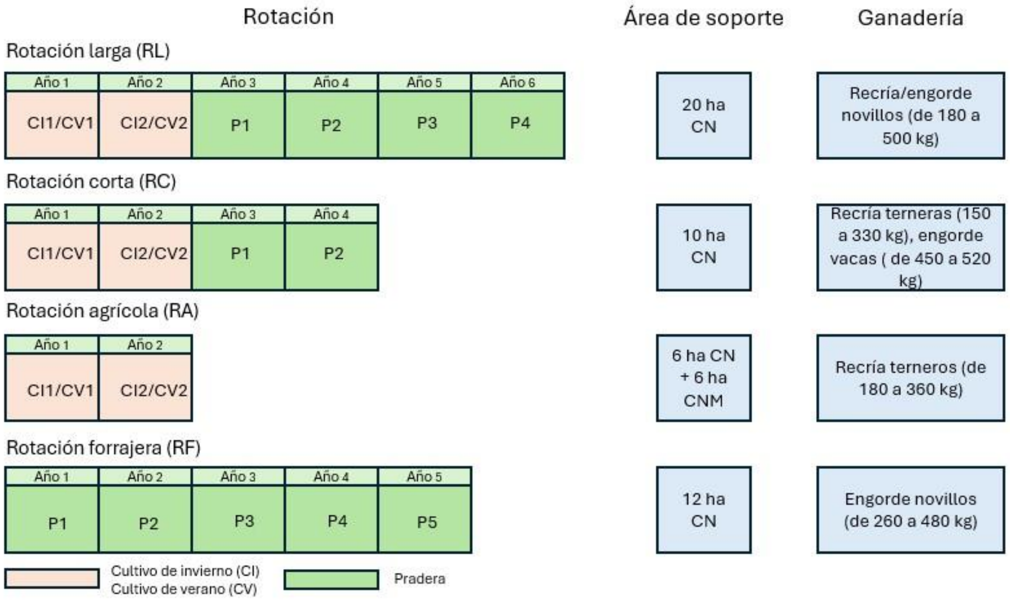


Figura 1

Rotaciones y estrategias ganaderas asociadas. Cada rectángulo equivale a 6 ha, que se divide a la mitad en un área para pastoreo y un área para producción de grano.

En la medida que el sistema se vuelve menos agrícola, el porcentaje de pasturas aumenta, el número de cultivos disminuye y los sistemas desarrollan el engorde de animales en mayor proporción. Los animales ingresan al experimento en el mes de mayo. En el caso de animales de recría, permanecen un año, mientras que, los animales en terminación (novillos y vacas) permanecen en el experimento hasta que alcanzan el peso de faena. Los pastoreos se realizan dentro de cada sistema, utilizando pasturas sembradas o campo natural. En los períodos en que es necesario se realiza suplementación con fardos y grano de sorgo (procedente del mismo sistema donde se suministra) o raciones comerciales.

El campo natural se encuentra presente en todas las rotaciones como área de soporte a las áreas mejoradas. Se utiliza de manera estratégica en períodos de exceso hídrico con la finalidad de no deteriorar las pasturas sembradas o en momentos críticos del crecimiento de estas (período de siembra e implantación). En la fase agrícola de los sistemas, se alternan cultivos para cosecha de granos: avena (*Avena byzantina*), sorgo (*Sorghum bicolor*), trigo (*Triticum aestivum*) y soja (*Glycine max*). La diferencia entre los sistemas está dada por la duración de la fase de pastura. En todos los sistemas se utilizan verdeos tanto estivales como invernales: raigrás (*Lolium multiflorum*), avena (*Avena byzantina*), sorgo (*Sorghum bicolor*), tef (*Eragrostis tef*), moha (*Setaria italica*) y sudangras (*Sorghum spp.*). En la rotación larga (RL) se incluye pradera con base en festuca (*Festuca arundinacea*), trébol blanco (*Trifolium repens*) y lotus (*Lotus corniculatus*), con una duración de cuatro años. En la rotación corta (RC) las praderas son con base en trébol rojo (*Trifolium pratense*) y holcus (*Holcus lanatus*), con una duración de dos años, y en la rotación agrícola (cultivo continuo, RA) se incluye un área de mejoramiento permanente compuesta por festuca, trébol blanco y lotus, que queda por fuera de la rotación. En la rotación forrajera (RF) se utiliza exclusivamente festuca en siembra pura.

• Obtención de la información

La información se está colectando desde el mes de mayo de 2019, a partir de muestreos de campo y registros realizados por funcionarios de INIA Treinta y Tres, así como también por parte de estudiantes de grado y posgrado que desarrollan sus tesis en el experimento.

• Difusión de los resultados obtenidos

Demás de las publicaciones en revistas arbitradas, en junio de 2023 se presentó un póster en el Long Term Experiment Congress realizado en Rothamsted Research, Harpenden, Reino Unido (20-22 de junio de 2023). El título del trabajo fue «Nitrogen use efficiency of integrated crop-livestock systems in the ‘Palo a Pique’ long-term experiment».

El 28 de julio de 2023 en INIA Treinta y Tres se desarrolló un seminario técnico denominado Claves para la Intensificación Sostenible de la Ganadería y la Agricultura en la Región Este, donde se presentaron resultados de investigación obtenidos en el experimento de Palo a Pique. En esta instancia, el título de la presentación fue «Intensificación sostenible de sistemas agrícola-ganaderos: indicadores de desempeño ambiental y eficiencia». Adicionalmente, se contribuyó con información para otras presentaciones para dicho seminario. También se participó en diversas oportunidades en presentaciones durante visitas a la unidad experimental y recorridas al experimento.

1.7. Otras actividades realizadas como parte del programa de doctorado

Como parte fundamental del proceso de formación, el programa de doctorado exige la realización de una pasantía fuera del núcleo académico donde se lleva adelante el trabajo de investigación. Para cumplir con este objetivo, durante 2022 (desde el 28 de junio al 1 de diciembre), financiado por una beca de movilidad de la Agencia Nacional de Investigación e Innovación (ANII), código MOV_CA_2021_1_171482, se llevó adelante la pasantía en Rothamsted Research (North Wyke, Devon, Reino Unido). El objetivo de dicha actividad fue adquirir experiencia y avanzar en el conocimiento de la metodología análisis de ciclo de vida y su aplicación en sistemas mixtos (agrícola-ganaderos); el referente técnico en esta actividad fue el Dr. Graham McAuliffe de Rothamsted Research.

A continuación se detallan otros proyectos o publicaciones en los que se participó.

• Contribución a la publicación «Taking the steps toward sustainable livestock: our multidisciplinary global farm platform journey» (Rivero, M. J., Evans, A. C. O., Berndt, A., Cartmill, A., Dowsey, A., Farruggia, A., Mignolet, C., Enriquez-Hidalgo, D., Chadwick, D., McCracken, D. I., Busch, D., Pereyra, F., Martin, G. B., Sanford, G. R., Sheridan, H., Wright, I., Brunet, L., Eisler, M. C., Lopez-Villalobos, N., … Lee, M. R. F. (2021). Taking the steps toward sustainable livestock: our multidisciplinary global farm platform journey. *Animal Frontiers, 11*(5), 52–58. https://doi.org/10.1093/af/vfab048) (Anexo 1).

• Contribución en el proyecto Systematic review on agricultural management practices to mitigate GHG emissions in UK, liderado por la Dra. Asma Jebari. Esta actividad consistió en revisión bibliográfica sobre las estrategias de mitigación a nivel de predio en Reino Unido y una posterior clasificación de acuerdo con las características de cada estrategia (subsector involucrado, porcentaje de reducción de emisiones, gas involucrado, análisis de incertidumbre, etc.). A través de esta actividad se avanzó en el conocimiento sobre estrategias de mitigación y el impacto de cada una de ellas, así como también en el uso de software específico para este tipo de revisiones (Jebari, A., Pereyra-Goday, F., Kumar, A., Collins, A. L., Rivero, M. J., McAuliffe, G. A. (2024). Feasibility of mitigation measures for agricultural greenhouse gas emissions in the UK. A systematic review. *Agronomy for Sustainable Development, 44*(1), 2. https://doi.org/10.1007/s13593-023-00938-0) (Anexo 2).

• Contribución en el proyecto *North Wyke Farm Platform Beef Life Cycle Assessment focussing on mitigation interventions*, liderado por la Dra. Asma Jebari. Las actividades realizadas en el marco de este proyecto consistieron en la organización de la información primaria obtenida en North Wyke Farm Platform, así como también el cálculo de emisiones (de animales y suelo), que constituyen una parte importante de las emisiones en predios de producción de carne. Esta actividad permitió la familiarización con los cálculos de emisiones, guías a utilizar, así como también información requerida.

• Contribución en el proyecto *Life Cycle Assessment methodological development synthesis over six years*, liderado por el Dr. Graham McAuliffe. En este proyecto, el aporte consistió en una contribución a un *synthesis paper* explicando el experimento de largo plazo de Palo a Pique, así como también los desafíos de este tipo de experimentos. (McAuliffe, G. A., Takahashi, T., Lee, M. R. F., Jebari, A., Cardenas, L., Kumar, A., Pereyra-Goday, F., Scalabrino, H., Collins, A. L. (2023). A commentary on key methodological developments related to nutritional life cycle assessment generated throughout a 6-year strategic scientific programme. *Food and Energy Security, 12*(4). https://doi.org/10.1002/fes3.480) (Anexo 3).

• Contribución al proyecto *AgZero+: Towards sustainable, climate-neutral farming*, liderado por Dr. Jonathan Storkey (Rothamsted Research) y Richard Pywell (CEH). El trabajo consistió en la toma de muestras a campo en diferentes momentos pre y post instalación de pasturas, para estimar las emisiones de gases de efecto invernadero desde el suelo utilizando cámaras de aislación en Rowden Farm (North Wyke); en este trabajo participaron también la Dra. Jordana Rivero y la Dra. Laura Cardenas. Esta actividad permitió el involucramiento directo en la toma de muestras a campo usando una técnica que no había utilizado antes (por lo tanto, requirió un entrenamiento previo) así como también conocer el funcionamiento y manejo de los sistemas de producción y las líneas de investigación que se están llevando adelante.

2. Management and Productivity of Key Integrated Crop–Livestock Systems in Uruguay: The Palo a Pique Long-Term Experiment's Third Phase

Fabiana Pereyra Goday [1,*,1], Pablo Rovira [1], Walter Ayala [1] and M. Jordana Rivero [2,*]

[1]Instituto Nacional de Investigación Agropecuaria (INIA), Treinta y Tres 33000, Uruguay

[2] Net Zero and Resilient Farming, Rothamsted Research, North Wyke, Okehampton EX20 2SB, UK

[*] Correspondence: fpereyra@inia.org.uy (F.P.-G.); jordana.rivero-viera@rothamsted.ac.uk (M.J.R.)

2.1. Resumen

Los Sistemas Integrados de Cultivo-Ganadería utilizan la diversificación productiva como una estrategia para mejorar la productividad y la eficiencia en el uso de la tierra. Las Rotaciones de Cultivo-Pastura son parte de estos sistemas e implican una fase de pastura incluida en la secuencia de cultivos. Las principales razones para incluir pasturas en los sistemas de cultivo son la baja productividad de los pastizales naturales y el aumento del rendimiento de los cultivos después de una fase de pastura. El objetivo fue analizar los indicadores de productividad y el manejo de cuatro sistemas que combinan la producción de cultivos y ganadería, con datos recolectados durante un período de 3 años (2019-2022). El sitio experimental fue el experimento de largo plazo ubicado en Palo a Pique (Treinta y Tres, Uruguay), instalado en 1995, ubicado en la zona climática subtropical y en suelos Oxyaquic Argiudolls (pendiente promedio de 3%). Los sistemas evaluados fueron CC (cultivo continuo), SR (dos años igual a CC, dos años de pasturas), LR (dos años igual a CC, cuatro años de pasturas) y FR (pastura continua con Festuca). Se evaluó la producción de peso vivo (PV), la producción de grano y la producción

[1] Facultad de Agronomía, Universidad de la República.

de materia seca (MS). La producción de peso vivo fue mayor en CC y SR (426 y 418 kg PV/ha) que en LR (369 kg PV/ha) y FR (310 kg PV/ha). La producción de materia seca fue mayor en FR y SR (6867 y 5763 kg MS/ha/año) respecto a LR (5399 kg MS/ha/año) y CC (5206 kg MS/ha/año). La producción de grano fue 10%, 16% y 9% menor en soja, trigo y sorgo en CC.

2.2. Summary

Integrated Crop Livestock Systems (ICLSs) use productive diversification as a strategy to improve productivity and land use efficiency. Pasture Crop Rotations are a part of ICLSs and imply a pasture phase included in the sequence of crops. The main reasons to include pastures in crop systems are low productivity of natural grasslands and increased crop yield after a pasture phase. Our objective was to analyze the productivity indicators and management of four ICLSs that combine crop and livestock production, with data collected over a 3 y period (2019–2022). The experimental site was The Palo a Pique (Treinta y Tres, Uruguay) long-term experiment installed in 1995, located in the subtropical climate zone and on Oxyaquic Argiudolls soils (3% average slope). Systems evaluated were CC (continuous cropping), SR (two years idem CC, two years of pastures), LR (two years idem CC, four years of pastures) and FR (continuous pasture with Tall Fescue). Liveweight (LW) production, grain production and dry matter (DM) production were evaluated. Liveweight production was higher in CC and SR (426 and 418 kg LW/ha) than in LR (369 kg LW/ha) and FR (310 kg LW/ha). DM production was higher in FR and SR (6867 and 5763 kg DM/ha/year) than in LR (5399 kg DM/ha/year) and CC (5206 kg DM/ha/year). Grain production was 10%, 16% and 9% lower in soybean, wheat and sorghum in CC.

Keywords: grazing-livestock systems; pasture crop rotations; meat production

2.3. Introduction

An important challenge in most food production systems is coping with the growing demand for livestock and agriculture products whilst, at the same time, ensuring environmental sustainability. Global food consumption is projected to increase 1.4% per year in the next decade, explained by demand recovery post 'COVID 19' pandemic, which represents an opportunity for producers. However, price fluctuations and contingent issues (e.g., war conflicts) affect food supply and add uncertainty [1].

Integrated Crop–Livestock Systems (ICLSs) use productive diversification as a strategy to cope with price fluctuations [2,3], improve land use efficiency [4], improve livestock and agriculture productivity [5] and are an interesting alternative to promote resilience and support the sustainable intensification of agriculture [6]. These systems are present in Australia [7], North and South America [8] and Europe [9]. In Uruguay, ICLSs occupy 13% of the total area used by livestock and they have gained relevance since the prevailing regulations on crop rotations set an upper limit to soil losses [10]. Meat production exports represent approximately 23% of the annual exports, whereas grain exports represent approximately 22%. The main grains exported are soybean, rice and wheat [11].

Pasture Crop Rotations (PaCrR) are a fundamental part of ICLSs and imply a rotation with perennial or annual pasture that are included in the sequence of crops. The main reasons to include pastures in crop systems are the low productivity of natural grasslands and increased crop yield after a pasture period [2]. These rotations with pastures have been shown to contain higher soil organic matter level, which is related to improving water infiltration, water quality, nutrient cycling and helps to mitigate greenhouse gas (GHG) emissions [8], when compared to lands that have continuous cropping. Rotations with pastures of 2 or 4 years of duration contain 5% more soil organic carbon (SOC) than continuous cropping [12]. Also, pastures contribute to improving grain productivity, reducing soil erosion and degradation [13], as well as reducing input demand [14].

In addition, including legumes in pastures has a positive effect on the nutrient supply into the soil, through biological fixation of nitrogen; approximately 30 kg of nitrogen is fixed by ton of dry matter (DM) of legumes produced above ground [15]. This, in turn, allows one to reduce fertilization costs [16]. Additionally, forage legumes improve the quality of the diet offered to livestock and this allows one to enhance animal performance [17], reducing GHG emissions per head [18,19]. Hence, livestock plays an important role in ICLSs since they can transform forages and crop residues from PaCrR into high-quality protein for human food [20–22] and diversify incomes in the systems [9]. Moreover, manure contributes to improving carbon (C) sequestration and soil fertility due to its high nutrient content [23,24]. Livestock's role aligns with the concept of circular economy, which provides an approach to explain how the complementarity between agriculture and livestock enables a reduction in the use of external inputs and improves the outputs in the systems [25].

Investigation about ICLS systems is complex to develop due to the need of substantial areas of land for experimental research, the economic resources involved, the decision-making challenges and labor required [26]. However, the development of long-term experiments (LTEs) can help to understand sustainability of ICLS systems, as well as their function as replicas of actual production systems [27]. Hence, LTEs provide important data about complex processes that could be confounded in small-scale experiments [28]. Further, LTEs allow one to evaluate the impacts of agronomic practices (e.g., fertilization, weed control, grazing) on natural resources with a long-term view [29] and obtain information for farmers or policy makers [30].

Therefore, the aim of this work was to analyze the productivity indicators of four ICLSs that combine crop and livestock production, with different intensities of soil use, with data collected over a 3 y period (May 2019 to April

2022). The underlying hypothesis behind those four ICLSs is that they can produce 400 kg liveweight (LW)/ha per year, with varying space and temporal patterns.

Measurements and indicators calculated in this work refer to the third phase of Palo a Pique Long-Term Experiment (Land Expansion and Livestock Intensification), which started in 2019, following a redesign, as described by Rovira et al. [31]. The main changes that occurred in this phase were: relocation of permanent pasture system, addition of grassland area as a support in each system and inclusion of a unique livestock strategy for each system.

2.4. Materials and Methods

2.4.1. Experimental Site

A long-term Pasture Crop Rotation (PaCrR) experiment under no-tillage was installed in 1995 at the 'Palo a Pique' Experimental Unit in Treinta y Tres (33◦16t S, 54◦29t W) belonging to the National Institute of Agricultural Research (INIA) in Uruguay. Uruguay is located in the subtropical climate zone. The annual mean (±SEM) accumulated rainfall in the experimental site for the last 28 years (1995–2022) was 1249 ± 72 mm per year distributed uniformly throughout the year. The mean maximum and minimum air temperatures for the same period were 23.0 ± 0.1 ◦C and 11.3 ± 0.6 ◦C, respectively. The research site has a 3% average slope and the loam soils are Oxyaquic Argiudolls according to USDA Soil Taxonomy [32] with moderate fertility [33] and a well-developed B_t horizon [34], with a soil depth of 51 cm.

2.4.2. Environmental Conditions during the Period May 2019–April 2022

Precipitation (P, mm), evapotranspiration (ETP, mm), relative humidity (RH, %) and dry bulb temperature (T, ◦C) measurements were obtained daily from the 'Palo a Pique' automatic meteorological station. A monthly soil water balance was calculated using P and ETP values, considering a soil water storage of 66 mm, according with Terra and Carámbula [35]. The temperature–humidity index (THI) was calculated based on the equation developed by

Thorn [36]. The cattle heat stress risk during summer was determined according to the Livestock Weather Safety Index [37] that established the following THI-based stress thresholds for cattle: normal ≤ 74; moderate 75–78; severe 79–83; very severe (emergency) ≥ 84.

2.4.3. Description of the Pasture–Crop Rotations

These PaCrR represent alternative pasture–crop arrangements with different temporal and spatial combinations in land use. The current design of PaCrR is detailed in Table 1. One rotation is based on continuous cropping (CC, 12 ha) which is represented by a rotation of 2 years with two crops per year (winter and summer). The crop area is divided into two halves within each paddock: one half corresponds to a forage-based crop rotation available for grazing, whereas the other half is an agricultural-based crop rotation destined to harvest grain or make hay. CC does not rotate with pastures but is complemented with an external area (6 ha) of a permanent improvement pasture (PI) composed of white clover (*Trifolium repens L.*), birdsfoot trefoil (*Lotus corniculatus L.*) and tall fescue (*Festuca arundinacea L.*) re-seeded every 5 years with the same species. Therefore, the crop and pasture areas are spatially separated in CC.

Table 1. Cropping and pasture sequences of the 4 pasture–crop rotations in the 'Palo a Pique' long-term experiment.

Rotation	Purpose of crop phase	Year of the Rotation					
		Year 1	Year 2	Year 3	Year 4	Year 5	Year 6
Continuous Cropping	Crop/hay	Oat/Sorghum	Black Oat/Soybean	Wheat/Sorghum			
	Grazing	Oat/Sorghum	Ryegrass/Moha				
Short Rotation	Crop/hay	Idem CC	Idem CC	Wheat + P1	P2	P3	P4
	Grazing	Idem CC	Idem CC	P1	P2	P3	P4
Long Rotation	Crop/hay	Idem CC and SR	Idem CC and SR	Wheat + P1	P2	P3	P4
	Grazing	Idem CC and SR	Idem CC and SR	P1	P2	P3	P4
Forage Rotation	Grazing	Fescue	Fescue	Fescue	Fescue	Fescue	Fescue

P: pasture, followed by pasture age (i.e., P2: second-year pasture). All pastures, including those following the grain/hay crop phase, were available for grazing.

The second PaCrR is a short rotation (SR, 24 ha) that alternates in the same land over 2 years of crops, identical to CC with 2 years of grass–legume pastures based on red clover (*Trifolium pratense L*.), associated with Yorkshire fog (*Holcus lanatus L*.) and/or Italian ryegrass (*Lolium multiflorum L*.). Similarly, the long rotation (LR, 36 ha) alternates in the same land over 2 years of crops identical to CC and SR with 4 years of grass– legume pastures, composed of white clover, birdsfoot trefoil and tall fescue. In the half corresponding to the agricultural-based crop rotation, the pasture is sown associated with wheat (*Triticum aestivum L*.) in SR and LR. The fourth PaCrR is forage rotation (FR, 24 ha) seeded with tall fescue that does not rotate with agricultural crops. Occasionally a 1-year cycle of a winter and summer forage crop can be planted as a strategy to reseed the tall fescue in paddocks with compromised number of plants due to proliferation of weeds, especially bermudagrass (*Cynodon dactylon*) and hairy-finger grass (*Digitaria sanguinalis*).

The experiment lacks synchronic replications, but all phases of the rotations are present each year represented by paddocks of 6 ha in CC, SR and LR. In FR, the 24 ha was divided into 5 paddocks of 4.8 ha each corresponding to fescue seeded in 2013 (4.8 ha), 2014 (9.6 ha) and 2020 (4.8 ha). Details of the key soil parameters for the different PaCrR at the beginning of the present period of evaluation are given in Table 2. Soil analyses were carried out in Soil, Plant and Water Laboratory of INIA La Estanzuela (Colonia, Uruguay) and pH was estimated according to Beretta et al. [38], %C was estimated according to Wright et al. [39], %N was estimated from combustion at 900◦ and detection of N2, through thermal conductivity according to Simmone et al. [40], P (ppm) was estimated according to Bray and Kurtz [41] and bases were estimated according to Jackson [42]. Differences in soil parameters can be attributed to carry over effects over time as CC, SR and LR started in 1995 and FR started in 2013. Each rotation has a support area of natural grasslands (NGs) to handle the animals, when necessary (i.e., during periods with low forage availability in PaCrR), keeping the animals independently within each system. The proportion of NG is 33%, 29%, 26% and 33% for CC, SR, LR and FR, respectively. The predominant species in NGs are *Paspalum notatum, Axonopus affinis, Cyperus spp., Coelorhachis selloana, Paspalum dilatatum, Stenotaphrum secundatum, Panicum milioides, Cynodon dactylon, Setaria geniculate* and *Axonopus argentinus*, according to Ayala [43].

Table 2. Soil properties (0–15 cm) in the 'Palo a Pique' pasture–crop rotations long-term experiment in Treinta y Tres, Uruguay (2019).

		Soil Parameter (Mean ± s.d.) [1]				
Rotation [2]	Paddocks	pH	C, %	N, %	P, ppm	Bases, meq/100 g
CC	2	5.38 ± 0.250	1.47 ± 0.243	0.15 ± 0.032	36.5 ± 17.96	6.41 ± 0.383
SR	4	5.17 ± 0.100	1.79 ± 0.114	0.18 ± 0.012	23.8 ± 6.32	7.45 ± 0.521
LR	6	5.35 ± 0.110	1.90 ± 0.182	0.18 ± 0.025	26.0 ± 3.79	8.16 ± 0.594
FR	4	5.27 ± 0.120	1.82 ± 0.055	0.18 ± 0.014	13.8 ± 2.23	5.42 ± 1.110

[1] C: organic carbon; N: nitrogen; P: phosphorus; Bases: Ca, Mg, K, Na. [2] CC: continuous cropping; SR: short rotation; LR: long rotation; FR: forage rotation.

2.4.4. Pasture and crop management

Winter crops and pastures were sown between March and June, and summer crops were planted in October and November. Winter crops for grain (oat and wheat) were usually harvested in December and summer crops (sorghum and soybean) in April. Cover crops (black oat) were harvested for hay in October. Weeds, pests and diseases were controlled according to standard agronomic recommendations. Levels of mineral N, P and K fertilizers are shown in Table 3. The fertilizers (N-P-K-S) used were 15-30-15-0, 9-25-25-0, 46-0-0-0 (urea) and 0-25-0-4. For legume-based pastures, fertilizer averaged 22.5 kg N/ha, 45 kg P/ha and 22.5 kg K/ha when the pasture was seeded, and a re-fertilization of 37.5 kg P/ha and 6 kg S/ha was applied every autumn during the pasture phase. The fescue-based pasture in FR was fertilized with 188 kg N/ha per year distributed in 46 kg N/ha per season and 37.5 kg P/ha in autumn.

Table 3. Mineral nitrogen, phosphorus, potassium and sulfur fertilizer inputs (kg/ha) per crop per year for grain and forage annual crops in the 'Palo a Pique' pasture–crop rotations long-term experiment in Treinta y Tres, Uruguay (2019/2020, 2020/2021 and 2021/2022).

	Grain Rotation					Forage Rotation			
Nutrient	Oat	Sorghum	Oat [1]	Soybean	Wheat [2]	Oat	Sorghum	Ryegrass	Foxtail Miller[3]
Nitrogen	41/69/53	73/69/11	41/35/41	14/14/23	86/93/92	87/75/86	87/87/80	77/72/87	109/115/82
Phosphorus	16/16/16	16/16/13	16/11/16	16/16/20	20/20/19	16/16/15	16/16/13	7/7/16	17/17/16
Potassium	15/15/15	15/19/25	15/10/15	31/31/38	19/19/18	15/16/14	15/15/25	6/7/15	32/32/31
Sulfur	3/6/5	0/0/4	0/0/0	5/5/0	8/9/6	-	-	-	-

[1] Black oat for hay, [2] Planted associated with a perennial pasture in the short and long rotation, [3] Replaced by sorghum in 2020/2021 and 2021/2022.

2.4.5. Matching pasture–crop rotations with different livestock strategies

A unique livestock strategy was established for each PaCrR in 2019. The livestock strategies had to be commercially available and adopted by producers (end up with an ani- mal category easy to sell) and be different from each other. Thus, 126, 133 and 141 6-month Aberdeen Angus calves were weaned in April 2019, 2020 and 2021, respectively, sorted by sex and LW and assigned to one of three PaCrR in May. Further, 32 (191 ± 16 kg LW), 34 (179 ± 17 kg LW) and 35 (200 ± 30 kg LW) male calves were allocated in CC in 2019, 2020 and 2021, respectively. The livestock strategy in CC (Figure 1a) focused on rearing calves for one year selling yearling steers ready to enter a feedlot (estimated final LW: 370 kg). Moreover, 44 (148 ± 17 kg), 49 (153 ± 16 kg) and 46 (167 ± 21 kg LW) female calves were allocated in SR in 2019, 2020 and 2021, respectively. The livestock strategy in SR (Figure 1b) is focused on rearing heifers for one year to produce replacement heifers for the breeding herd (estimated final LW: 330 kg). This system was complemented with 15 (2019) and 10 (2020 and 2021) finishing culled beef cows between May and September (estimated initial LW: 484 ± 72 kg, 446 ± 19 kg and 483 ± 24 kg, respectively). Fifty male calves were allocated to LR in 2019 (190 ± 14 kg), 2020 (185 ± 15 kg) and 2021 (199 ± 31 kg). The livestock strategy in LR (Figure 1c) has the objective of rearing and finishing steers over an 18-month period producing a finished steer ready for slaughter (estimated final LW: 530 kg). Unlike CC and SR, the cycle of production in LR lasts more than a year; therefore, the new generation of weaned calves and finishing steers (that entered as calves the previous year) concur during winter and spring. Finally, FR is the only system that begins by the end of the spring (November–December) with yearling steers instead of weaned calves. The objective of the livestock strategy in FR (Figure 1d) is to produce a finished steer ready for slaughter in 12–15 months. Thus, 47 (318 ± 28 kg), 30 (250 ± 12 kg), 35 (253 ± 32 kg) and 41 (263 ± 65 kg) Aberdeen Angus steers entered the system in May 2019, December 2019, November 2020 and November 2021, respectively.

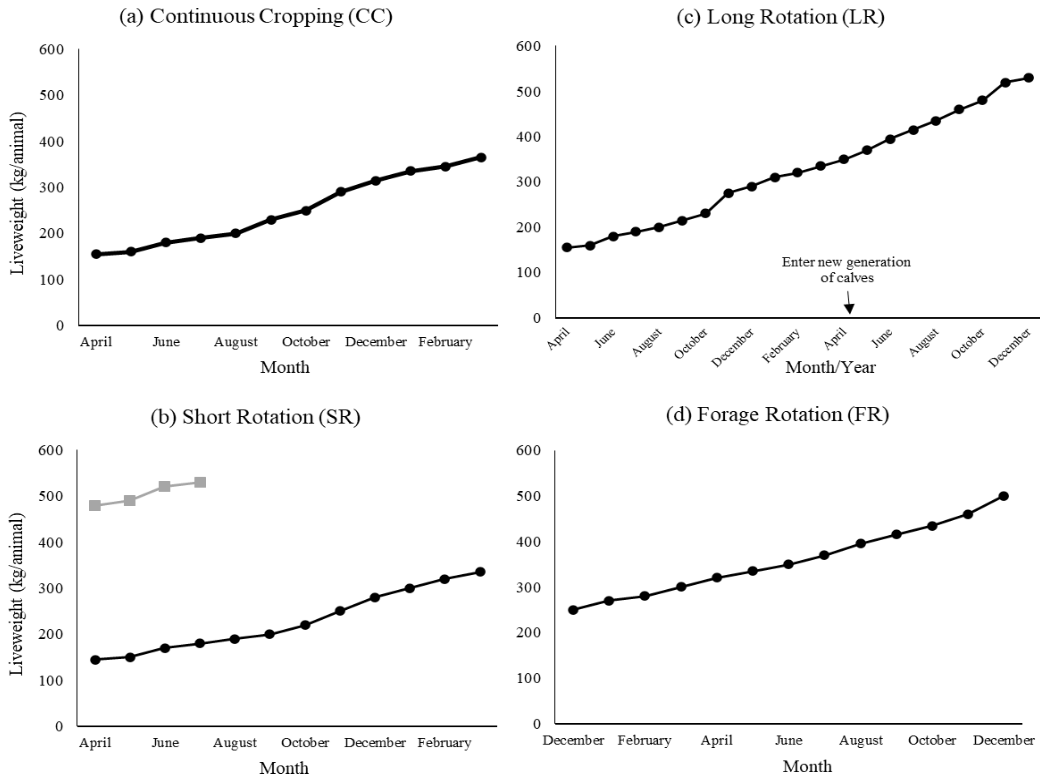


Figure 1. Estimated animal liveweight evolution in the different livestock strategies assigned to each pasture–crop rotation. (a) Continuous cropping (CC)–rearing calves; (b) Short Rotation (SR)–rearing heifers (black line) +

culled cows (grey line); (c) Long rotation (LR)–rearing and finishing steers; (d) Forage rotation (FR)–finishing steers.

2.4.6. Pasture and animal measurements

Three grazing exclusion cages (0.4 × 1.0 m) were used per grazing paddock (3–5 ha) to estimate daily pasture growth (kg DM/ha/d) every 30 days according to the methodology proposed by Lynch [44]. Forage inside the grazing exclusion cages was also used to assess the botanical composition of the pasture at each sampling date by quantifying the contribution (%, DM basis) at the species level (i.e., tall fescue, white clover, lotus, etc.). After collecting the forage cuts, cages were moved and placed in a new area where the pasture was representative of the overall paddock to start measuring a new 30 d cycle. Herbage mass stock was estimated once per month; 100 random points were measured with Rising Plate Meter (RPM) (FarmWorks, New Zealand) to obtain an average value for each paddock. A rectangle (0.2 × 0.5 m) was cut at ground level with a value of RPM similar to the paddock average. Crude protein (CP, %), metabolizable energy (ME, Mcal/kg DM) and neutral detergent fiber (NDF, %) analyses were conducted using standard methods [45] in the Animal Nutrition Laboratory of INIA La Estanzuela (Colonia, Uruguay), from herbage mass stock data. Assessment of pasture herbage mass (kg DM/ha) and height (cm) was carried out pre- and post-grazing by cutting six rectangles at ground level in each grazing paddock to estimate the amount of forage that disappears after each grazing period (% utilization). Nitrogen input (biological nitrogen fixation) was estimated from biomass of legumes aboveground, according to [15].

All animals were weighed every 30 days and individual performance was calculated as daily LW gain (kg/d). The stocking rate for each system (kg LW/ha) was calculated after each weighing of the animals. Liveweight gain per ha (kg LW/ha) was calculated by multiplying LW gain per animal by the number of animals per ha for each period. Feed to gain ratio (F/G) was estimated as the kg of DM (pasture + supplement) required to achieve 1 kg of LW. The amount of supplement fed to animals grazing in each system was recorded (kg DM/ha) each year. Supplements included hay, high-moisture sorghum grain complemented with a protein ration (48% CP) and an energetic-protein ration (14% CP). In general, supplements were fed to cattle during winter to maintain growth rates of steers and calves and, occasionally, during summer associated with prolonged drought periods.

2.4.7. Data analysis

Three grazing exclusion cages (0.4 × 1.0 m) were used per grazing paddock (3–5 ha) to estimate daily pasture growth (kg DM/ha/d) every 30 days according to the methodology proposed by Lynch [44]. Forage inside the grazing exclusion cages was also used to assess the botanical composition of the pasture at each sampling date by quantifying the contribution (%, DM basis) at the species level (i.e., tall fescue, white clover, lotus, etc.). After collecting the forage cuts, cages were moved and placed in a new area where the pasture was representative of the overall paddock to start measuring a new 30 d cycle. Herbage mass stock was estimated once per month; 100 random points were measured with Rising Plate Meter (RPM) (FarmWorks, New Zealand) to obtain an average value for each paddock. A rectangle (0.2 × 0.5 m) was cut at ground level with a value of RPM similar to the paddock average. Crude protein (CP, %), metabolizable energy (ME, Mcal/kg DM) and neutral detergent fiber (NDF, %) analyses were conducted using standard methods [45] in the Animal Nutrition Laboratory of INIA La Estanzuela (Colonia, Uruguay), from herbage mass stock data. Assessment of pasture herbage mass (kg DM/ha) and height (cm) was carried out pre- and post-grazing by cutting six rectangles at ground level in each grazing paddock to estimate the amount of forage that disappears after each grazing period (% utilization). Nitrogen input (biological nitrogen fixation) was estimated from biomass of legumes aboveground, according to [15].

All animals were weighed every 30 days and individual performance was calculated as daily LW gain (kg/d). The stocking rate for each system (kg LW/ha) was calculated after each weighing of the animals. Liveweight gain per ha (kg LW/ha) was calculated by multiplying LW gain per animal by the number of animals per ha for each period. Feed to gain ratio (F/G) was estimated as the kg of DM (pasture + supplement) required to achieve 1 kg of LW. The amount of supplement fed to animals grazing in each system was recorded (kg DM/ha) each year. Supplements included hay, high-moisture sorghum grain complemented with a protein ration (48% CP) and an energetic-protein ration (14% CP). In general, supplements were fed to cattle during winter to maintain growth rates of steers and calves and, occasionally, during summer, associated with prolonged drought periods.

2.5. Results

2.5.1. Environmental conditions

Soil water balance (Figure 2) during the experimental period was characterized by a deficit between November and January (summer) in the three years. In Y1, the deficit was prolonged in time and covered the sowing period of pastures and crops in autumn (March and April). Soil water recharge occurred mainly in winter (June–September), when ETP was minimal, creating occasional muddy conditions in the grazing paddocks.

Figure 3 shows the monthly average of maximum and minimum temperatures (Ts). The maximum T was 41.4 ◦C and the minimum T was −5.1 ◦C, with a marked seasonal pattern. THI average was 62.1 ± 8.3. The maximum value was 81 and minimum was 41. During the experimental period, medium heat-stress conditions occurred on 6.1% of the days, whereas severe heat-stress conditions occurred on 0.8% of the days. These conditions were mainly in summer, where heat-stress conditions occurred on 16.2% of the days.

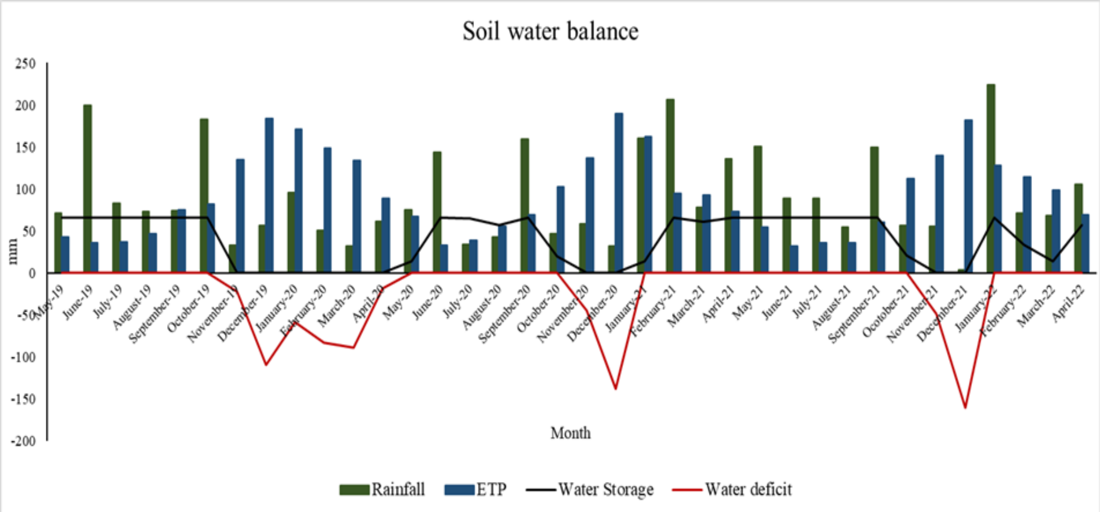


Figure 2. Soil water balance between May (2019) and April (2022) in the 'Palo a Pique' pasture–crop rotations long-term experiment in Treinta y Tres, Uruguay.

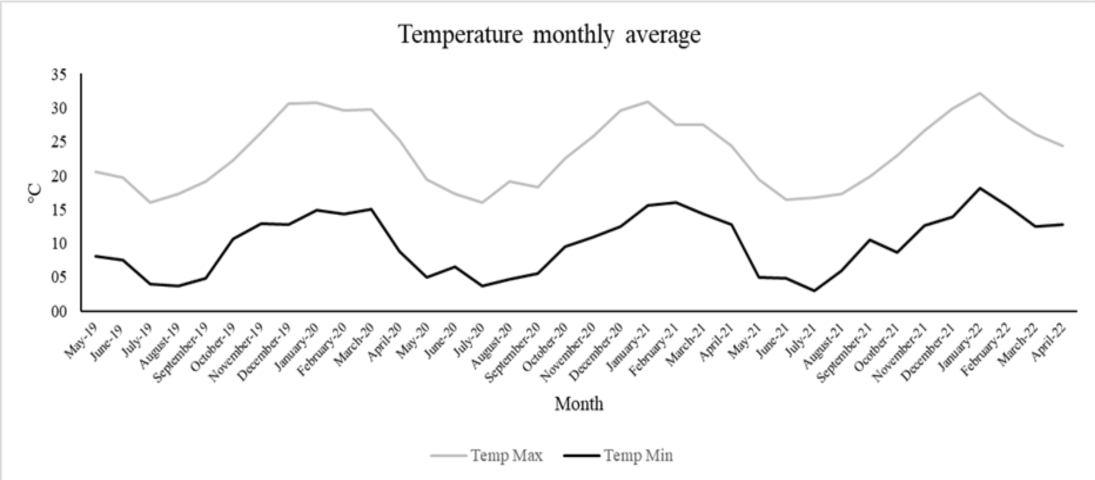


Figure 3. Evolution of monthly average maximum (T max) and minimum (T min) temperatures (◦C) between May 2019 and April 2022 in the 'Palo a Pique' pasture–crop rotations long-term experiment in Treinta y Tres, Uruguay.

2.5.2. <u>Crop production</u>

Table 4 shows grain yields for each crop in Year 1 (Y1, 2019–2020), Year 2 (Y2, 2020–2021) and Year 3 (Y3, 2021–2022) for the different rotations. Crop yield in CC was consistently lower than the yield obtained in crops rotating with perennial pastures (SR and LR). In Y1, grain yield reduction in CC was 36%

(wheat), 11% (sorghum) and 17% (soybean) compared with the yield average observed in SR and LR. The same tendency was obtained in Y2 (10%, 8% and 15% yield reduction in CC for wheat, sorghum and soybean, respectively). During Y3, soybean grain yield in CC was 3% higher than the average of LR and SR. Due to adverse climatic conditions, oat crops were harvested only in Y2 and Y3 for SR and LR and Sorghum crops were not harvested in Y3.

Table 4. Grain yield (t/ha) for crops in the ‘Palo a Pique’ pasture–crop rotations long-term experiment in Treinta y Tres, Uruguay (2019/2020, 2020/2021 and 2021/2022).

Rotation	Crops			
	Oat	Sorghum	Soybean	Wheat
	2019–2020 (Y1)			
Continuous cropping	-	4.12	2.28	0.76
Short Rotation	-	4.51	2.87	1.26
Long Rotation	-	4.77	2.60	1.16
	2020–2021 (Y2)			
Continuous cropping	-	5.81	2.29	3.66
Short Rotation	2.43	6.79	2.37	4.11
Long Rotation	2.16	5.82	3.02	4.02
	2021–2022 (Y3)			
Continuous cropping	-	-	2.52	-
Short Rotation	1.20	-	2.20	-
Long Rotation	2.20	-	2.66	-

Hay was produced in Y1, Y2 and Y3 from black oat paddocks (CC, SR and LR) and from one block of tall fescue (FR). Hay production (kg DM/ha) in Y1 was 50.1 in CC, 632.2 in SR, 90.1 in LR and 466.7 in FR, whereas in Y2, 916.7 was produced in CC, 417.8 in SR, 174,1 in LR and 458.3 in FR. During Y3, hay production was 516.7, 589.3, 276 and 441.6 in CC, SR, LR and FR, respectively.

2.5.3. Forage growth

Data are presented as an average of different paddocks for oat, Italian ryegrass, natural grassland, permanent improvement and tall fescue, whereas in the permanent pasture, data are presented as an average of different ages in LR and SR (Table 5).

Table 5. Forage growth for each type of pasture (average ± s.d.) and year in the 'Palo a Pique' pasture–crop rotations long-term experiment in Treinta y Tres, Uruguay.

	Year		
Pasture	2019–2020 (Y1)	2020–2021 (Y2)	2021–2022 (Y3)
Permanent Pasture (LR) [1]	19.5 ± 18.52	18.4 ± 11.71	22.4 ± 15.61
Permanent Pasture (SR) [2]	19.8 ± 13.01	15.4 ± 14.02	34.9 ± 24.74
Ryegrass	23.8 ± 2.51	30.2 ± 3.05	28.6 ± 13.83
Oat	14 ± 6.6	18.1 ± 15.50	12.6 ± 12.37
Tall Fescue	22.9 ± 15.47	20.2 ± 11.24	30.7 ± 20.95
Permanent Improvement	18.4 ± 11.11	13.8 ± 11.30	17.2 ± 14.47
Natural Grassland	13.7 ± 7.57	15.4 ± 12.29	16.9 ± 15.11

[1] Grass-legume pastures composed by white clover, birdsfoot trefoil, and tall fescue. [2] Grass-legume pastures based on red clover (Trifolium pratense L.) associated with Yorkshire fog (Holcus lanatus L.) and/or Italian ryegrass (Lolium multiflorum L.).

In Y1, the white-clover-based permanent pasture (PP) in LR grew (kg DM/ha/day) 36.9 ± 28.03, 17.6 ± 15.26, 16.4 ± 8.60 and 14.8 ± 14.40, for first-, second-, third- and fourth- year pasture, respectively, and average daily growth decreased with the age of the pasture. In red-clover-based PP, maximum values were recorded in October (54.1 kg DM/ha/day) and minimum were in December and January (0 kg DM/ha/day). In both annual pastures (oat and ryegrass), maximum growth was registered in July (29.1 and 28.2 kg DM/ha/day, respectively). Tall fescue seeded in FR registered a maximum growth in October (57.2 kg DM/ha/day) and minimum growth in December (2.20 kg DM/ha/day). NG and PI had a marked peak of production in spring–summer, with maximum values registered during October–November (23.4 kg DM/ha/day) and minimum in June–July (0 kg DM/ha/day).

Similar forage growth results were obtained in Y2. Maximum values in PP were obtained in February (40.7 and 36.8 to LR and SR) and minimum values were observed in July (11.2 and 11.3 kg DM/ha/day). Maximum values in Oat and Ryegrass were recorded in September and August. In Tall Fescue, maximum values were observed in March (39.5 kg DM/ha/day). Finally, maximum values in NG and PI were observed in February–March (43.8 and 25.9 kg DM/ha/day).

During Y3, maximum values in PP were observed in September (106 kg DM/ha/day) and minimum values were observed in summer (0 kg DM/ha/day). In Oat and Rye- grass, maximum values were observed in May (56.1 kg DMD/ha/day) and August (39.2 kg DM/ha/day). Maximum values in NG were recorded in February, after a dry period, and the minimum values were observed during winter months and November and January. Maximum values in PI were obtained during March (43.5 kg DM/ha/day) and minimum values during June. In three years, Y1, Y2 and Y3, forage growth of summer crops (sorghum and moha) was estimated using values from the bibliography. Reference values were 100 kg DM/ha/day and 70 kg DM/ha/day for sorghum and moha, respectively [47,48].

2.5.4. Forage production

Table 6 shows DM production for the four systems. Statistical differences were detected among systems. The highest productivity was observed in FR and SR, whereas the highest variability was observed in LR and SR. On the other hand, the lowest variability was observed in FR and CC (with the highest and the lowest DM production on average).

Table 6. Forage production (kg DM/ha/year) and coefficient of variation (CV%) in the 'Palo a Pique' pasture–crop rotations long-term experiment in Treinta y Tres, Uruguay.

	Forage Production [1]	
Rotation	kg DM/ha	CV (%)
Forage Rotation	6867 a	4.2
Short Rotation	5763 a	16.5
Long Rotation	5399 ab	14.2
Continuous Cropping	5206 b	4.4
p value	0.0394	-

[1] Annual DM production/ha (average of three years). Different letters in the same column mean significant differences.

Forage production was higher in spring compared with the rest of the seasons (Figure 4). The second forage production peak was registered in summer, associated with active growth of summer annual crops (sorghum and moha). Critical periods were observed in late spring (November–December) and early autumn (March–April), where systems had low DM production, associated with the presence of low-productive fallows, after glyphosate application, preparing the land for seeding first-year pastures (autumn) and annual crops (autumn and spring). The proportion of fallows within the PaCrR in- creases as the length of the pasture decreases, e.g., 100%, 75% and 50% of the area under the PaCrR corresponds to fallows in autumn for CC, SR and LR, respectively.

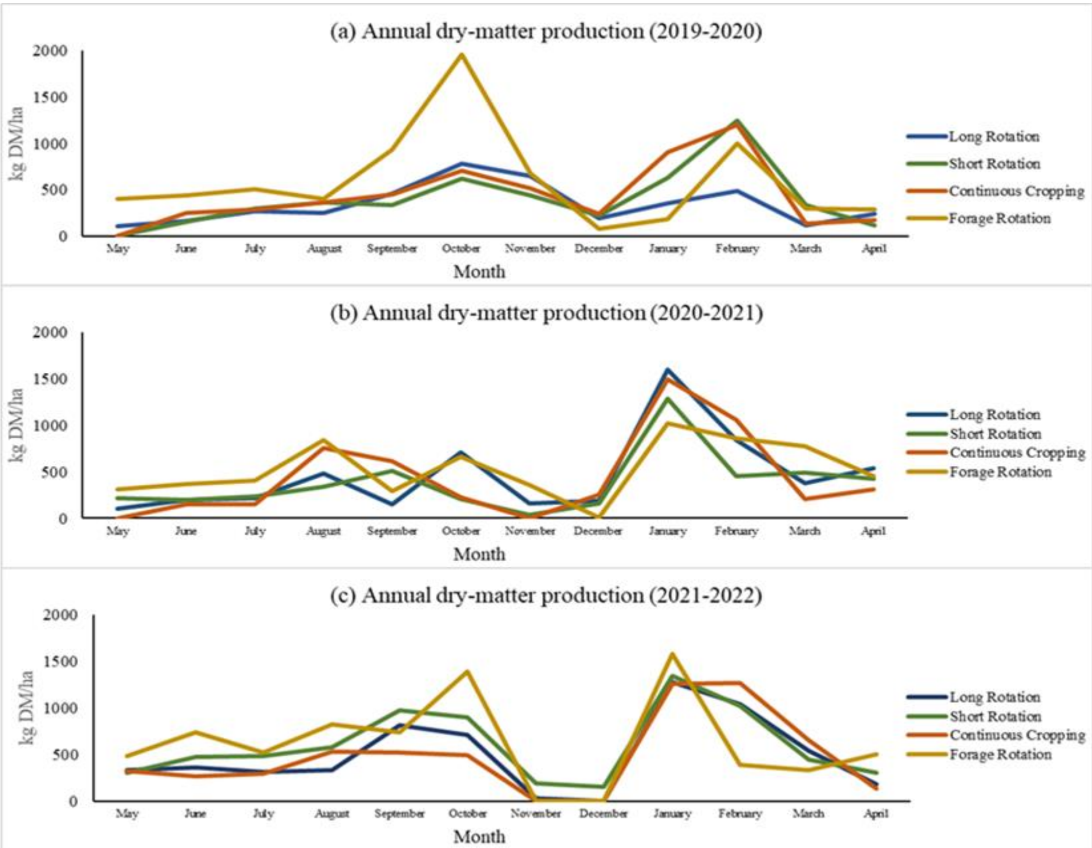


Figure 4. Distribution of dry-matter (DM) production (kg DM/ha) in the ‘Palo a Pique pasture–crops rotations long-term experiment in Treinta y Tres, Uruguay (a) Year 1 (2019–2020); (b) Year 2 (2020–2021) and (c) Year 3 (2021–2022).

Established perennial pastures produced forage throughout the year. Species com- position of perennial pastures determined the distribution of forage production. Pastures with white clover, tall fescue and birdsfoot trefoil in LR produced 26.3 ± 8.21% of the total annual DM production during winter, 34.8 ± 19.03% during spring, 23.4 ± 13.15% in summer and 15.5 ± 7.08% in autumn, averaging across Y1, Y2 and Y3. Short pastures in SR, comprising Yorkshire fog and red clover, had a more even distribution of forage production throughout the year compared with pastures in LR. They produced 23.9 ± 8.55%, 30.8 ± 14.43%, 28.1% ± 8.09% and 17.1 ± 10.33% of the total forage production in winter, spring, summer and autumn, respectively. Permanent improvement pasture in CC, which had a similar botanical composition to the pasture in LR, produced 16.2 ± 3.26% of the total DM production in winter, 41.5 ± 9.36% in spring, 21.1 ± 12.10% in summer and 21.2 ± 14.18% in autumn. Tall fescue in FR produced 38.6 ± 6.09% of the total annual DM production in winter, 34.3 ± 17.28% in spring, 19.8 ± 14.01% in summer and 7.3 ± 6.05% in autumn. Annual forage production of NG was 2947, 3811 and 3413 kg DM/ha for Y1, Y2 and Y3, respectively.

LR, SR and CC include pastures with legumes in a proportion of 48, 43 and 33% of the total area of the system, respectively. In LR, DM legume production was 39.5 ± 24.25%, 14.3 ± 14.01%, 17.2 ± 13.09% and 4.91 ± 2.079% of the total DM production for the 1st, 2nd, 3rd and 4th year of pasture, respectively. In SR, legumes contributed to 39.7 ± 28.22% and 21.5 ± 11.25% of the total DM (1st and 2nd year of pasture, respectively). The PI in CR had a legume contribution of 8.2 ± 5.35% of total DM production, averaging across Y1, Y2 and Y3. Data about forage quality are detailed in Supplementary Materials (Tables S1–S3).

Total nitrogen contribution to the soil is presented in Figure 5. Data are presented as an average across Y1 (2019–2020), Y2 (2020–2021) and Y3 (2021–2022) for each pasture, according to age of pasture (1st year, 2nd year, 3rd year and 4th year). There was a trend to decrease N fixed as pasture age increased in both pastures (LR and SR).

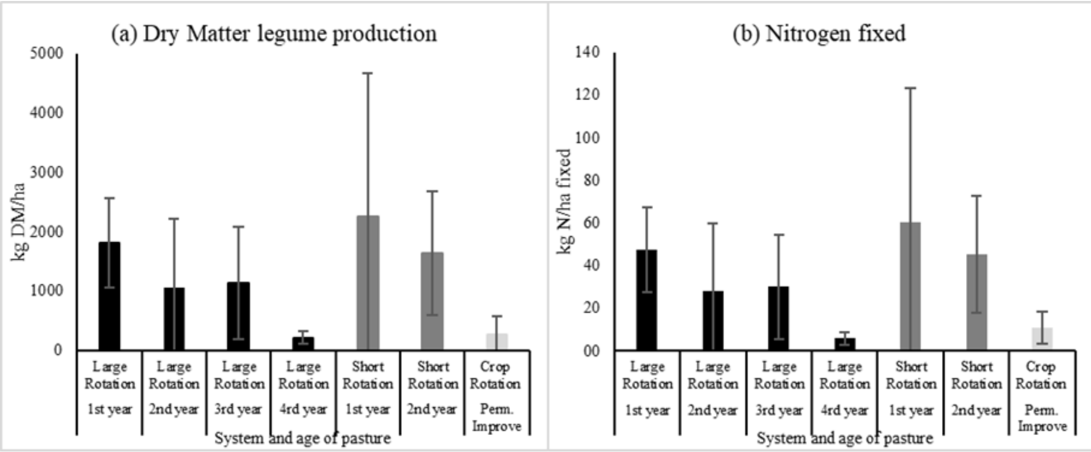


Figure 5. Dry-matter production of legumes (a) and Nitrogen fixed (b) by age of pasture in the 'Palo a Pique pasture–crops rotations long-term experiment in Treinta y Tres, Uruguay.

2.5.5. <u>Supplementation</u>

Animals from all the systems received strategic supplementation when the available forage was not enough to prevent LW losses from the animals. Levels of supplementation are presented as kg feed DM/ha (Table 7). In all years, supplementation was carried out during winter. In addition, summer supplementation was carried out in Y2 and Y3 associated with a prolonged dry period.

Table 7. Level (kg DM/ha) and type of supplementation in each Pasture Crop Rotation in 'Palo a Pique' long-term experiment in Treinta y Tres, Uruguay.

	Year 1			Year 2				Year 3			
	Winter			Winter			Summer	Winter			Summer
	Hay [1]	PC [2]	HSMG [3]	Hay [1]	PC [2]	HSMG [3]	BR [4]	Hay [1]	PC [2]	HSMG [3]	BR [4]
Rotation [5]											
CC	294	25.2	132	39.2	-	-	-	435	37.4	235	-
SR	1155	37.4	197	770	-	38.7	-	444	-	270	-
LR	996	53.7	282	1043	6.81	698	117	780	-	867	73.2
FR	-	-	-	414	-	68.7	499	-	-	-	-

[1] Hay: 6.7% crude protein (CP), metabolizable energy (ME) = 5.8 MJ/kg DM; [2] Protein concentrate (PC): 46.5% CP, ME = 10.5 MJ/kg DM; [3] High Moisture Sorghum Grain (HMSG): 8.1% CP, ME = 12.6

MJ/kg DM; [4] Balanced Ration (BR): 14% CP, ME = 11.7 MJ/kg DM. [5] CC: continuous cropping; SR: short rotation; LR: long rotation; FR: forage rotation.

2.5.6. Grazing management

Table 8 shows percentages of pasture occupation for each system. On average, pastures outside the area of the PaCrR were occupied by animals 45.1%, 40.2% and 40.7% of the time in Y1, Y2 and Y3, respectively. The combined use of NG and PI in CC had the maximum occupation rate (75.1% and 64.1% and 58.6%, respectively), whereas NG in FR had the minimum occupation rate (21.1%, 26.6% and 21.9%, respectively).

Table 8. Occupation of pastures (% of time per year) in the 'Palo a Pique' pasture–crop rotations long-term experiment in Treinta y Tres, Uruguay.

	Rotation			
Year/Pasture	CC	SR	LR	FR
2019-2020				
Annual Summer	4.40	10.1	5.60	-
Annual Winter	20.5	15.3	7.40	-
Perennial Pasture	-	40.3	36.6	78.9
Natural Grassland	75.1*	34.3	50.4	21.1
2020-2021				
Annual Summer	10.8	9.70	9.90	-
Annual Winter	25.1	19.5	9.60	-
Perennial Pasture	-	37.3	43.2	74.0
Natural Grassland	64.1*	33.5	37.3	26.
2021-2022				
Annual Summer	20.7	7.40	4.40	-
Annual Winter	20.7	23.2	18.6	-
Perennial Pasture	-	26.2	37.9	78.1
Natural Grassland	58.6*	43.2	39.1	21.9

[1] CC: continuous cropping; SR: short rotation; LR: long rotation; FR: forage rotation. * Includes permanent improvement pasture.

Within PaCrR, PP had an average occupation of 52.3%, 51.5% and 47.4% in Y1, Y2 and Y3, respectively. In both years, PP in FR had the highest occupation rate due to the absence of annual forage crops. LR and SR had similar occupation rates for PP. On average, each grazing period in PP lasted

6.2, 7.5 and 4.4 days in Y1, Y2 and Y3, respectively, whereas each grazing event in the annual forage crops lasted 4.3, 3.2 and 6.6 days in Y1, Y2 and Y3, respectively.

2.5.7. Animal performance

Table 9 shows seasonal average daily gain (ADG) for the different livestock categories. The highest and lowest individual ADG was observed in spring and winter, respectively. Animal categories closer to slaughter (finishing steers and cows) registered numerically higher ADG compared to rearing categories (calves). However, younger animals (<18 months old) registered a better efficiency (lower numeric values) than older animals. On average, growing categories (calves and heifers) required 46.2% and 25.9% less feed to gain 1 kg of LW than culled cows and finishing steers, respectively.

Table 9. Seasonal average daily gain (ADG, kg LW/d per animal) of cattle for the different livestock categories in the 'Palo a Pique' pasture–crop rotations long-term experiment in Treinta y Tres, Uruguay.

	Rotation [1]					
	CC	SR		LR		FR
ADG [2]	Calves	Heifers	Cows	Calves	Steers	Steers
Wi	0.34 ± 0.172^{d}	0.38 ± 0.107^{d}	0.59 ± 0.308	0.37 ± 0.352^{d}	0.45 ± 0.207^{c}	0.44 ± 0.270^{c}
Sp	0.87 ± 0.343^{a}	0.85 ± 0.139^{a}	-	0.73 ± 0.164^{a}	0.80 ± 0.235^{a}	1.12 ± 0.295^{a}
Su	0.75 ± 0.341^{b}	0.57 ± 0.203^{b}	-	0.55 ± 0.302^{b}	0.69 ± 0.379^{b}	0.49 ± 0.253^{bc}
Au	0.38 ± 0.419^{c}	0.52 ± 0.420^{c}	-	0.46 ± 0.553^{c}	-	0.55 ± 0.412^{b}
p value	<0.0001	<0.0001	-	<0.0001	<0.0001	<0.0001

[1] CC: continuous cropping; SR: short rotation; LR: long rotation; FR: forage rotation. [2] Au: autumn; Wi: winter; Sp: spring, Su; summer. Different letters in the same column mean significant differences.

Efficiency in each system was calculated from F/G ratio (Table 10), considering the proportion of kg of LW produced in each system according to each animal category. Al- though no significant differences were found, a tendency to obtain better efficiencies was observed in those systems with a higher proportion of rearing. Forage utilization varied between 50 and 60% in LR, 55 and 62% in SR, 48 and 52% in CC and 35 and 39% in FR.

Average animal stocking rate (±s.d.) during the 3 years was 614 ± 33 (CC), 600 ± 44 (SR), 575 ± 15 (LR) and 498 ± 23 (FR) kg LW/ha. The minimum and maximum stocking rates were registered in FR (Y2: 473 kg LW/ha) and CC (Y3: 648 kg LW/ha), respectively.

Table 10. Feed to Gain ratio (kg feed/kg LWP) and coefficient of variation (CV, %) in the 'Palo a Pique' pasture–crop rotations long-term experiment in Treinta y Tres, Uruguay.

	Feed to Gain Ratio [1]	
Rotation	Kg Feed/kg LW	CV (%)
Continuous Cropping	14.1	29.2
Short Rotation	15.1	17.9
Long Rotation	16.1	18.2
Forage Rotation	19.2	35.8
p value	n.s.	-

[1] Average of three years.

Overall, CC and SR were the systems with the highest LW production and lowest variability over the years (Table 11). CC and SR achieved the highest annual LW production in Y1 (404 and 393 kg LW/ha/year, respectively), Y2 (438 and 444 kg LW/ha/year, respectively) and Y3 (437 and 418 kg LW/ha/year). On the other hand, FR was the system with the lowest LW production in the three years (307, 344 and 280 kg LW/ha/year, Y1, Y2 and Y3, respectively), whereas LR achieved an intermediate level of production (316, 394 and 399 kg LW/ha/year, Y1, Y2 and Y3, respectively). In all systems, spring was the season with the highest contribution to the total LW production (35–48% in Y1, 38–46% in Y2 and 29–49% in Y3), while autumn had the lowest contribution (6–7% in Y1, 3–17% in Y2 and 8–22% in Y3).

Table 11. Liveweight (LW) production (kg LW/ha) and coefficient of variation (CV, %) in the 'Palo a Pique' pasture–crop rotations long-term experiment in Treinta y Tres, Uruguay.

Liveweight production[1]		
Rotation	Kg LW/ha	CV (%)
Continuous Cropping	426 a	4.5
Short Rotation	418 a	6.1
Long Rotation	369 b	12.6
Forage Rotation	310 c	10.4
p value	0.0034	-

[1] Average of three years. Different letters in the same column determine significant differences.

In CC and FR, 100% of the annual LW production per ha was obtained from rearing calves and finishing steers, respectively. Both stages of production were carried out in LR, contributing to 57% (rearing calves) and 43% (finishing steers) of the total annual LW production averaging over the years. In SR, rearing heifers was the main contributor to the total LW production (92%), followed by finishing cows (8%).

2.6. Discussion

Integrated Crop Livestock Systems allow one to improve food production while, at the same time, reducing negative environmental impacts and, therefore, are an option to achieve economic, sociological, ecological, energy, environmental and biogeochemical synergies and efficiencies [49]. The four systems evaluated in this work present different intensities of soil use and, at the same time, each system has a specific associated livestock strategy. The concept behind this rotation–livestock differential strategy association is that those systems that feature more intensive soil use, with more use of inputs (e.g., fertilizers, fuel, herbicides), are associated with more efficient livestock strategies (e.g., less feed to gain ratio, less GHG emissions), whereas systems with less intensity of soil use, including pasture phase in their rotation, are associated with less efficient livestock strategies evaluated (e.g., finishing animals), presenting trade-off to reduce negative impacts of agriculture or livestock production [31].

This work reports results of productivity and management from four ICLSs for three years and aimed to characterize the systems according to crop production (t/ha); forage growth (kg DM/ha/day); forage production (kg DM/ha); and N fixation (kg N/ha) from legume production. Further, results about animal and system performance, such as liveweight production, liveweight gain, stocking rate and feed to gain ratio, were presented. Regarding management, supplementation data (kg DM/ha), fertilization (kg/ha) and pasture occupation were presented with the objective to understand how systems work.

Liveweight production (LWP, kg LW/ha/year) varied among systems. In general, CC and SR, i.e., those systems that included rearing stock in high proportion, had more LWP than LR and FR, which are associated with finishing cattle. This can be explained by the different biological efficiency of each stage, i.e., rearing vs. finishing [50]. This is evidenced by the differences in F/G ratio among systems, with CC and SR requiring, on average, 17.3% less kg of DM forage per each kg of LW produced. These LWP levels were similar to those reported by Terra and García-Préchac (1996–2000) [51] and Pereyra (2013–2017) [52], in the same experimental site on permanent pastures and annual grazing crops, without support area.

There were differences in LWP across years. Y1 had the lowest levels of production associated with climatic conditions that made seeding of pastures difficult (autumn–early winter), along with the fact that Y1 could be considered as a management adjustment year. Year two had higher levels of meat production than Y1, explained by a greater number of animals in CC and SR and higher levels of supplementation in LR and FR (in this system with fewer animals than Y1). During Y3, levels of production were similar to Y2 in CC (−1 kg LWP/ha) and LR (+5 kg LWP/ha), whereas in SR and FR, levels of production were reduced (−26 kg LWP/ha and −34 kg LWP/ha, respectively). Although DM production was higher in Y3 than Y1 and Y2, dry conditions and high temperatures during summer, which affected forage production and quality and determined heat-stress conditions to animals, could explain the reduction in LWP.

Strategic supplementation played an important role in systems, improving LWP. This effect was observed mostly in those systems with lower efficiency (finishing animals), where the use of supplements was the highest on average (LR) or low but with high impact, improving LWP (FR). This allows one to infer a certain dependency on supplementation in these systems compared with those that achieved higher levels of LWP with lower levels of supplement.

Autumn and winter were critical periods for liveweight gain (LWG, kg/ha/day), associated with fallows, seeding of pastures, high water content and low DM forage mass in pastures. The highest LWGs were observed during spring, explained by a peak of DM production and improvements in climatic conditions. This determined the moment when the most kg of liveweight was produced along the year and the moment when animals were ready to slaughter.

DM production had slight differences among years, despite variation in climatic conditions among years. These conditions affected the seasonal productivity and the intra-annual distribution more than the annual total production of DM. Further, FR and SR had the highest production on average. High levels of nitrogen fertilization in FR and the absence of fallow periods and growth rates of permanent pasture in SR could explain these results. However, dry conditions during summer strongly affected tall fescue in FR production and quality and gave rise to weed growth (mainly *Cynodon dactylon*).

Natural grassland is a key component in ICLSs and had a strategic use during adverse conditions, as a supporting area. These grasslands are mostly composed of C4 grasses with high DM production in spring–summer [53]. On the other hand, permanent pastures had high DM production in winter–spring, which allowed for complementary use of both grassland types and avoided overgrazing during critical periods for NG. Occupation of NG was different between systems; the highest occupation was in CC and SR. These systems with a short and without-pasture phase, respectively, had an important proportion of area in fallow period in autumn and spring (75% and 100% of area in rotation, respectively), which explained most of the use of NG, due to a reduction in the improved area. At the same time, these systems had low stocking rate during autumn, when grazing area is reduced. The use of permanent pastures was predominant in LR and FR and NG use was less than that for CC and SR.

Grain production varied among systems and there was a substantial effect of the pasture phase in grain yields. In Y1 and Y2, CC had less grain production than SR and LR. During Y3, CC had soybean production with similar values to LR. Along these lines, various authors report that the inclusion of pastures in a rotation with crops promotes better soil quality, associated with higher SOC, than those that do not include pastures [54]. Results presented by Terra and Macedo [55] showed that, in the same experiment, between 1995 and 2005, CC had significantly lower SOC than systems that rotated with pastures (i.e., LR and SR). Similarly, it has been reported that Brazilian ICLSs, with grazing animals, allow one to improve grain yield after the pasture phase, due to improved soil properties, i.e., soil microbial (mass, diversity) and soil structure (composition, density, porosity, nutrients) [56].

Climatic conditions (wet conditions in winter and dry conditions in summer) affected oat grain production in Y1 and sorghum grain and wheat grain production in Y3, respectively, which allowed us to only obtain by-products that were used as fibrous feed in livestock production. Although grain production in the current scheme of production is considered as an output of the systems, in some cases, it could be considered as an input to LWP (to feed animals), depending on variation in international prices, environmental conditions and the needs of each system. This flexibility in resource use is presented as an advantage in ICLS management.

Legume inclusion in the rotation supplied nitrogen to the system. Pasture phase fixed 27.8 ± 2.59 kg N per ha/year in LR, 52 ± 45.2 kg/ha/year in SR and 10.8 ± 7.42 in CC, on average. These values had high variability, depending on the age of pasture, driven by botanical composition and year, though represented an important contribution given the current fertilizer prices. Further, biological fixation of nitrogen is more efficient in terms of GHG emissions and energy use than N inputs from inorganic fertilizers, with similar values of losses to waterways [57]. Moreover, sowing legumes with high levels of condensed tannins, e.g., L. corniculatus L., as conducted in the permanent pasture of LR and permanent improvement in CC, is a way of reducing emissions per kg of DM consumed [58] and reducing N losses through leaching [59].

Livestock production contributes nutrients through excreta. Russelle et al. [24] high- lighted the importance of manure use to reduce costs and improve soil fertility. In Palo a Pique LTE, excreta are distributed homogenously within the boundaries of the systems, due to rotational stocking with a few days of permanence in each paddock and high stocking density. According to Ward et al. [25], N fixation and livestock excreta allow for nutrient cycling. These authors discuss the importance of the circularity of nutrients in livestock systems, associated with lower costs of production and lower environmental impacts. In this regard, Moraes et al. [56] reported that recycling of nutrients in the livestock phase is influenced by stocking rate and, in consequence, these systems export less nutrients out of the system than the crop phase.

Ruminant livestock can produce human food from human-inedible feedstuffs [19]. In the four systems evaluated, livestock played an important role by transforming grass into high-quality protein, i.e., kg of meat. The pasture phase allows one to produce feed for animals in marginal soils, where continuous cropping is unsustainable [31] and, at the same time, the use of high-quality pastures allows for improved liveweight production. The use of human-edible grains to feed animals is minimum, reducing the competition for resources [60].

Although the systems analyzed here lacked spatial replication because of the large- scale and multidisciplinary crop–livestock research approach, we presented three years of data that were considered as a replication in time. The main objective is to report the real results and coefficients from mixed livestock systems in Uruguay. In this regard, Murison and Scott [61] reported several published studies that used unreplicated treatments related to grazing livestock. They concluded that while treatments need to be replicated to allow for measurement of the experiment error, there are circumstances where appropriate scale may have priority over replication. On the other hand, the same authors reported the importance of assessing the whole-farm effects, emergent properties of the systems and, at the same time, individual productivity.

ICLSs present some opportunities related to international prices of commodities. However, there are also challenges, namely: (i) the dependence of external inputs to maintain high DM production in a scenario of price variability (i.e., fertilizer use); (ii) environmental issues associated with the need to reduce emissions per unit of product while maintaining high levels of production over time without wasting resources (i.e., forage quality and productivity, grazing management and C sequestration in soils, particularly in CC, where the rotation did not include a pasture phase); (iii) the need to adapt this kind of system through technologies to reduce the impact of climate change (i.e., diversification of forage basis in FR); (iv) the necessity to improve productivity, particularly in those systems that did not reach the proposed production levels (FR and LR), without increasing the use of human-edible food to feed animals (i.e., through improved forage utilization).

2.7. Conclusions

The four ICLSs evaluated had different levels of production. Those systems that included high proportion of rearing stock (Continuous Cropping and Short Rotation) reached the production target (400 kg LW/ha/year) and produced significantly more LW/ha than those with high proportion of finishing animals (Long Rotation and Forage Rotation), during the three years of evaluation. Therefore, the hypothesis was not fulfilled by the four systems evaluated. DM production was statistically different among systems, being higher in Forage Rotation and Short Rotation. Systems that rotate with pasture tended to have higher levels of crop production.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/agronomy12123023/s1, Table S1: Crude protein content in each pasture in the 'Palo a Pique' pasture–crop rotations long-term experiment in Treinta y Tres, Uruguay; Table S2: Metabolisable energy content in each pasture in the 'Palo a Pique' pasture–crop rotations long-term experiment in Treinta y Tres, Uruguay; Table S3: Neutral detergent fiber content in each pasture in the 'Palo a Pique' pasture–crop rotations long-term experiment in Treinta y Tres, Uruguay.

Author Contributions: Conceptualization, P.R. and W.A.; Methodology, F.P-G., P.R., W.A. and M.J.R.; Formal Analysis, F.P.-G.; Writing—Original Draft Preparation, F.P.-G., P.R., W.A. and M.J.R.; Writing—Review and Editing, F.P.-G., P.R., W.A. and M.J.R.; Supervision, P.R., W.A. and M.J.R.; Project Administration, P.R. and W.A. All authors have read and agreed to the published version of the manuscript.

Funding: This research was funded by: Long Term Experimental Platforms project (INIA), posgradu- ate fellowship (F.P.-G.): National Institute of Agricultural Research doctoral fellowship and National Agency of Research and Innovation (ANII) code MOV_CA_2021_1_171482 (movility fellowship). M.J.R contributions were funded by the Biotechnology and Biological Sciences Research Council (BBSRC) through the strategic program Soil to Nutrition (S2N; BBS/E/C/000I0320) at Rothamsted Re- search. The contributions from M.J.R. were also funded by the Natural Environment Research Council

(NERC) under research Program NE/W005050/1 AgZero+: Towards sustainable, climate-neutral farming. AgZero+ is an initiative jointly supported by NERC and BBSRC.

Data Availability Statement: Not applicable.

Acknowledgments: We thank INIA’s field and laboratory staff and students who participated in the collection and processing of data.

Conflicts of Interest: The authors declare no conflict of interest.

2.8. References

1. OECD/FAO. OECD-FAO Agricultural Outlook 2022–2031; OECD Publishing: Paris, France, 2022; ISBN 978-92-64-67537-7.

2. García-Préchac, F.; Ernst, O.; Siri-Prieto, G.; Terra, J.A. Integrating no-till into crop-pasture rotations in Uruguay. Soil Tillage Res.

2004, 77, 1–13. [CrossRef]

3. Bell, L.W.; Moore, A.D. Integrated crop–livestock systems in Australian agriculture: Trends, drivers and implications. Agric. Syst. 2012, 111, 1–12. [CrossRef]

4. Carvalho, P.C.F.; Anghinoni, I.; Moraes, A.; de Souza, E.D.; Sulk, R.M.; Lang, C.R.; Cassol, J.P.; Lazzarotto, M.; Silva, J.L.; Conte, O.; et al. Managing grazing animals to achieve nutrient cycling and soil improvement in no-till integrated systems. Nutr. Cycl. Agroecosys. 2010, 88, 259–273. [CrossRef]

5. Bell, L.W.; Moore, A.D.; Kirkegaard, J.A. Evolution in crop–livestock integration systems that improve farm productivity and environmental performance in Australia. Eur. J. Agron. 2014, 57, 10–20. [CrossRef]

6. Szymczak, L.S.; Carvalho, P.C.F.; Lurette, A.; Moraes, A.; de Albuquerque, P.A.; Posselt, A.; Moulin, C.H. System diversification and grazing management as resilience-enhancing agricultural practices: The case of crop-livestock integration. Agric. Syst. 2020, 184, 102904. [CrossRef]

7. Hochman, Z.; Carberry, P.S.; Robertson, M.J.; Gaydon, D.S.; Bell, L.W.; McIntosh, P.C. Prospects for ecological intensification of Australian agriculture. Eur. J. Agron. 2013, 44, 109–123. [CrossRef]

8. Franzluebbers, A.J.; Sawchik, J.; Taboada, M.A. Agronomic and environmental impacts of pasture–crop rotations in temperate North and South America. Agric. Ecosyst. Environ. 2014, 190, 18–26. [CrossRef]

9. Peyraud, J.L.; Taboada, M.; Delaby, L. Integrated crop and livestock systems in Western Europe and South America: A review. Eur. J. Agron. 2014, 57, 31–42. [CrossRef]

10. Ministerio de Ganadería, Agricultura y Pesca. Normativa de Suelos y Aguas, Uruguay. 2020. Available online: https://www.gub. uy/ministerio-ganaderia-agricultura-pesca/politicas-y-gestion/normativa-suelos-aguas (accessed on 18 July 2022). (In Spanish)

11. DIEA. Anuario Estadístico Agropecuario. Oficina de Estadísticas Agropecuarias, Ministerio de Ganadería, Agricultura y Pesca, Uruguay. 2021. Available online: https://www.gub.uy/ministerio-ganaderia-agricultura-pesca/comunicacion/noticias/diea- presento-anuario-estadistico-agropecuario-2021 (accessed on 18 July 2022). (In Spanish)

12. García-Préchac, F.; Ernst, O.; Siri-Prieto, G.; Salvo, L.; Quincke, A.; Terra, J.A. Long-term effect of different agricultural soil use and management systems on the organic carbon content of Urguay prairie soils. In Proceedings of the Global Symposium on Soil Organic Carbon, Rome, Italy, 21–23 March 2017.

13. Franco, J.G.; Bert, M.T.; Grabber, J.H.; Hendrickson, J.R.; Nieman, C.C.; Pinto, P.; Van Tassel, D.; Picasso, V. Ecological Intensifica- tion of Food Production by Integrating Forages. Agronomy 2021, 11, 2580. [CrossRef]

14. Díaz-Zorita, M.; Duarte, G.A.; Grove, J.H. A review of no-till systems and soil management for sustainable crop production in the subhumid and semiarid Pampas of Argentina. Soil Till. Res. 2002, 65, 1–18. [CrossRef]

15. Lussich, F. Variabilidad de la Fijación Biológica de Nitrógeno de Leguminosas Forrajeras en Uruguay: Posibles Causas y Consecuencias Nutricionales. Master’s Thesis, Universidad de la Republica, Montevideo, Uruguay, September 2020. (In Spanish)

16. Peoples, M.B.; Herridge, D.F.; Ladha, J.K. Biological nitrogen fixation: An efficient source of nitrogen for sustainable agricultural production? Plant Soil. 1995, 174, 3–28. [CrossRef]

17. Simioni, T.A.; Gomes, F.J.; Gomes Texeira, U.H.; Fernandes, F.A.; Botini, L.A.; Mousquer, C.J.; Rodrigues de Castro, W.J.; Hoffmann, A. Potencialidade da consorciação de gramíneas e leguminosas forrageiras em pastagens tropicais. Pubvet 2014, 8, 1551–1697. [CrossRef]

18. Kanter, D.R.; Schwoob, M.H.; Baethgen, W.; Bervejillo, J.E.; Carriquiry, M.; Dobermann, A.; Ferraro, B.; Lanfranco, B.; Mondelli, M.; Penego, C.; et al. Translating the Sustainable Development Goals into action: A participatory backcasting approach for developing national agricultural transformation pathways. Glob. Food Sec. 2016, 10, 71–79. [CrossRef]

19. Picasso, V.D.; Modernel, P.D.; Becoña, G.; Salvo, L.; Gutiérrez, L.; Astigarraga, L. Sustainability of meat production beyond carbon footprint: A synthesis of case studies from grazing systems in Uruguay. Meat Sci. 2014, 98, 346–354. [CrossRef]

20. Poffenbarger, H.; Artz, G.; Dahlke, G.; Edwards, W.; Hanna, M.; Russell, J.; Sellers, H.; Liebman, M. An economic analysis of integrated crop-livestock systems in Iowa, U.S.A. Agric. Syst. 2017, 157, 51–69. [CrossRef]

21. Broderick, G.A. Review: Optimizing ruminant conversion of feed protein to human food protein. Animal 2018, 12, 1722–1734. [CrossRef]

22. Oltjen, J.W.; Beckett, J.L. Role of Ruminant Livestock in Sustainable Agricultural Systems. J. Anim. Sci. 1996, 74, 1406–1409. [CrossRef]

23. Nie, Z.; McLean, T.; Clough, A.; Tocker, J.; Christy, B.; Harris, R.; Riffkin, P.; Clark, S.; McCaskill, M. Benefits, challenges and opportunities of integrated crop-livestock systems and their potential application in the high rainfall zone of southern Australia: A review. Agric. Ecosyst. Environ. 2016, 235, 17–31. [CrossRef]

24. Russelle, M.P.; Hentz, M.H.; Franzluebbers, A.J. Reconsidering Integrated Crop Livestock Systems in North America. Agron. J. 2007, 99, 325–334. [CrossRef]

25. Ward, S.M.; Holden, N.M.; White, E.P.; Oldfield, T. The 'circular economy' applied to the agriculture (livestock production) sector—Discussion paper. In Proceedings of the Workshop on the Sustainability of the EU's Livestock Production Systems, Brussels, Belgium, 14–15 September 2016.

26. Tanaka, D.L.; Karn, J.F.; Scholljegerdes, E.J. Integrated crop/livestock systems research: Practical research considerations. Renew. Agric. Food Syst. 2008, 23, 80–86. [CrossRef]

27. Scott, J.M.; Gaden, G.A.; Edwards, C.; Paull, D.R.; Marchant, R.; Hoad, J.; Sutherland, H.; Coventry, T.; Dutton, P. Selection of experimental treatments, methods used and evolution of management guidelines for comparing and measuring three grazed farmlet systems. Anim. Prod. Sci. 2013, 53, 628–642. [CrossRef]

28. Sayre, N.F.; deBuys, W.; Bestelmeyer, B.T.; Havstad, K.M. "The Range Problem" After a Century of Rangeland Science: New Research Themes for Altered Landscapes. Rangel. Ecol. Manag. 2012, 65, 545–552. [CrossRef]

29. Terra, J.A. Experimentos de largo plazo como plataforma agroambiental para la intensificación sostenible. Revista INIA 2017, 48, 67–72. (In Spanish)

30. Johnston, A.E.; Poulton, P.R. The importance of long-term experiments in agriculture: Their management to ensure continued crop production and soil fertility; the Rothamsted experience. Eur. J. Soil Sci. 2018, 69, 113–125. [CrossRef] [PubMed]

31. Rovira, P.; Ayala, W.; Terra, J.; García-Préchac, F.; Harris, P.; Lee, M.R.F.; Rivero, M.J. The 'Palo a Pique' long-term research platform: First 25 years of a crop–livestock experiment in Uruguay. Agronomy 2020, 10, 441. [CrossRef]

32. USDA (United States Department of Agriculture)—NRCS (National Resources Conservation Services). NSSC-SSL Report Uruguay. Soil characterization Data. In Primary Characterization Data (Uruguay); USDA-NRCS: Washington, DC, USA, 1996.

33. Terra, J.A.; Garcia-Préchac, F. Soil Organic Carbon content of a Typic Argiudoll in Uruguay under Forage crops and pasture for direct grazing: Effect of tillage intensity and rotation system. In Making Conservation Tillage Conventional: Building a Future on 25 Years of Research. In Proceedings of the 25th Annual Southern Conservation Tillage Conference for Sustainable Agriculture, Auburn, AL, USA, 22–24 June 2002.

34. Pravia, M.V.; Kemanian, A.R.; Terra, J.A.; Shi, Y.; Macedo, I.; Goslee, S. Soil carbon saturation, productivity, and carbon and nitrogen cycling in pasture-crop rotations. Agric. Syst. 2019, 171, 13–22. [CrossRef]

35. Terra, J.A.; Carámbula, M. Las Sequias, Antes, Durante y Después—Boletín de Divulgación 74; INIA: Montevideo, Uruguay, 2000. (In Spanish)

36. Thorn, E.C. The discomfort index. Weatherwise 1959, 12, 57–59. [CrossRef]

37. NOAA. Livestock hot weather stress. Oper. Man. Lett. 1976, C-31–C-76.

38. Beretta, A.; Bassahun, D.; Musselli, R. Medir el pH del suelo en reposo o agitando la mezcla suelo: agua? Agrociencia 2014, 18, 90–94.

39. Wright, A.F.; Bailey, J.S. Organic carbon, total carbon, and total nitrogen determinations in soils of variable calcium carbonate contents using a Leco CN-2000 dry combustion analyzer. Commun. Soil Sci. Plant Anal. 2001, 32, 3243–3258. [CrossRef]

40. Simmone, A.H.; Simmone, E.H.; Eitenmiller, R.R.; Mills, H.A.; Cresman III, C.P. Could the Dumas Method Replace the Kjeldahl Digestion for Nitrogen and Crude Protein Determinations in Foods? J. Sci. Food Agric. 1997, 73, 39–45. [CrossRef]

41. Bray, R.H.; Kurtz, L.T. Determination of total, organic and available forms of phosphorus in soils. Soil Sci. 1945, 59, 39–46. [CrossRef]

42. Jackson, M.L. Análisis Químico de Suelos; Omega: Barcelona, Spain, 1964; p. 662.

43. Ayala, W.; Carriquiry, E.; Carámbula, M. Caracterización y Estrategias de Utilización de Pasturas Naturales en la Región Este—Boletín de Divulgación 49; INIA: Treinta y Tres, Uruguay, 1993; pp. 1–28. (In Spanish)

44. Lynch, P.B. Methods of measuring the production from grasslands. N. Z. J. Sci. Tec. 1947, 28, 385–405.

45. AOAC. Official Methods of Analysis, 15th ed.; Association of Official Analytical Chemist: Washington, DC, USA, 1990.

46. Di Rienzo, J.A.; Casanoves, F.; Balzarini, M.G.; Gonzalez, L.; Tablada, M.; Robledo, C.W. InfoStat Versión 2020. Centro de Transferencia InfoStat, FCA, Universidad Nacional de Córdoba, Argentina. Available online: http://www.infostat.com.ar (accessed on 18 July 2022).

47. Terra, J.A.; Scaglia, G.; García-Prçhac, F. Moha: Características del Cultivo y Comportamiento en Rotaciones Forrajeras con Siembra Directa—Serie Técnica 111; INIA: Montevideo, Uruguay, 2000. (In Spanish)

48. Perrachón, J. Verdeos de Verano. Un Seguro Para épocas Difíciles. Available online: https://www.planagropecuario.org.uy/publicaciones/revista/R135/R_135_61.pdf (accessed on 19 July 2022).

49. Lemaire, G.; Franzluebbers, A.; Carvalho, P.C.F.; Dedieu, B. Integrated crop -livestock systems: Strategies to achieve synergy between agricultural production and environmental quality. Agric. Ecosyst. Environ. 2014, 190, 4–8. [CrossRef]

50. CSIRO. Nutrient Requirements of Domesticated Ruminants; CSIRO Publishing: Clayton, Australia, 2007.

51. Terra, J.A.; Garcia-Préchac, F. Siembra Directa y Rotaciones Forrajeras en las Lomadas del Este: Síntesis 1995–2000—Serie Técnica 125; INIA: Treinta y Tres, Uruguay, 2001. (In Spanish)

52. Pereyra, F. Efecto de la Inclusión del Endófito AR584 en la Producción de Festuca arundinacea y la Performance Animal Asociada. Master’s Thesis, Universidad de la República, Montevideo, Uruguay, June 2019. (In Spanish)

53. Modernel, P.; Rossing, W.A.H.; Corbeels, M.; Dogliotti, S.; Picasso, V.; Tittonell, P. Land use change and ecosystem service provision in Pampas and Campos grasslands of southern South America. Environ. Res. Lett. 2016, 11, 113002. [CrossRef]

54. Studdert, G.A.; Echeverría, H.E.; Casanovas, E.M. Crop-Pasture Rotation for Sustaining the Quality and Productivity of a Typic Argiudoll. Soil Sci. Soc. Am. J. 1997, 61, 1466–1472. [CrossRef]

55. Terra, J.A.; Macedo, I. Twenty years no-till crop-pasture rotation systems impacts on soil organic carbon. In Proceedings of the 20th International Soil

Tillage Research Organization Conference (ISTRO), Nanjing, China, 14–18 September 2015.

56. Moraes, A.; Carvalho, P.C.F.; Anghinoni, I.; Lustosa, S.B.C.; de Andrade Costa, S.E.V.G.; Kunrath, T.R. Integrated crop-livestock systems in the Brazilian subtropics. Eur. J. Agron. 2014, 57, 4–9. [CrossRef]

57. Ledgard, S.; Schils, R.; Eriksen, J.; Luo, J. Environmental impacts of grazed clover/grass pastures. Irish J. Agric. Food Res. 2009, 48,

209–226.

58. Woodward, S.L.; Waghorn, G.C.; Ulyatt, M.J.; Lassey, K.R. Early indications that feeding Lotus will reduce methane emissions from ruminants. Proc. N. Z. Soc. Anim. Prod. 2001, 61, 23–26.

59. Carulla, J.E.; Kreuzer, M.; Machmu˝ller, A.; Hess, H.D. Supplementation of Acacia mearnsii tannins decreases methanogenesis and urinary nitrogen in forage-fed sheep. Aust. J. Agric. Res. 2005, 56, 961–970. [CrossRef]

60. Wilkinson, J.M.; Lee, M.R.F. Review: Use of human-edible animal feeds by ruminant livestock. Animal 2018, 12, 1735–1743. [CrossRef] [PubMed]

61. Murison, R.; Scott, J.M. Statistical methodologies for drawing causal inference from an unreplicated farmlet experiment conducted by the Cicerone Project. Anim. Prod. Sci. 2013, 53, 643–648. [CrossRef]

3. Carbon footprint of mixed farming crop-livestock rotational-based grazing beef systems using long term experimental data

Fabiana Pereyra Goday[1,2*,2]; Asma Jebari[2]; Taro Takahashi[3,4]; Pablo Rovira[1]; Walter Ayala[1]; Michael R.F. Lee[5]; M. Jordana Rivero[2]; Graham A. McAuliffe[2,5]

[1]Instituto Nacional de Investigación Agropecuaria (INIA), Treinta y Tres, 33000, Uruguay

[2]Rothamsted Research, Net Zero & Resilient Farming, Okehampton, EX20 2SB, UK

[3]Agri-Food and Biosciences Institute, Hillsborough, BT26 6DR, UK

[4]University of Bristol, Langford, BS40 5DU, UK

[5]Harper Adams University, Edgmond, Shropshire, TF10 8NB, UK

[*]Corresponding author: Fabiana Pereyra Goday (fpereyragoday@gmail.com)

3.1. Resumen

En un contexto de demanda de alimentos y preocupaciones asociadas con los impactos ambientales de los sistemas agrícolas, hay un interés creciente en las explotaciones agrícolas mixtas para lograr una mayor sostenibilidad en comparación con los sistemas de producción agrícolas exclusivos. Sin embargo, las evaluaciones de estos sistemas son complejas y requieren datos de alta resolución para determinar el verdadero valor e interconectividad. Dada la escasez de información sobre sistemas mixtos de cultivos y ganadería y las dificultades de su análisis, realizamos un análisis de ciclo de vida utilizando datos de alta resolución temporal (2019-2022) de un experimento a largo plazo en América del Sur, para evaluar las intensidades de emisiones de gases de efecto invernadero de 'cuna a la portera' de cuatro sistemas rotacionales de cultivos y ganadería. Los sistemas evaluados fueron: Cultivo Continuo: dos años de cultivo continuo; Rotación Corta: dos años de Cultivo Continuo más dos años de pasturas; Rotación Larga: dos años de Cultivo

[2] Facultad de Agronomía, Universidad de la República.

Continuo seguidos por cuatro años de pasturas; y Rotación Forrajera: pastura continuo. Las intensidades de emisiones para la producción de carne se reportaron como kg de dióxido de carbono equivalente (CO_2-eq)/kg de ganancia de peso vivo (GPV) utilizando los factores de caracterización del dióxido de carbono del Sexto Informe de Evaluación (AR6 2021) del Panel Intergubernamental sobre Cambio Climático. Los resultados de estimación fueron: 11.3, 11.8, 11.8 y 16.4 kg CO_2-eq/kg GPV para Cultivo Continuo, Rotación Corta, Rotación Larga y Rotación Forrajera, respectivamente. Los promedios de emisiones derivadas de los cultivos, que se separaron de las emisiones basadas en animales utilizando la asignación económica, fueron: 1.23, 0.53 y 0.52 kg CO_2-eq/kg para soja, trigo y avena, respectivamente. La inclusión de los cambios en el stock de carbono orgánico del suelo tuvo efectos notables en la reducción de las emisiones de cada sistema: en un 22.4%, 19.2%, 25.3% y 42.1% en Cultivo Continuo, Rotación Corta, Rotación Larga y Rotación Forrajera, respectivamente, cuando se incluyó el carbono orgánico del suelo. Dado que hay pocos estudios de análisis de ciclo de vida disponibles sobre estos sistemas mixtos, particularmente con datos primarios de alta resolución, este estudio añade conocimiento crítico a la literatura sobre sostenibilidad relacionada con la producción de alimentos al abordar problemas ambientales en sistemas de producción complejos en comparación con la cobertura existente y amplia de los sistemas agrícolas.

3.2. Summary

In the context of ever-growing demand for food and associated concerns regarding the environmental impacts of high-input agricultural systems, there is growing interest in mixed farm enterprises to deliver greater sustainability compared with mono-enterprise production systems. However, assessments of such systems are complex and require high resolution data to determine the true value and interconnectivity across enterprises. Given the scarcity of information on mixed crop livestock systems and the difficulties of its analysis, we perform Life Cycle Assessment using temporally high-resolution data (2019-2022) from a long-term experiment in South America to evaluate the ‘cradle-to-farmgate exit’ greenhouse gas emissions intensities of four rotational crop-livestock systems. Systems evaluated were Continuous Cropping: two-years of continuous cropping; Short Rotation: two-year Continuous Cropping plus two-year pasture; Long Rotation: two-year Continuous Cropping followed by four-year pasture; and Forage Rotation: continuous pasture. Emissions intensities for beef throughput were reported as kg carbon dioxide equivalents (CO_2-eq)/kg liveweight gain using the Intergovernmental Panel for Climate Change’s Sixth Assessment Report (AR6 2021) CO_2 characterization factors. Point estimate results were found to be: 11.3, 11.8, 11.8 and 16.4 kg CO_2-eq/kg/LWG for Continuous Cropping, Short Rotation, Long Rotation, and Forage Rotation, respectively. Emission averages arising from crops, which were separated from animal-based emissions using economic allocation were: 1.23, 0.53 and 0.52 kg CO_2-eq/kg for soybean, wheat, and oat, respectively. The inclusion of soil organic carbon stock changes had notable effects on reducing each system’s emissions: by: 22.4%, 19.2%, 25.3% and 42.1% under Continuous Cropping, Short Rotation, Long Rotation and Forage Rotation, respectively when soil organic carbon was included. Given there are few Life Cycle Assessments studies available on such mixed-enterprise ‘semi-circular’ systems, particularly with novel primary data, this study adds critical knowledge to agri-food related sustainability literature by addressing environmental issues in complex production systems compared to extant and broad coverage of mono enterprise systems.

Keywords: carbon footprint; sustainability; food security; grazing

3.3. Introduction

Demand for agricultural produce is expected to grow between 1.1 and 1.5% per year over the next ten years driven primarily by an ever-increasing global population (OECD/FAO 2022). Meeting this new market demand presents many broad sustainability challenges, not least optimizing agricultural land use to ensure adequate and equitable nutritional provision whilst increasing crop intensity (i.e., yields) and improving herd efficiency through, for instance, higher feed conversion ratios in livestock systems (McAuliffe et al. 2017; OECD/FAO 2022). Global agricultural productivity will need to be increased by 28% over the next decade. To make matters even more complicated, to achieve the Paris Climate Change Agreement's reduction targets related to agriculturally-sourced greenhouse gas (GHG) emissions, production increases cannot be solved globally simply by increasing quantities of material inputs (e.g., inorganic fertilization; imported feed) as observed in 'conventional' or intensive farming practices.

The livestock sector is thought to be responsible for ~11% of anthropogenically induced GHG emissions globally (FAO 2023). Animal feed related GHG emissions contribute considerably to both monogastric and ruminant production systems' which are predominantly associated with direct and indirect soil emissions (i.e., nitrous oxide, N_2O, and carbon dioxide, CO_2; McAuliffe et al. 2017) as well as direct and indirect land use change (e.g., deforestation and soil inversion CO_2 emissions). Depending on the level of pre- and post-farmgate manufacturing and processing, high proportions of system-scale GHGs are typically produced at the farm level as demonstrated by one of the few extant studies which explores the full supply-chain of beef systems (Asem-Hiablie et al. 2019). In Uruguay, meat represented 18% of total exportations in 2023 (Uruguay XXI 2023), thus demonstrating the importance of domestic animal-based agricultural enterprises. According to the National Inventory Report (SNRCC - MA 2021), which uses Global Warming Potential over a 100-year time-horizon (GWP_{100}), emissions from enteric fermentation in livestock, predominately from ruminants, accounted for 45.7% whilst emissions of N_2O from managed soils accounted for 20.3% of the sector's total emissions. Most Uruguayan ruminant livestock production occurs on natural grasslands (NG, i.e., native pastures with low, or no, inputs and generally extensive land occupation; de Faccio Carvalho et al. 2021). However, due to the aforementioned increase in food demand globally (in addition to higher international prices of material input commodities driven by global shocks, e.g., the war in Ukraine; Rawtani et al. 2022), Uruguayan farmers are diversifying their activities. For instance, non-native pasture species are being introduced to swards to improve animal performance and soil quality (García-Préchac et al. 2004) whilst potentially reducing their GHG emissions via lower nitrogen content in excreta (Soteriades et al. 2019) and/or increased digestibility in forage (Takahashi et al. 2019). Mixed crop livestock systems fit into a (semi-)circular economy concept by producing crop on-farm to either sell directly or feed animals, rather than purchasing it externally. Such approaches reduce feed purchase risks with respect to market volatility (Mustafa et al. 2023) and have a knock-on effect of reduced land use and resource depletion (e.g., rock phosphate) as most feed is produced using the same farm's by-products (e.g., manure and excreta producing natural soil nutrient regeneration).

Vast amounts of scientific publications address environmental impacts associated with livestock production using Life Cycle Assessment (LCA) (de Vries and de Boer 2010; de Vries et al. 2015; Takahashi et al. 2019). However, addressing environmental impacts through LCA in crop-livestock systems is a challenge due to inherent complexities surrounding shared land producing multiple co-products exacerbated by scarce primary data availability. Furthermore, complexities surrounding soil organic carbon (SOC) sequestration and how to evaluate soil carbon dynamics in agricultural LCA is also problematic (Goglio et al. 2015). Crop-livestock systems represent 17% of the total agricultural area in Uruguay (DIEA - MGAP 2022), meaning there are substantial opportunities to simultaneously explore environmental trade-offs of rotational systems whilst adding to the knowledge base both globally (of mixed-farming systems) and nationally (of underrepresented nations with high-quality life cycle inventory analysis (LCI) data). The primary aim of this work, therefore, was to compare 'semi-circular', multi-produce cattle rearing systems widely adopted in Uruguay with a ‘traditional’ forage-only cattle system, whilst analyzing their potential impacts related to climate change. To achieve this, we evaluated ‘cradle-to-farmgate exit’ GHG emission intensities of four mixed crop-livestock systems in Uruguay using GWP_{100} (IPCC 2019a; IPCC 2021) with different intensities in land use over a three-year period (2019-2022). As the use of GWP has been questioned in terms of appropriate allocation of environmental burden in livestock systems (Manzano et al., 2023), in addition, we evaluated the methodological effect of using an alternative climate-related metric, specifically global temperature change potential over a 100-year time horizon (GTP_{100}; based on the modelled temperature impact of different gases relative to CO_2 at a specified time following an emission pulse) using IPCC’s (2021) GTP characterization factors which bestow substantially lower CO_2-eq coefficients to biogenic methane (~6 compared to GWP_{100}’s ~27).

3.4. Materials and methods

This study follows international protocols to calculate carbon footprints using an LCA approach as recommended by BSI PAS 2050 (2011) and ISO 14044 (ISO 2006) to compare emissions intensities of different pasture-based cattle production systems integrated with cropping systems (Segura et al. 2023). Typically, as opposed to other novel methodological applications described by McAuliffe et al. (2020), LCA comprises four steps: (1) goal and scope definition, (2) life cycle inventory analysis (LCI), (3) life cycle impact assessment (LCIA), and (4) interpretation (e.g., sensitivity and uncertainty analyses). We covered the entire production cycle in the case of crops from winter 2019 to summer 2022 (Southern Hemisphere seasons). In the case of livestock production, however, the system boundary focusses on post-weaning stages of the cattle life cycle as the systems under investigation raise cattle at various growth stages (i.e., rearing and finishing). Multi-produce systems, which will be described in detail in Section 2.1, are entirely interlinked; in other words, crops receive nutrients from grazing cattle whilst the same animals receive feed from crops, thereby making each output a co-product at the system boundary scale.

Figure 1. Palo a Pique long term experiment, Treinta y Tres, Uruguay. The picture shows several plots of crop–livestock systems. Credits: M. Oxley.

3.4.1. Study site

The long-term Pasture Crop Rotation experiment adopting no-tillage management was installed in 1995 at the 'Palo a Pique' experimental platform in Treinta y Tres (33º 16' S, 54° 29' W), a multifunctional farm-scale trial supported by the National Institute of Agricultural Research (INIA) in Uruguay (**Figure 1**). The annual mean (± SD) accumulated rainfall in the experimental site from 1995 to 2022 was 1249 ± 72.0 mm per year. The mean, maximum, and minimum air temperatures for the same period were 23 ± 0.1 °C and 11 ± 0.6 °C, respectively. The experimental design of each system is shown in **Table 1** (Pereyra Goday et al., 2022).

It should be noted that measurements and subsequent LCI development in this paper utilizes data from the third phase of the Palo a Pique Long-Term Experiment ('Land Expansion and Livestock Intensification'), which started in 2019. In 2019, an experimental redesign was carried out to better-reflect local, 'on the ground' farming as described by Rovira et al. (2020). Relevant changes occurring during this transition were the relocation of the permanent pasture system, the addition of grassland as a support area (i.e., a ‘safety net’ of land dedicated to minimizing effects of potential biotic and abiotic stresses) in each system, and the inclusion of unique livestock production strategies for each system which reflects typical farming practices in the study site’s region.

Table 1. Pasture and crop sequence for each rotation at Palo a Pique long term experiment, Treinta y Tres, Uruguay. P: Pasture follows by the age of the pasture (1 to 2 in short rotation and 1 to 4 in long rotation). Note that primary data from 4 to 6 year has not yet been collected, but due to the rotational nature within each system on an annual basis, the full six-year cycle can be represented from primary data collected during years one-to-three).

Rotation	Purpose of crop phase	Year 1	Year 2	Year 3	Year 4	Year 5	Year 6
Continuous Cropping	Crop	Oat/Sorghum	Black Oat/ Soybean	Wheat/Sorghum			
	Grazing	Oat/Sorghum	Ryegrass/Moha	Oat/Sorghum			
Short Rotation	Crop	Idem[2] CC	Idem CC	Wheat + P1	P2		
	Grazing	Idem CC	Idem CC	P1	P2		
Long Rotation	Crop	Idem CC and SR	Idem CC and SR	Wheat + P1	P2	P3	P4
	Grazing	Idem CC and SR	Idem CC and SR	P1	P2	P3	P4
Forage Rotation	Grazing	Fescue	Fescue	Fescue	Fescue	Fescue	Fescue

The continuous cropping system (CC, 12 ha) operates under a rotation with two crops per year. Continuous cropping does not rotate with pastures, but it is complemented with an external area (6 ha) of an improved pasture comprising tall fescue (*Festuca arundinacea* L.), birdsfoot trefoil (*Lotus corniculatus L*.), and white clover (*Trifolium repens L*.) which is re-seeded every five years with the same species to ensure sustained establishment. The short rotation (SR, 24 ha) alternates two years of crop production identical to CC followed by two years of grass-legume pastures utilizing Yorkshire fog (*Holcus lanatus L*.) and/or Italian ryegrass (*Lolium multiflorum L*.) interspersed with red clover (*Trifolium pratense L.*) at a target coverage rate of 50%. The long rotation system (LR, 36 ha) also alternates two years of crops identical to CC and SR followed by four years of grass-legume pastures composed of tall fescue, birdsfoot trefoil and white clover. The fourth system, forage rotation system (FR, 24 ha) is seeded with tall fescue and does not rotate with arable crops just between forage (tall fescue) paddocks.

Each crop-livestock system (i.e., systems that rotate crop and pastures: CC, SR and LR) was split into two halves within paddocks (Table 1): one half for human-edible crop production (defined as ‘crop area’), which was seeded with oat (*Avena byzantina L.*), black oat in CC (*Avena strigosa*), and wheat (*Triticum aestivum*) in winter, and soybean (*Glycine max*) and sorghum (*Sorghum sudanense L.*) in summer. The remaining areas were allocated to grazing cattle (defined as ‘livestock area’) which were seeded as follows: Italian ryegrass and oat in winter, with sorghum and moha (*Setaria italica L.*) in the following summer. Winter crops and pastures were sown between March and June and typically harvested in November. Summer crops were planted in October and November with harvesting occurring in April. Cover crops (i.e., black oat) were harvested for hay in October thus providing additional feed provision for cattle. For CC, SR and LR, animals enter their respective experimental farm platforms in April or May each year and remained for one year (rearing animals) or, in the case of finishing animals, until delivery of target weights for the slaughterhouse. GHG emissions were calculated for all animals using IPCC (2019) equations using an individual-animal approach originally detailed in McAuliffe et al. (2018) which were included in the LCI to ensure that systems with poorer performing animal’s GHGs were captured (e.g., particularly in the case of animals spending > one year on-farm). As reported by Pereyra Goday et al. (2022), CC system focused on rearing male calves with 32 reared in 2019, 34 in 2020 and 35 in 2021. In SR, rearing heifers were managed, with 44, 49, and 46 reared in 2019, 2020 and 2021, respectively; finishing cattle during May and September, with 15, 10, and 10 cattle finished in 2019, 2020 and 2021, respectively. In LR, the objective was rearing male calves and finishing steers over a period of 18 months with 50 male calves (~six months old) allocated to LR in 2019, 2020, and 2021. FR was the only system that begins at the end of the spring (Nov-Dec) with yearling steers. The objective of the livestock strategy in FR was to produce a finished steer ready for slaughter in 12-15 months (47, 30, 35, and 41 steers entered the system in December 2018, December 2019, November 2020, and November 2021, respectively). The four systems maintained British beef breeds (Aberdeen Angus and Hereford – Angus cross), randomly distributed. Data pertaining to animal performance is shown in **Table 2**.

Table 2. Animal performance average (2019-2022) in pasture crop rotation at the Palo a Pique long term experiment in Treinta y Tres, Uruguay. (ADG: average daily gain; TOF: time on farm as days).

Pasture Crop Rotation	Entry Weight (kg/animal)	Exit Weight (kg/animal)	ADG[1] (kg/animal/day)	TOF[2]
Continuous Cropping – Calves	190±23.6	377±35.9	0.59±0.264	330±22
Short rotation – Heifers	156±19.7	333±31.5	0.58±0.197	316±8
Short Rotation – Cows	473±51.8	529±51.2	0.59±0.308	102±21
Long Rotation – Calves	185±26.6	361 ±36.8	0.58±0.153	337±39
Long Rotation – Steers	369±34.1	511±37.2	0.65±0.179	224±25
Forage Rotation – Steers	313±65.5	497±45.1	0.65±0.316	279±6

Importantly, the experiment lacks complete replications for the full duration of the trial; however, due to the statistical design of the four individual systems, all phases of rotations occur each year represented by paddocks of 3 ha in CC, SR, and LR, thus enabling modelling of the entire rotations by proxy. In other words, the systems are assumed to be operating at steady state over six years based on three years of high-resolution primary data collection and analysis. Forage rotation's 24-ha area was divided into five paddocks of 4.8 ha each corresponding with tall fescue seeded in 2013 (9.6 ha), 2014 (9.6 ha), and 2020 (4.8 ha). Each rotation has an auxiliary 'support area' of natural grassland (NG) to ensure the animals have grazing access under conditions outside of a farmers' control (e.g., during periods with low forage availability in the systems due to (a)biotic stressors), thus ensuring the animals are maintained independently within each system (i.e., each system comprising livestock has its own dedicated area to avoid cross-system 'contamination'). The proportion of NG area (in addition to primary seeded pasture land occupation) was: 33%, 29%, 26%, and 33% of the total area for CC, SR, LR, and FR, respectively. Cattle grazed annual forage crops (Italian ryegrass, oat, sorghum, and moha), permanent pastures (in forage and crop areas), permanent improved pasture (in CC) and NG. Detailed information about the experimental design, management and yields can be found in Pereyra Goday et al. (2022) and Rovira et al. (2020).

3.4.2. System boundaries and functional unit

A schematic of system components and boundaries is provided in **Figure 2**. As mentioned above, the two subsystems (livestock and crop) are interconnected through livestock grazing on permanent pastures whilst simultaneously receiving and providing nutrients to the crop area. This is accomplished via the circularization of nutrients from urine and dung, and the production of sorghum and hay in crop areas to feed animals in the livestock area.

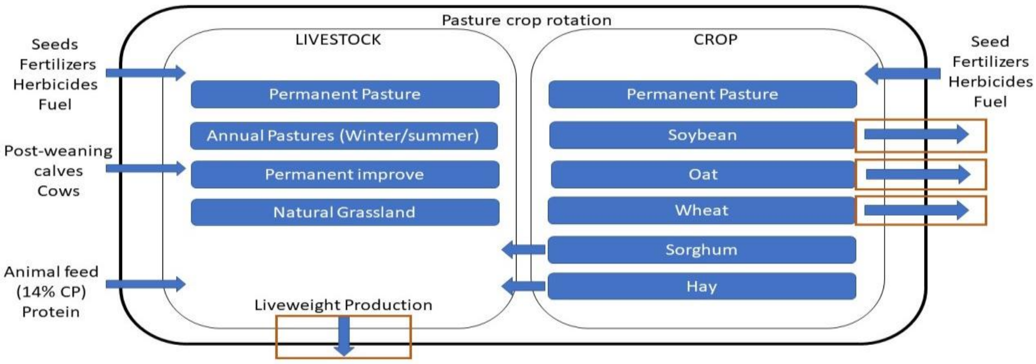


Figure 2. Pasture crop rotation components and boundaries at Palo a Pique. The external black line represents system boundary of the study. Red

rectangles and arrows show outputs from the long-term experiment in Treinta y Tres, Uruguay (i.e., produce sold to downstream stakeholders).

The boundary adopted was ‘cradle to farmgate exit’ as described by McAuliffe et al. (2018) which focused on finishing beef systems. The logic behind this is that the suckler herd is not part of Palo a Pique long term experiment, and therefore there is little-to-no data available and furthermore, the objective of this study was to quantify environmental impacts of the mixed crop-livestock trial’s systems and their potential differences in terms of GHG emissions, which would have been diluted if the suckler herd was included using secondary data (e.g., commercial LCA databases) thereby obscuring differences between common farming practices in the study-site’s region.

Regarding crop production, the entire cycle from seed production to harvest was considered, as well as all up-stream emissions associated with material inputs such as fertilizer, in line with the crop-livestock systems (CC, SR, and LR; Figure 2). Each crop cycle (summer and winter, respectively) takes about six months under regional, seasonal climatic conditions. All inputs (e.g., fertilizers, pesticides, etc.) and outputs (i.e., GHG losses to air and co-products) related to each production process were quantified, whereas farm buildings and infrastructure processes were excluded as they are considered negligible (McAuliffe et al. 2018) in certain agricultural systems, particularly at the study site where animals remain outdoors all year with the exception of calving, which is outside the current system boundary.

Given that the assessed systems have different outputs (four crops, including grass, and beef liveweight gain (LWG)), the functional unit considered was (1) 1 kg of product obtained, with results presented as: kg carbon dioxide equivalent (CO_2-eq)/(a) kg of LWG; (b) kg of soybean; (c) kg of oat; and (d) kg of wheat; and (2) 1 hectare. Given complexities surrounding disaggregating GHG emissions between animals and plants (i.e., they both ‘share’ the same land and both produce multiple co-products beyond the farmgate; Guinée et al., 2004), livestock production was separated from crop production using farm records of sales (i.e., the primary source of income for each of the systems) and subsequently economic allocation (see section 2.4 for further information).

3.4.3. Inventory analysis and impact assessment

The majority of model parameters utilized in this study were collected as primary trial-based data. All animals were weighed every 30 days and individual performance was calculated as daily LWG assuming linear growth between weighing events. Three grazing exclusion cages (0.4 × 1.0 m) were used per grazing paddock (3–5 ha) to estimate daily pasture growth as kg dry matter (DM)/ha/d every 30 days according to the methodology proposed by Lynch (1947). Crude protein (CP, %), metabolizable energy (ME, MJ/kg DM), and neutral detergent fibre (NDF, %) were analyzed monthly and analysis was conducted using standard methods (AOAC 1990) in the Animal Nutrition Laboratory of INIA La Estanzuela (Colonia, Uruguay). Annual SOC stock rate changes were estimated using best available measured data between 2015 and 2021 (30 cm depth), as described by Pravia et al. (2019) and then multiplied by the number of experimental years (i.e., three) for each paddock. Soil samples were analyzed according to Wright and Bailey (2001) in the Plant, Soil and Water Laboratory in INIA La Estanzuela (Colonia, Uruguay).

Information of all inputs in each system (2019-2020, 2020-2021 and 2021-2022) are reported in **Table 3**. The LCI was calculated from data reported by Pereyra Goday et al. (2022) and from the experiment's management records. Background processes such as transport-based emissions were sourced from the ecoinvent database (Wernet et al. 2016). Embedded emissions associated with the production of fertilizers, pesticides, seed production, and minor quantities of supplemental feed were sourced from geographically representative data provided by INIA (2022). Emissions from livestock, pastures, and crops were estimated using IPCC's (2019b) Tier 2 approach. GHG emissions arising directly and indirectly from animals were calculated for each period between two weighing intervals (30 days between each weighing event) using data from individual animals within the LCI. Pasture quality was also measured during each interval to align CP (%) and digestible energy (DE, %) with temporal growth rates (**Supplementary Material 1**). Once the LCI was conducted capturing temporal variability, all GHGs were summed to obtain a value of total emissions per system to provide clear interpretation. Emissions from pastures and crops were additionally calculated separately and summed to obtain the total of each system. Specific equations and constants used are detailed in **Supplementary Material 2**.

Lastly, GHG emissions were estimated according to IPCC (2019) refinements using the global warming potential over a 100-year time horizon (GWP_{100}) characterization factors detailed in the Sixth Assessment Report (AR6; IPCC 2021). All systems were modelled in SimaPro V9.3.0.3 (PRé Consultants 2022) and LCIAs were subsequently interpreted using the same software (details regarding interpretation provided in Section 2.4). Within SimaPro's latest IPCC GWP_{100} impact assessment (excluding carbon feedback), biogenic CH_4 and N_2O are respectively assumed to have 27.2 and 273 times greater climatic impacts than CO_2 (IPCC 2021; PRé Sustainability).

Table 3. Inventory of all major material inputs and outputs for each pasture crop rotation (three years of data, 2019-2022), at the Palo a Pique long term experiment, Treinta y Tres, Uruguay. (CP: crude protein).

Parameter	Unit	Continuous Cropping	Short Rotation	Long Rotation	Forage Rotation
Total area [1]	ha	24	34	56	36
Pasture crop rotation area	ha	12	24	36	24
Permanent improved area	ha	6	0	0	0
Natural Grassland area	ha	6	10	20	12
Yield					
Soybean	kg	21,276	22,335	24,840	-
Wheat	kg	13,272	16,125	15,531	-
Sorghum [2]	kg	29,772	33,903	31,758	-
Oat	kg	-	10,896	13,074	-
Liveweight production	kg	19,659	27,610	48,796	29,792
Fertilizer					
N	kg	3,547	5,337	5,474	11,615
P (P_2O_5)	kg	2,759	3,623	3,897	2,045

K (K_2O)	kg	1,572	1,981	1,916	432
S	kg	191	227	335	0
Pesticides [3]	L	533	634	625	113
Seeds	kg	4,018	5,235	6,229	393
Diesel for machinery	L	1,145	1,727	1,724	654
Feed for animals					
Protein (48% CP)	kg	385	846	4,293	-
Supplement (14% CP)	kg	-	-	8,327	16,280

3.4.4. Interpretation

For sensitivity analyses, given the unique mixed-farm 'semi-circular' systems, we presented results primarily on an output basis disaggregated by each commodity's total revenue across the three years of primary data collection. Based on best practice and evidence that allocation can have a profound effect on interpretation (Rice et al. 2017), we also report decomposed emissions between co-products using mass allocation to separate multifunctional-system outputs (i.e., the total yield of each product leaving the farmgate).

In addition to testing allocation assumptions, recent work has demonstrated the effect of functional unit choices on agri-food LCA results (e.g., McAuliffe et al. 2023; Manzano et al., 2023) in the context of nutritional value (e.g. protein), and Zira et al. (2021) who explored differences between mass/volume and area-based functional units. As each system has a different land occupation and various combinations of co-products (or indeed a single commodity in the case of FR), we also calculated LCIA on an area basis (1 ha) and reported the results to add novel evidence to earlier work carried out on agricultural functional units, particularly given the low representation of mixed crop-livestock systems in the sustainability literature combined with the interconnectivity between each (co)product as discussed at the beginning of this section.

Given on-going debates concerning LCA subjectivity and the effect of impact assessment method choices (e.g., Lynch 2019), following the procedure proposed by McAuliffe et al. (2023b) we also calculated global temperature change potential over a 100-year time horizon (GTP_{100}; based on the modelled temperature impact of different gases relative to CO_2 at a specified time following an emission pulse) using IPCC's (2021) GTP characterization factors which bestow substantially lower CO_2-eq coefficients for biogenic methane (~6 compared to GWP_{100}'s ~27). Carbon dynamics were tested for sensitivity by analyzing each system with and without SOC uptake included.

Finally, a Monte Carlo analysis was carried out to assess uncertainties both within (i.e., 95% confidence intervals) and across systems using pairwise iterations in the latter case. Monte Carlo simulations were conducted within SimaPro V9.3.0.3 (Pré Consultants 2022) and each assessment was run under 1,000 permutations. Distributions of individual GHGs were calculated manually using IPCC (2019) emission factor ranges (see McAuliffe et al. 2018's supplementary material for individual gas's distribution shapes), whilst ecoinvent's Pedigree Matrix was used to determine uncertainties associated with background (i.e., embedded) emissions.

3.5. Results and discussion

3.5.1. Intersystem comparison

Productivity and subsequent yields were found to be different among systems (Pereyra Goday et al. 2022). From May 2019 to April 2022, systems that included high proportions of rearing animals (CC and SR) produced higher levels of LWG per ha than finishing animals due to the typical growth curve of beef cattle (CSIRO 2007). LWG was 13% lower in LR (369 kg/ha/year) and 26% lower in FR (310 kg/ha/year) relative to CC and SR (426 and 418 kg/ha/year respectively). Crop production per ha was influenced by the presence of the pasture phase in the rotation and was consequently higher in LR and SR than CC. Climatic conditions (drought and water excess) during the experimental period explain resultant high variability in wheat and oat yields. Crop yield (t/ha/year) for soybean was 2.36 ± 0.136, 2.48 ± 0.348 and 2.76 ± 0.227 in CC, SR and LR respectively, whereas crop yield for wheat was

2.21 ± 2.051, 2.68 ± 2.015 and 2.59 ± 2.022 in CC, SR and LR, respectively. Oat yield was 1.82± 0.871 and 2.18 ± 0.028 to SR and LR.

Although area (i.e., land use/occupation) is not necessarily relevant in agri-food LCAs (e.g., comparing housed monogastric livestock systems with similar feed rations), in the current case (i.e., mixed crop-livestock systems), total emissions intensity reported as kg CO_2-eq/ha allows the expression of impacts from the viewpoint of local producers (Picasso et al. 2014). Values reported as emissions per ha were: 2,795, 2,734, 2,727 and 2,607 kg CO_2-eq/ha for CC, SR, LR and FR, respectively. These values included emissions arising from both crop and livestock areas of each farming system.

The minor differences observed among systems could be explained by the fact that the crop livestock systems, when they are analyzed, present trade-offs to reduce negative impacts of crop or livestock agriculture; namely, the most efficient livestock strategy (CC) in terms of low GHG emissions per kg of LWG is associated with a rotation with more intensive utilization of synthetic inputs and the highest intensity of land use without pastures in rotation (reflected by the lowest carbon restoration prediction as will be discussed in Section 3.3.). Conversely, the least efficient livestock strategy (finishing cattle in LR and FR) is associated with a rotation with fewer inputs and the inclusion of pastures in rotation, as described by Rovira et al. (2020). The values obtained in the current study are similar to others reported by Picasso et al. (2014), where values of emissions intensity per area were within the range of 2,000 and 2,500 kg CO_2-eq/ha, exclusively for livestock systems. The authors also identified a trend of decreasing emissions per ha when productivity per ha increased, a trend supported by Styles et al. (2018) in the context of dairy intensification, albeit with caveats such as displaced production which may reduce local emissions whilst increasing net emissions as the trading nation may be less environmentally efficient than domestic production.

Total emissions intensity per kg of product under economic allocation is provided in **Table 4**. In general, LWG leaving the farmgate provided 85 – 94%

of the total revenue in systems which rotated with crops, whereas it was 100% in FR. Total revenue from crops (soybean, oat, and wheat) was 15% in CC, 11.7% in SR and 6% in LR (see **Supplementary Material 3**). Economic prices of crops were equal among systems, as harvested crops were sold to the same industry, and the values changed only across years. However, livestock prices were different according to date of sale, animal category, and final liveweight. For perspective, kg CO_2-eq/US$ was: 2.4, 1.8, 1.7 and 1.5 under CC, SR, LR, and FR, respectively.

According to Pelletier et al. (2015), economic allocation is an effective way to reflect the hierarchy in systems where there are multiple co-products by appropriately assigning responsibility for the associated environmental burdens to the primary economic outputs, which largely aligns with the function of agricultural systems. In other words, economic allocation is generally the preferred approach for biological systems such as primary food production (and beyond, pending the system boundary). Recently, Kyttä et al. (2022) described economic allocation as an accurate method, particularly when utilized in livestock systems, as it reflects the reality that drives the production system as described above and discussed in more detail by Ardente and Cellura (2012). In the current study, LWG is the main product under the hierarchical logic proposed by Pelletier et al. (2015) and it was therefore targeted as the main product for deep interpretation.

Table 4. Emissions (kg CO_2-eq/kg of liveweight gain, soybean, oat, and wheat), using economic allocation for each pasture crop rotation at the Palo a Pique long term experiment, Treinta y Tres, Uruguay.

Product	Continuous Cropping	Short Rotation	Long Rotation	Forage Rotation
Liveweight	11.3	11.8	11.8	16.4
Soybean	1.36	1.24	1.01	-
Wheat	0.61	0.54	0.43	-
Oat	-	0.57	0.47	-

Emissions intensity reported as kg CO_2-eq/kg LWG, apart from FR, impacts were similar among systems, regardless of livestock strategy. Results are consistent with the aforementioned trade-offs between livestock strategies and crop rotation interactions. Results reported in terms of CO_2-eq/kg LWG were similar to others reported by Picasso et al. (2014) who investigated backgrounding and finishing systems in a review of different production systems in Uruguay with different feed rations (6.9 - 16.7 kg CO_2-eq/kg LWG); Dick et al. (2015) compared extensive versus intensive systems in Southern Brazil (9.2 – 22.5 CO_2-eq/kg LWG) and Ruviaro et al. (2015) for systems including cow-calf operation (18.3 – 42.6 CO_2-eq/kg LWG). Most comparable cases considered grazing animals with the inclusion of legumes, fertilization, and grazing management. On the other hand, results obtained by McAuliffe et al. (2018) showed values of 16-20 kg CO_2-eq/kg LWG in grazing systems in Southwest England for finishing cattle (steers and heifers equally split within three herds) with a housing period during winter under humid temperate conditions.

Crop production results obtained should be interpreted cautiously due to oat and wheat being highly affected by climatic conditions during the trial, as reported by Pereyra Goday et al. (2022). Shrestha et al. (2020) conducted an LCA of wheat rotations and evaluated different scenarios of allocation for wheat production. Their study showed similar values to our findings: 0.79 kg CO_2-eq/kg of wheat under economic allocation and 0.62 kg CO_2-eq/kg of wheat under mass allocation. The same authors conclude that rotations and diversification in crop production systems, combined with the necessity to understand synergies and trade-offs when evaluating environmental impacts of crops, require deeper exploration in the context of global GHG reduction ambitions (e.g., the Paris Climate Change Agreement). Bearing this in mind, our results could be considered as a novel evidence base for values of kg CO_2-eq/kg of wheat, soybean and oat, as management in the four trials reported herein is similar to management carried out on commercial farms in the study region (Rovira et al. 2020).

3.5.2. Intrasystem emissions

The process contribution per kg of LWG is presented in **Table 5**. The largest share of emissions per kg of LWG was derived from enteric fermentation, which presents 52-72% of total livestock GHG emissions. Similar values were reported by several authors, referring to grazing ruminants with direct deposition of manure in the field (de Figueiredo et al. 2017; Dick et al. 2015; Picasso et al. 2014). Conversely, the proportion of enteric fermentation's emissions were lower on a mass-based functional unit in systems that included manure management (Ogino et al. 2007; Weiss and Leip 2012).

The second largest source of emissions was direct N_2O from soils followed by fertilizer production, contributing between 8.4-14.2% and 8.4-16.1%, respectively. These results could be explained by the dependence of fertilizers in all crops. Naturally, Uruguayan soils are deficient in phosphorus (P) and nitrogen (N) (Madeira 2019), which drive the use of fertilizers in crops every year. For instance, FR showed higher N_2O emissions compared to the other systems, due to the use of 184 kg N/ha/year, as detailed in Table 3.

Table 5. Processes contribution (kg CO_2-eq/kg liveweight gain) for each pasture crop rotation at the Palo a Pique long term experiment, Treinta y Tres, Uruguay.

Process	Continuous Cropping	Short Rotation	Long Rotation	Forage Rotation
Enteric Fermentation (CH_4)	7.49	8.62	7.30	8.53
Direct Emissions (N_2O)	1.06	1.00	1.42	2.63
Indirect Emissions (N_2O)	0.32	0.24	0.33	1.37
Fertilizers use	1.34	0.99	1.59	2.32
Inputs (seeds, fuel, feed, pesticides)	0.40	0.25	0.39	0.17
Urea emissions	0.12	0.08	0.12	0.63
Manure Management (CH_4)	0.01	0.01	0.01	0.01
Manure Management (N_2O)	0.56	0.65	0.59	0.72
Total	11.3	11.8	11.8	16.4

3.5.3. Soil organic carbon inclusion

Given climate-focused actions to reduce emissions in line with achieving net-zero carbon economies (CIEL 2020); in order to consider the potential of mitigation through SOC sequestration, we included SOC stock change rates per year. Significant changes were obtained when SOC sequestration was included (Goglio et al. 2015). The mitigation potential through SOC sequestration during May 2019-April 2022 was 22.4%, 19.2% and 25.3% for CC, SR, and LR respectively. FR had the highest value of carbon uptake, perhaps unsurprisingly given the additional carbon inputs from feces and DM production, thus potentially off-setting emissions up to 42.1%. That being said, potential GHG off-setting as a percentage showed similar values among CC, SR and LR; as a result, it is important to consider the total emissions (i.e., farm-scale rather than per unit of produce) when considering local mitigation measures and/or technologies. This value was substantially lower in CC (201,214 kg CO_2-eq) compared with SR (229,622 kg CO_2-eq), LR (458,186 kg CO_2-eq) and FR (281,509 kg CO_2-eq). In terms of kg CO_2-eq mitigated through SOC sequestration, the best performance was obtained in FR and LR, which supports results presented by Teague et al. (2016).

Emissions intensity (kg CO_2-eq/kg product) considering economic allocation and SOC mitigation is presented in **Table 6**. Although extant literature recommends 10 years between two soil samples to estimate SOC variation (Goglio et al. 2015), in our study we used six years. This is arguably a limitation of our study; however, we assessed our assumption based on best practice by reporting emissions with (baseline results) and without SOC stock changes per year.

Table 6. Emissions (kg CO_2-eq/kg of liveweight gain, soybean, oat, and wheat), using economic allocation and soil organic carbon mitigation for each pasture crop rotation at the Palo a Pique long term experiment, Treinta y Tres, Uruguay.

Product	Continuous Cropping	Short Rotation	Long Rotation	Forage Rotation
Liveweight	8.77	9.44	8.81	9.45
Soybean	1.06	1.00	0.76	-
Wheat	0.47	0.44	0.32	-
Oat	-	0.46	0.35	-

SOC sequestration in LCA is difficult to quantify accurately as historical land use is often unknown and carbon-stock changes are frequently excluded from system boundaries (Goglio et al. 2015). Results obtained can be different depending on scale (i.e., farm, plot, life cycle), with a greater geographical scope generally associated with greater heterogeneity in soil properties and thus the soil's potential as a carbon sink (Soussana et al. 2010). In this regard, results could show a wide variability according to soil quality and structure, farm type, climate (and microclimates), as well as farm management, as explained by Lal (2004). Results presented here show the potential mitigation of GHG at the farm level but cannot be extrapolated to broader geographic regions. Picasso et al. (2014) estimated mitigation in GHG emissions through SOC sequestration of 17% in livestock systems in Uruguay, whereas Dollé et al. (2011), reported mitigation of emissions in livestock systems in France of 24% and 53% including SOC sequestration. It should be noted however that SOC stock changes across studies are typically incomparable due to different methodological options for the calculation of soil carbon uptake as demonstrated by Mogensen et al. (2014).

3.5.4. Methodological comparisons

Figure 3 demonstrates estimations of uncertainty through Monte Carlo simulations, considering SOC mitigation, with a confidence interval of 95%. Differences detected across systems were not significant when calculated under pairwise permutations. Interestingly, and supportive of the earlier discussion of SOC stock change uncertainty, pairwise calculations to test the inclusion versus the exclusion of SOC were significant in all comparisons ($p < 0.05$). However, readers should be aware that the uncertainty results are dependent on numerous factors as outlined above which are difficult to capture using Monte Carlo analysis, and therefore we are simply reporting our findings rather than claiming SOC can truly reduce emissions by nearly half in all systems.

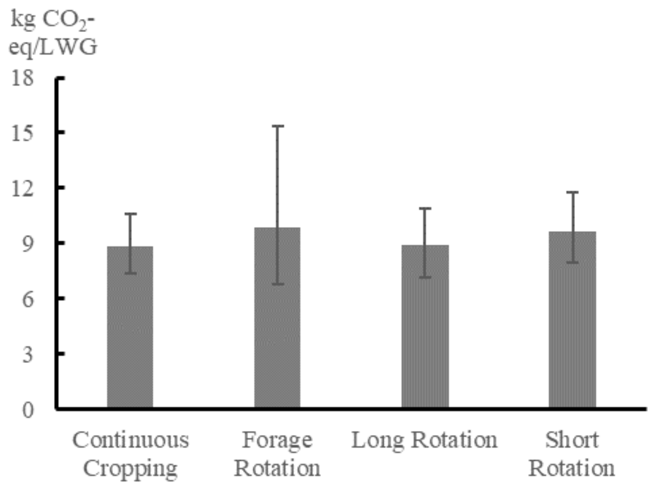


Figure 3. Results of Monte Carlo simulations for each pasture crop rotation at Palo a Pique long term experiment, Treinta y Tres, Uruguay. Error bars represent 95% confidence intervals. LWG: liveweight gain; CO_2-eq: CO_2 equivalent.

3.5.5. <u>Sensitivty analysis</u>

3.5.5.1. Mass allocation

Results for our sensitivity analysis to allocation methods showed differences in allocation of emissions to each product in each system. In crop production, variations in emissions' distributions were detected. For the three systems on average, net emissions increased by 87-95%, 71-79% and 82-88% for wheat, soybean, and oats, respectively, when they were analyzed considering mass allocation compared to economic allocation. Emissions for liveweight decreased considerably when mass allocation was tested, being 59%, 57% and 14% lower for CC, SR and LR, respectively. Conversely to economic allocation which is predicated on the value of a given product either at a point in time or as a rolling average, mass allocation does not consider value which could arguably reflect the quality of throughput (Kyttä et al. 2022). Within our study, this is a central aspect as mixed pasture-crop rotations evaluated herein produced different crops (soybean, wheat, and oat) and LWG (meat), which have different nutrient profiles: e.g., protein content, structure (e.g., limiting amino acids and anti-nutritional factors such as phytates in crops), and ultimately, digestibility (McAuliffe et al. 2023). The diverging results of the different allocation procedures underline the importance of avoiding arbitrary choices and selecting an appropriate method that fits the objectives of the analysis (Michiels et al. 2021). An alternative approach could consider nutrient density content of outputs in each system to evaluate environmental impacts from a nutritional standpoint as performed by Lee et al. (2021) for suckler beef. However, this would require substantial nutritional analysis of all outputs of the system, which was not possible in the current study based on primary data.

3.5.5.2. Global temperature change potential

As discussed in Section 2, a sensitivity analysis was carried out to test differences between GTP_{100} and GWP_{100}, or the climate change impact as quantified by the predicted change in radiative forcing and relative temperature, respectively, over a 100-year period, considering the significant emissions of biogenic methane originating from enteric fermentation in ruminants, and the ongoing debate regarding GWP_{100} due to its potential tendency to overestimate methane's impacts (McAuliffe et al 2023b). Values to convert CH_4 to CO_2-eq were 5.38 and 27.9, whereas values to convert N_2O were 233 and 273 under GTP_{100} and GWP_{100} for both gases, respectively. As explained by Reisinger and Ledgard (2013), alternative impact assessments such as GTP_{100} significantly change the balance between CH_4 and N_2O and could change the overall cost and associated profitability for farmers if a price was applied to agricultural emissions (e.g., carbon credits). Understandably given the drastically different CH_4 characterization factors, enteric CH_4 is the most affected GHG under contrasting metrics. As a result, the relative proportion of CH_4 emissions in the overall emissions intensity was reduced by: 27.6%, 34.6%, 24.2% and 17.5% in CC, LR, SR and FR, respectively. The relative contribution of N_2O to the overall emissions intensity increased on average by 11% under GTP_{100} compared to GWP_{100}. Total emissions intensities per kg of LWG were reduced by: 4.7, 4.3, 5.2, and 8.5 kg CO_2-eq/kg LWG in CC, LR, SR, and FR, respectively (i.e., emissions intensities as kg CO_2-eq/kg LWG using GTP were 58.4%, 63.5%, 55.9% and 48.2% lower than emissions using GWP_{100} applied to CC, SR, LR, and FR). For a country like Uruguay, where the agricultural sector represents the most important source of GHG emissions, the use of alternative metrics could elucidate novel GHG emission mitigation and/or off-setting strategies by demonstrating that gaseous emissions other than CH_4 also require urgent abatement attention (N_2O in the case of agriculture; Takahashi et al. 2019). For example, agriculture's contribution to national emissions reduced from 73% of total GHGs under GWP_{100} to 55% under GTP_{100} (SNRCC – MA 2021).

However, it is important to note that there is no 'right' metric: an impact assessment should be chosen to answer a specific research question (e.g., if individual gaseous temporal changes are a study's focus, then GWP* may be the most appropriate metric; Cain et al. 2019). Although climate change impact assessments are gaining more attention (Allen et al. 2022), this is not a new issue for carbon footprints and other sustainability assessments focusing on GHG emissions. As discussed by Reisinger and Ledgard (2013), the quantification of emissions is important, but moreover, the authors conclude that different impact assessments answer different questions. Finally on the topic of LCIA, it is important to acknowledge that IPCC (2021) recommend testing GHG emissions calculations and associated impact assessments through sensitivity analyses (either cross-time horizons or cross-impact assessments, pending the research question and system context). This invariably makes interpretation and communication more challenging, but it is a critical exercise to demonstrate to stakeholders, policymakers, and consumers that there are considerable complexities involved in assessing a product or service's contribution to climate change.

3.5.6. Implications for mixed pasture crop rotations

Herein we address the GHG emissions intensities associated with mixed pasture crop rotation, underpinned by temporally high-resolution agronomical data. Although the systems evaluated have different livestock production strategies, we can draw useful conclusions in terms of general responses to various management practices.

First, we note the importance of including pastures in rotational farming and the potential of pastoral swards to reduce input use (e.g., synthetic inorganic N fertilizers) through an increase in biological N fixation (Carswell et al. 2022; McAuliffe et al., 2018), potentially improved SOC sequestration in certain production systems (Pravia et al. 2019), and greater productivity through the use of high-quality, managed pastures (Szymczak et al., 2023). Although this is a geographically restricted study (i.e., conclusions cannot be made from a global perspective), such management practices appear to achieve high average daily growth and improved biomass productivity in temperate climates, thereby reducing emissions per kg produced which has been demonstrated previously (Carvalho et al. 2018; de Souza Filho et al. 2019). Enhancing productivity has the potential to yield improvements in the economic strength of crop-livestock rotational systems (Leahy et al., 2020). Secondly, the exclusive utilization of food produced within the system for animal feed facilitates emission reductions (e.g., from transportation) and enhances the utilization of crop residues, concurrently integrating nutrients into the soil through the deposition of manure and urine. Nevertheless, achieving an optimal balance between crops and livestock within these systems poses several challenges associated with land utilization and nutrient use efficiency (Xu et al., 2023).

On the other hand, data sourced from long-term, large-scale trials enables us to explore potential risks and benefits of agricultural systems currently underrepresented in extant LCA literature. This is particularly helpful to the LCA evidence base as our work predominantly adopts temporally high-resolution primary data, thereby better informing those who consume, and indeed produce, the four included (co)products. Finally, pasture crop rotations have the potential to produce ecosystem services (e.g., reduce erosion, pollination and biological control of pests) and reduce environmental impacts without compromising economic sustainability. This presents an opportunity for future research to explore broader sustainability trade-offs including different impact categories such as eutrophication, water scarcity and fossil fuel depletion, as well as a more holistic viewpoint considering biodiversity, economic (e.g., projected changes in supply and demand for Uruguay's primary agricultural exports) and social (e.g., human and animal welfare) ramifications of different multifunctional systems. Considering the importance of meat towards the nation's total income, it is of critical importance to explore the 'steps to sustainable livestock' (Eisler et al. 2014; Rivero et al. 2021) from as many lenses as possible to ensure land use is optimized and consumers are provided with transparent and unbiased information which acknowledges weaknesses in modelling exercises via uncertainty analyses and testing the sensitivity of subjective decision making (e.g., allocation and impact assessment characterization factors, the role of livestock in a circular bio-economy (utilizing 'waste' streams and land not suited-to or in combination with crops).

3.6. Conclusions

Our findings present a novel evidence base simultaneously tackling environmental modelling issues in mixed crop-livestock systems, whilst providing insights into locally representative and understudied farming practices in South America to produce food and feed. These practices, allow to reduce environmental degradation, creating semi-circular multifunctional farming systems which feed both human and animals simultaneously. The underlying data also explore a variety of material inputs and outputs, as well as flows to and from nature, differing across land management trials thus further elucidating optimal local practices which may realize sustainable solutions. Given there are few Life Cycle Assessments studies available on such mixed-enterprise 'semi-circular' systems, particularly with novel primary data, this study adds critical knowledge to agri-food related sustainability literature by addressing environmental issues in complex production systems compared to extant and broad coverage of mono enterprise systems.

We emphasize the significance of our findings in light of the widespread use of these production systems in South America and the lack of information regarding their environmental impacts. Furthermore, methodological approaches to assess said impacts of such complex, multi-produce farming systems is scant in extant literature, and as such, we propose a robust framework to inform relevant stakeholders about uncertainties, some of which are substantial, when conducting carbon footprints. Whilst a broader assessment of impact categories (e.g., eutrophication, acidification, fossil depletion, water scarcity, etc.) is required to fully reveal the benefits and risks associated with mixed crop and livestock systems, our study contributes to improving geographical coverage of LCA data in the context of a growing demand for information concerning production systems including mixed crop-livestock enterprises.

Acknowledgments

We thank INIA's field and laboratory staff and students who participated in the collection and processing of data. INIA, Rothamsted Research, University of Bristol and Harper Adams University are all members of the Global Farm Platform initiative (http://www.globalfarmplatform.org) collaboratively working towards sustainable ruminant livestock production systems.

Funding

This research was funded by: Long Term Experimental Platforms project (INIA), F.P-G was funded by posgraduate fellowship: National Institute of Agricultural Research doctoral fellowship and National Agency of Research and Innovation (ANII) code MOV_CA_2021_1_171482 (mobility fellowship). Rothamsted Research receives strategic funding from the Biotechnological and Biological Sciences Research Council (BBSRC) of the United Kingdom. G.A.M. and M.J.R. were funded by Soil to Nutrition (S2N), Rothamsted Research's Institute Strategic Programme supported by UK Research and Innovation (UKRI) and BBSRC (BBS/E/C/000I0320 and BBS/E/C/000I0330). G.A.M. also acknowledges funding from BBSRC's new Strategic Programme 'Resilient Farming Futures' (BB/X010961/1). M.J.R. also acknowledges funding from BBSRC's new Strategic Programme Growing Health (BB/X010953/1). A.J. was funded by BBSRC's Science Initiative Catalyst Award (SICA).

Data availability. The integrity of the data used in this study are included in it and in the Supplementary Material.

Code availability. Not applicable.

Authors' contributions. Conceptualization: FPG, GAM, AJ, PR, WA, MJR; methodology: FPG, GAM, AJ, TT; data analysis: FPG, GAM, AJ; writing original draft, FPG, GAM, AJ; writing-review and editing: FP-G, GAM, AJ, TT, PR, WA, MRFL, MJR.

Declarations

Ethical approval. This study is exempt from ethics approval since livestock were submitted to common farming practices and not regulated procedure was carried out.

Consent to participate. All the research participants gave their informed consent to participate in this study.

Consent for publication. The authors confirm that all the participants gave their informed consent to participate in the study.

Conflict of interests. The authors declare no competing interests.

References

Allen MR, Friedlingstein P, Girardin CAJ, Jenkins S, Malhi Y, Mitchell-Larson E, Peters GP, Rajamani L (2022) Net Zero: Science, Origins, and Implications. Annu Rev Env Resour 47: 849–887. https://doi.org/10.1146/annurev-environ-112320-105050

AOAC (1990) Official Methods of Analysis, 15th ed. Association of Official Analytical Chemist, Washington DC, USA.

Ardente F, Cellura M (2012) Economic Allocation in Life Cycle Assessment. J Ind Ecol 16: 387–398. https://doi.org/10.1111/j.1530-9290.2011.00434.x

Asem-Hiablie S, Battagliese T, Stackhouse-Lawson KR, Alan Rotz C (2019) A life cycle assessment of the environmental impacts of a beef system in the USA. Int J Life Cycle Ass 24, 441–455. https://doi.org/10.1007/s11367-018-1464-6

BSI (2011) PAS 2050:2011: Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. British Standard Institution (BSI), London, UK. ISBN 978 0 580 71382 8

Cain M, Lynch J, Allen MR, Fuglestvedt JS, Frame DJ, Macey AH (2019) Improved calculation of warming-equivalent emissions for short-lived climate pollutants. NPJ Clim Atmos Sci 2: 29. https://doi.org/10.1038/s41612-019-0086-4

Carswell A, Sánchez-Rodríguez A R, Saunders K, le Cocq K, Shaw R, Cotton J, Zhang Y, Evans J, Chadwick D R, Jones D L, Misselbrook T (2022). Combining targeted grass traits with red clover improves grassland performance and reduces need for nitrogen fertilisation. Eur J Agron, 133: 126433. https://doi.org/10.1016/j.eja.2021.126433

Carvalho PC de F, Peterson CA, Nunes PA de A, Martins AP, de Souza Filho W, Bertolazi VT, Kunrath TR, de Moraes A, Anghinoni I (2018) Animal production and soil characteristics from integrated crop-livestock systems: toward sustainable intensification. J Anim Sci 96: 3513–3525. https://doi.org/10.1093/jas/sky085

CIEL (2020) NET ZERO CARBON & UK LIVESTOCK. York: Centre for Innovation Excellence in Livestock. https://cielivestock.co.uk/expertise/net-zero-carbon-uk-livestock/report-october-2020/ (accessed: 20/05/2023)

de Faccio Carvalho PC, Savian JV, della Chiesa T, de Souza Filho W, Terra JA, Pinto P, Martins AP, Villarino S, da Trindade JK, de Albuquerque Nunes PA, Pineiro G (2021) Land-Use Intensification Trends In The Rio De La Plata Region Of South America: Toward Specialization Or Recoupling Crop And Livestock Production. Front Agric Sci Eng https://doi.org/10.15302/J-FASE-2020380

de Figueiredo EB, Jayasundara S, de Oliveira Bordonal R, Berchielli TT, Reis RA, Wagner-Riddle C, La Scala Jr N (2017) Greenhouse gas balance and carbon footprint of beef cattle in three contrasting pasture-management systems in Brazil. J Clean Prod 142: 420–431. https://doi.org/10.1016/j.jclepro.2016.03.132

de Souza Filho W, Nunes PA de A, Barro RS, Kunrath TR, de Almeida GM, Genro TCM, Bayer C, de Faccio Carvalho PC (2019) Mitigation of enteric methane emissions through pasture management in integrated crop-livestock systems: Trade-offs between animal performance and environmental impacts. J Clean Prod 213: 968–975. https://doi.org/10.1016/j.jclepro.2018.12.245

de Vries M, van Middelaar CE, de Boer IJM (2015) Comparing environmental impacts of beef production systems: A review of life cycle assessments. Livest Sci 178: 279–288. https://doi.org/10.1016/j.livsci.2015.06.020

de Vries M, de Boer IJM (2010) Comparing environmental impacts for livestock products: A review of life cycle assessments. Livest Sci. https://doi.org/10.1016/j.livsci.2009.11.007

Dick M, Abreu da Silva M, Dewes H (2015) Life cycle assessment of beef cattle production in two typical grassland systems of southern Brazil. J Clean Prod 96: 426–434. https://doi.org/10.1016/j.jclepro.2014.01.080

DIEA - MGAP (2022) Anuario Estadístico Agropecuario. https://www.gub.uy/ministerio-ganaderia-agricultura-pesca/comunicacion/publicaciones/anuario-estadistico-agropecuario-2022 (accesed 27 April 2023).

Dollé JB, Agabriel J, Peyraud JL, Faverdin P, Manneville V, Raison C, Gac A, le Gall A (2011) Les gaz à effet de serre en élevage bovin: évaluation et leviers d’action. INRAE Productions Animales 24: 415–432. https://doi.org/10.20870/productions-animales.2011.24.5.3275

Eisler MC, Lee MRF, Tarlton JF, Martin GB, Beddington J, Dungait JAJ, Greathead H, Liu J, Mathew S, Miller H, Misselbrook T, Murray P, Vinod VK, Van Saun R, Winter M (2014) Agriculture: Steps to sustainable livestock. Nature 507: 32–34. https://doi.org/10.1038/507032a

FAO (2023) Methane emissions in livestock and rice systems – Sources, quantification, mitigation and metrics. Rome. https://doi.org/10.4060/cc7607en

García-Préchac F, Ernst O, Siri-Prieto G, Terra JA (2004) Integrating no-till into crop–pasture rotations in Uruguay. Soil Tillage Res 77, 1–13. https://doi.org/10.1016/j.still.2003.12.002

Guinée JB, Heijungs R, Huppes G (2004). Economic allocation: Examples and derived decision tree. Int J Life Cycle Assess 9(1): 23. https://doi.org/10.1007/BF02978533

Goglio P, Smith WN, Grant BB, Desjardins RL, McConkey BG, Campbell CA, Nemecek T (2015) Accounting for soil carbon changes in agricultural life cycle assessment (LCA): a review. J Clean Prod 104: 23–39. https://doi.org/10.1016/j.jclepro.2015.05.040

INIA (2022) Factores de emisión y coeficientes para estudios de Huella de Carbono en Uruguay: Sector Ganadero. Instituto Nacional de Investigación Agropecuaria (INIA), Montevideo, Uruguay. ISBN 978 9974 38 474 3

IPCC (2021) Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge, UK. https://doi.org/10.1017/9781009157896

IPCC (2019a) Climate Change and Land: an IPCC Special Report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems. Intergovernmental Panel on Climate Change, Geneva, Switzerland. https://www.ipcc.ch/srccl/ (accessed 19 June 2024)

IPCC (2019b) 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Intergovernmental Panel on Climate Change, Geneva, Switzerland. ISBN 978-4-88788-232-4

ISO (2006) ISO 14040: Environmental management - Life cycle assessment - Principles and framework. International Organization for Standardisation, Geneva, Switzerland. https://www.iso.org/obp/ui/#iso:sTd:iSo:14040:eD-2:v1:en (accesed 19 June 2024)

Kyttä V, Roitto M, Astaptsev A, Saarinen M, Tuomisto HL (2022) Review and expert survey of allocation methods used in life cycle assessment of milk and beef. Int J Life Cycle Assess 27: 191–204. https://doi.org/10.1007/s11367-021-02019-4

Lal R (2004) Soil carbon sequestration to mitigate climate change. Geoderma. https://doi.org/10.1016/j.geoderma.2004.01.032

Leahy S, Clark H, Reisinger A (2020) Challenges and Prospects for Agricultural Greenhouse Gas Mitigation Pathways Consistent with the Paris Agreement. Front Sustain Food S 4. https://doi.org/10.3389/fsufs.2020.00069

Lee MRF, McAuliffe GA, Tweed JKS, et al (2021) Nutritional value of suckler beef from temperate pasture systems. Animal 15:100257. https://doi.org/10.1016/j.animal.2021.100257

Lynch J (2019) Availability of disaggregated greenhouse gas emissions from beef cattle production: A systematic review. Environ Impact Asses 76: 69–78. https://doi.org/10.1016/j.eiar.2019.02.003

Lynch PB (1947) Methods of measuring the production from grasslands. A review of the techniques employed by the Fields Division, Department of Agriculture. New Zealand Journal of Science and Technology Section A: 385–405. https://doi.org/10.33584/jnzg.1951.13.957

Madeira W (2019) Efectos de la fertilización primavero-estival nitrógeno-fosfatada y del riego suplementario en la productividad y eficiencia de uso de nutrientes del campo natural. Universidad de la República, Montevideo, Uruguay. https://www.colibri.udelar.edu.uy/jspui/bitstream/20.500.12008/29801/1/MadeiradeQuadrosWilliam.pdf (accesed 19 June 2024)

Manzano P, Rowntree J, Thompson L, del Prado A, Ederer P, Windisch W, Lee MRF (2023). Challenges for the balanced attribution of livestock's environmental impacts: the art of conveying simple messages around complex realities. Anim Front. https://doi.org/10.1093/af/vfac096

McAuliffe GA, Takahashi T, Beal T, Huppertz T, Leroy F, Buttriss J, Collins AL, Drewnowski A, McLaren SJ, Ortenzi F, van der Pols JC, van Vliet S, Lee MRF

(2023a) Protein quality as a complementary functional unit in life cycle assessment (LCA). Int J Life Cycle Assess 28: 146–155. https://doi.org/10.1007/s11367-022-02123-z

McAuliffe GA, Lynch J, Cain M, Buckingham S, Rees RM, Collins AL, Allen M, Pierrehumbert R, Lee MRF, Takahashi T (2023b). Are single global warming potential impact assessments adequate for carbon footprints of agri-food systems? Environ Res Lett 18: 084014. https://doi.org/10.1088/1748-9326/ace204

McAuliffe GA, Takahashi T, Lee MRF (2020) Applications of nutritional functional units in commodity-level life cycle assessment (LCA) of agri-food systems. Int J Life Cycle Assess 25: 208–221. https://doi.org/10.1007/s11367-019-01679-7

McAuliffe GA, Takahashi T, Mogensen L, Hermansen JE, Sage CL, Chapman DV, Lee MRF (2017) Environmental trade-offs of pig production systems under varied operational efficiencies. J Clean Prod 165: 1163–1173. https://doi.org/10.1016/j.jclepro.2017.07.191

McAuliffe GA, Takahashi T, Orr RJ, Harris P, Lee MRF (2018) Distributions of emissions intensity for individual beef cattle reared on pasture-based production systems. J Clean Prod 171: 1672–1680. https://doi.org/10.1016/j.jclepro.2017.10.113

Michiels F, Hubo L, Geeraerd A (2021) Why mass allocation with representative allocation factor is preferential in LCA when using residual livestock products as organic fertilizers. J Environ Manage 297: 113337. https://doi.org/10.1016/j.jenvman.2021.113337

Mogensen L, Kristensen T, Nguyen TLT, Knudsen MT, Hermansen JE (2014) Method for calculating carbon footprint of cattle feeds – including contribution from soil carbon changes and use of cattle manure. J Clean Prod 73: 40–51. https://doi.org/10.1016/j.jclepro.2014.02.023

Mustafa Z, Vitali G, Huffaker R, Canavari M (2023) A systematic review on price volatility in agriculture. J Econ Surv. https://doi.org/10.1111/joes.12549

OECD/FAO (2022) OECD-FAO Agricultural Outlook 2022-2031. OECD Publishing, Paris, France. https://doi.org/10.1787/f1b0b29c-en

Ogino A, Orito H, Shimada K, Hirooka H (2007) Evaluating environmental impacts of the Japanese beef cow?calf system by the life cycle assessment method. Anim Sci J 78: 424–432. https://doi.org/10.1111/j.1740-0929.2007.00457.x

Pelletier N, Ardente F, Brandão M, De Camillis C, Pennington D (2015). Rationales for and limitations of preferred solutions for multi-functionality problems in LCA: is increased consistency possible? Int J Life Cycle Assess 20: 74–86. https://doi.org/10.1007/s11367-014-0812-4

Pereyra Goday F, Rovira P, Ayala W, Rivero MJ (2022) Management and Productivity of Key Integrated Crop–Livestock Systems in Uruguay: The Palo a Pique Long-Term Experiment's Third Phase. Agronomy 12: 3023. https://doi.org/10.3390/agronomy12123023

Picasso VD, Modernel PD, Becoña G, Salvo L, Gutiérrez L, Astigarraga L (2014) Sustainability of meat production beyond carbon footprint: A synthesis of case studies from grazing systems in Uruguay. Meat Sci 98: 346–354. https://doi.org/10.1016/j.meatsci.2014.07.005

Pravia MV, Kemanian AR, Terra JA, Shi Y, Macedo I, Goslee S (2019) Soil carbon saturation, productivity, and carbon and nitrogen cycling in crop-pasture rotations. Agr Syst 171: 13–22. https://doi.org/10.1016/j.agsy.2018.11.001

Pré Consultants (2022) SimaPro. Amersfoort: Pré Consultants.

Rawtani D, Gupta G, Khatri N, Rao PK, Hussain CM (2022) Environmental damages due to war in Ukraine: A perspective. Sci Total Environ 850: 157932. https://doi.org/10.1016/j.scitotenv.2022.157932

Reisinger A, Ledgard S (2013) Impact of greenhouse gas metrics on the quantification of agricultural emissions and farm-scale mitigation strategies: a New Zealand case study. Environ Res Lett 8: 025019. https://doi.org/10.1088/1748-9326/8/2/025019

Rice P, O'Brien D, Shalloo L, Holden NM (2017) Evaluation of allocation methods for calculation of carbon footprint of grass-based dairy production. J Environ Manage, 202: 311–319. https://doi.org/10.1016/j.jenvman.2017.06.071

Rivero MJ, Evans ACO, Berndt A, Cartmill A, Dowsey A, Farruggia A, Mignolet C, Enriquez-Hidalgo D, Chadwick D, McCracken DI, Busch D, Pereyra F, Martin GB, Sanford GR, Sheridan H, Wright I, Brunet L, Eisler MC, Lopez-Villalobos N, Rovira P, Harris P, Murphy P, Williams AP, Jackson RD, Machado R, Suraj PT, Puech T, Boland TM, Ayala W, Lee MRF (2021) Taking the steps toward sustainable livestock: our multidisciplinary global farm platform journey. Anim Front 11: 52–58. https://doi.org/10.1093/af/vfab048

Rovira P, Ayala W, Terra J, García-Préchac F, Harris P, Lee MRF, Rivero MJ (2020) The 'Palo a Pique' long-term research platform: First 25 years of a crop–livestock experiment in Uruguay. Agronomy 10. https://doi.org/10.3390/agronomy10030441

Ruviaro CF, de Léis CM, Lampert V do N, Barcellos JOJ, Dewes H (2015) Carbon footprint in different beef production systems on a southern Brazilian farm: a case study. J Clean Prod 96: 435–443. https://doi.org/10.1016/j.jclepro.2014.01.037

Segura C, Horrocks C, Lopez-Aizpun M, Blackwell MSA, Darch T, Hood J, Le Cocq K, McAuliffe GA, Lee MRF, Cardenas L (2023). Response of soil health indicators to dung, urine and mineral fertilizer application in temperate pastures. J Environ Manage 330: 117096. https://doi.org/10.1016/j.jenvman.2022.117096

Shrestha P, Karim R A, Sieverding HL, Archer DW, Kumar S, Nleya T, Graham CJ, Stone JJ (2020). Life cycle assessment of wheat production and wheat-based crop rotations. Journal Environ Qual 49: 1515–1529.

SNRCC - MA (2021). Inventario Nacional de Gases de Efecto Invernadero (INGEI) 1990-2019. https://www.gub.uy/ministerio-ambiente/politicas-y-gestion/inventarios-nacionales-gases-efecto-invernadero-ingei (accesed 27 April 2023).

Soteriades AD, Foskolos A, Styles D, Gibbons JM (2019) Diversification not specialization reduces global and local environmental burdens from livestock production. Environ Int 132: 104837. https://doi.org/10.1016/j.envint.2019.05.031

Soussana JF, Tallec T, Blanfort V (2010) Mitigating the greenhouse gas balance of ruminant production systems through carbon sequestration in grasslands. Animal 4: 334–350. https://doi.org/10.1017/S1751731109990784

Styles D, Gonzalez-Mejia A, Moorby J, Foskolos A, Gibbons J (2018) Climate mitigation by dairy intensification depends on intensive use of spared grassland. Global Change Biol 24: 681e693. https://doi.org/10.1111/gcb.13868.

Szymczak, L. S., de Moraes, A., Sulc, R. M., Barker, D., Monteiro, A. L. G., Lang, C. R., Moraes, R. F., Lemaire, G., & de Faccio Carvalho, P. C. (2023). Convergence points of optimal herbage accumulation and intake rate by sheep grazing tall fescue. Grass Forage Sci, 78(4), 578–589. https://doi.org/10.1111/gfs.12630

Takahashi T, McAuliffe GA, Lee MRF (2019) Assessing the environmental impact of ruminant production systems. pp. 121–138. https://doi.org/10.19103/as.2018.0044.14

Teague WR, Apfelbaum S, Lal R, Kreuter UP, Rowntree J, Davies CA, Conser R, Rasmussen M, Hatfield J, Wang T, Wang F, Byck P (2016) The role of ruminants in reducing agriculture's carbon footprint in North America. J Soil Water Conserv 71: 156–164. https://doi.org/10.2489/jswc.71.2.156

Uruguay XXI (2023) Informe Anual Comercio Exterior. https://www.uruguayxxi.gub.uy/es/centro-informacion/articulo/informe-anual-de-comercio-exterior-de-uruguay-2023/ (accessed 13 March 2024).

Weiss F, Leip A (2012) Greenhouse gas emissions from the EU livestock sector: A life cycle assessment carried out with the CAPRI model. Agric Ecosyst Environ 149: 124–134. https://doi.org/10.1016/j.agee.2011.12.015

Wernet G, Bauer C, Steubing B, Reinhard J, Moreno-Ruiz E, Weidema B (2016) The ecoinvent database version 3 (part I): overview and methodology. Int J Life Cycle Assess 21: 1218–1230. https://doi.org/10.1007/s11367-016-1087-8

Wright AF, Bailey JS (2001) Organic carbon, total carbon, and total nitrogen determinations in soils of variable calcium carbonate contents using a Leco CN-2000 dry combustion analyzer. Commun Soil Sci Plant Anal 32: 3243–3258. https://doi.org/10.1081/CSS-120001118

Xu X, Xu Y, Li J, Lu Y, Jenkins A, Ferrier RC, Li H, Stenseth NC, Hessen DO, Zhang L, Li C, Gu B, Jin S, Sun M, Ouyang Z, Mathijs E (2023). Coupling of crop and livestock production can reduce the agricultural GHG emission from smallholder farms. iScience 26: 106798. https://doi.org/10.1016/j.isci.2023.106798

Zira S, Rydhmer L, Ivarsson E, Hoffmann R, Röös E (2021) A life cycle sustainability assessment of organic and conventional pork supply chains in Sweden. Sustai Prod and Consump, 28: 21–38. https://doi.org/10.1016/j.spc.2021.03.028

4. Nitrogen use efficiency of integrated crop-livestock systems at different levels of intensification

Pereyra Goday, F.[1, *,3]; Castillo, J.[1]; Rovira, P.[1]; Ayala, W.[1]; Lee M.R.F.[2] Rivero, M.J.[3]

1Instituto Nacional de Investigación Agropecuaria (INIA), Treinta y Tres, 33000, Uruguay

2Harper Adams University, Edgmond, Shropshire, TF10 8NB, UK

3 Net Zero and Resilient Farming, Rothamsted Research, North Wyke, Okehampton, Devon, EX20 2SB, UK

*Corresponding author: fpereyragoday@gmail.com

4.1. Resumen

El desarrollo de métricas que evalúen la sostenibilidad de los sistemas de producción de alimentos se ha convertido en una herramienta importante para la realización de sistemas agroalimentarios globales sostenibles. La eficiencia en el uso del nitrógeno (N) (salidas de N en alimentos en relación con todas las entradas de N; NUE) es un indicador clave de eficiencia utilizado en sistemas de cultivos y ganadería. El objetivo de este estudio fue cuantificar y comparar la NUE, el excedente de N (NSURP, todas las entradas de N menos las salidas de N en productos alimenticios) y el balance de N/retención (todas las entradas menos las salidas de N y las pérdidas de N) de cuatro rotaciones de cultivo-pasturas con diferente intensidad de uso de suelo, a nivel de componente (cultivos y ganadería) y sistema, durante un período de 3 años (2019-2022). Los sistemas evaluados fueron: cultivo continuo (CC), rotación corta (SR): CC de 2 años (2a) más 2a de pasturas, rotación larga (LR): un CC de 2a más 4a de pasturas, y rotación forrajera (FR), pastura continua con festuca. La información utilizada en este estudio fue recolectada como datos primarios a nivel de campo. Los valores de NUE para cultivos fueron: 62.5%, 83.8% y 77.5% para CC, SR y LR, respectivamente. La NUE para ganadería

[3] Facultad de Agronomía, Universidad de la República.

fue de 24.4%, 9.9%, 14.3% y 5.5% para CC, SR, LR y FR, respectivamente. La NUE del sistema fue de 43.4%, 28.1%, 29.3% y 5.5% para CC, SR, LR y FR, respectivamente. El balance de N en el sistema fue de 4.9±1.52, 41±3.1, 29±2.2 y 64±3.5 para CC, SR, LR y FR, lo que sugiere mayores cantidades de N retenidas en el suelo en aquellos sistemas que incluyen pasturas. Nuestros hallazgos muestran diferencias entre componentes y sugieren la necesidad de aplicar estrategias diferenciales para optimizar los resultados obtenidos y mover la NUE hacia un lugar de 'operación segura', principalmente en el componente gaandero, considerando la relevancia de desarrollar sistemas de producción de alimentos sostenibles.

4.2. Summary

Development of metrics which evaluate sustainability of food production systems has become an important tool towards the realization of sustainable global agri-food systems. Nitrogen (N) use efficiency (NUE, food N outputs relative to all N inputs) is a key indicator of efficiency used in crop and livestock systems. The aim of this study was to quantify and compare the NUE, N surplus (NSURP, all N inputs minus N outputs in food products) and N balance/retained (all inputs minus N outputs and N losses) of four pasture crop rotations at different intensity of soil use, at the component (crop and livestock) and system level for a period of 3 years (2019-2022). The systems evaluated were: continuous cropping (CC), short rotation (SR)- a 2 year (2-y) CC plus 2-y pasture, long rotation (LR) - a 2-y CC plus 4-y pasture, and forage rotation (FR), a continuous pasture with tall fescue. The information used in this study was predominately collected as primary data at the field level. Crop NUE values were: 62.5, 83.8 and 77.5% for CC, SR and LR, respectively. Livestock NUE was 24.4, 9.9, 14.3 and 5.5% for CC, SR, LR and FR, respectively. System NUE was 43.4, 28.1, 29.3 and 5.5% for CC, SR, LR and FR respectively. N balance at system level was 4.9±1.52, 41±3.1, 29±2.2 and 64±3.5 to CC, SR, LR and FR, which suggest greater amounts of nitrogen retained in soil in those systems that include pastures. Our findings show differences between components and suggest the need to apply differential strategies to optimize the results obtained and move NUE towards to ‘safe operation’ place, mainly in the livestock component, considering the relevance of developing sustainable food production systems.

Keywords: sustainability; food security; crop-pasture systems; nitrogen use efficiency

4.3. Introduction

Food production systems around the world present several challenges associated with nutrient provision and food security. Global demand for food is expecting to grow between 1.1 and 1.5% per year in the next decade (OECD/FAO, 2022), explained by growing global population and economic development. At the same time, variability of international prices of material input commodities driven by global shocks including the war in Ukraine (Rawtani et al., 2022) is also predicted. On the other hand, there is an increased pressure on land and water resources (Spiertz, 2010), and growing concerns about the impact of modern agriculture on natural resources and how production processes are implemented (Xue et al., 2010).

Sustainable intensification (SI) of agriculture has the potential to enable meeting demands for food, while simultaneously meeting environmental and ecological goals (Haughey et al., 2023; Soria-Lopez et al., 2023). The aim of SI is the transformation of the whole food chain into a fully sustainable procedure, developing practices to exploit natural resources whilst reducing the harms derived from agricultural activities (Cassman and Grassini, 2020). In this way, management strategies that increase environmental sustainability, such as increasing agroecosystem diversity (Bowles et al., 2020), and inclusion of pastures in crop rotations (Carswell et al., 2022; García-Préchac et al., 2004) may increase resilience to weather extremes without forgoing yield, whilst the use of high-yielding crop varieties, fertilization, irrigation, and pesticides, have become important tools to improve productivity (Franzluebbers and Martin, 2022; Paruelo and Sierra, 2023).

Developing indicators which quantify and analyze sustainability and management in crop and livestock systems are essential (Chukalla et al., 2020) given the relevance that these systems have in the global food chain (Gerber et al., 2014). As explained by Paruelo & Sierra, (2023), defining indicators is a critical step to quantitatively compare the level of intensification. In addition, a rigorous control and evaluation of the indicators is needed to define a 'safe operating space' for resource-use efficiency (Quemada et al., 2020).

Given the relevance of N in agricultural processes (Godinot et al., 2015), it is crucial to avoid nutrient imbalances (Fowler et al., 2013; Goulding et al., 2008) to deliver an optimal level of Nitrogen (N) input which maximizes N use efficiency (NUE) (Löw et al., 2020; Powell and Rotz, 2015; Quemada et al., 2020). This, in turn, helps minimize the amount of N potentially released into the environment, contributing to the definition of a 'safe operating N space'. However, until now there is not a uniform, robust methodology and protocol to be used when accounting for these risks. Several studies have estimated NUE often using different systems boundaries, scales, regions, input data and assumptions (Gerber et al., 2014; Uwizeye et al., 2020). The EU Nitrogen Expert Panel (2015) proposed a graphical approach to evaluate NUE. The authors defined a target NUE zone delimited by two NUE thresholds (high and low), a minimum N system productivity associated with N removed in food products and including also a fourth threshold linked with the maximum N surplus (NSURP) that a system could admit avoiding a potential release of this nutrient to the environment. This value is considered a proxy of potential environmental losses of N (van Eerdt and Fong, 1998), and a lower value is associated with an efficient use of the N applied (Rose et al., 2023). Although this approach has been frequently used in pure crop (Milroy et al., 2019; Shen et al., 2023; Sung et al., 2023; Yan et al., 2022) or livestock systems (Bratti et al., 2022; Castillo et al., 2021; Groenestein et al., 2019; Vingerhoets et al., 2023), the literature from mixed systems is limited.

In Uruguay, mixed crop-livestock systems have gained relevance since the prevailing regulations on crop rotations (MGAP, 2020), occupying 17% of the total area used by livestock (DIEA - MGAP, 2022), whilst meat and grain production represented 23 and 22% of total exports in 2021, respectively (Uruguay XXI, 2021). Hence, there is a need to understand the drivers and deepen the knowledge around efficiency of these systems, considering the dependence of synthetic fertilizers in production systems and the growing environmental impacts of agricultural systems concerns (Xue et al., 2010).

Therefore, the aim of this study was to quantify and compare the NUE, the associated NSURP and the N balance/retained (at crop and livestock component and at system level) of four pasture crop rotations at different intensity levels utilizing data collected over a 3-y period (2019-2022). We hypothesized that mixed crop livestock systems have the potential to circulate nutrients between both components, improving their availability and potentially reducing external contribution.

4.4. Materials and Methods

Measurements and metrics calculated in this study refer to the third phase of the Palo a Pique Long-Term Experiment (‘Land Expansion and Livestock Intensification’), which started in 2019, after a redesign, as described by Rovira et al. (2020). The main changes that occurred in this phase were: i) relocation of permanent pasture system, ii) addition of a support grassland area to each system, and iii) implementation of a bespoke livestock strategy for each system.

4.4.1. Experimental Site

A long-term pasture crop rotation experiment under no-tillage was installed in 1995 at the ‘Palo a Pique’ Experimental Unit in Treinta y Tres (33◦160 S, 54◦290 W), Uruguay, at the National Institute of Agricultural Research (INIA) facilities. Uruguay is in the subtropical climate zone; the annual mean (± SD) accumulated rainfall in the experimental site for the last 28 years (1995-2022) was 1249 ± 72.x mm per year distributed uniformly throughout the year. The mean maximum and minimum air temperatures for the same period were 23 ± 0.1◦C and 11 ± 0.6◦C, respectively. The research site has a 3% average slope, and the loam soils are Oxyaquic Argiudolls according to (USDA-NRCS, 1996).

4.4.2. Description of the pasture crop rotations

As described in Pereyra Goday et al. (2022), four systems were evaluated. Table 1 describes the crops included in each rotation (pasture-crop rotation or pasture rotation) and purpose of the crop phase (crop or grazing). Pasture crop rotations represent alternative pasture–crop arrangements with different temporal and spatial combinations of land use.

Table 1. Pasture and crop sequence for each system evaluated at Palo a Pique long term experiment, Treinta y Tres, Uruguay (Pereyra Goday et al., 2022).

System[1]	Purpose of crop phase	Rotational year					
		Year 1	Year 2	Year 3	Year 4[2]	Year 5[2]	Year 6[2]
CC	Crop	Oat/Sorghum	Black Oat/ Soybean	Wheat/Sorghum			
	Grazing	Oat/Sorghum	Ryegrass/Moha	Oat/Sorghum			
SR	Crop	Idem[3] CC	Idem CC	Wheat + P1[4]	P2[5]		
	Grazing	Idem CC	Idem CC	P1	P2		
LR	Crop	Idem CC and SR	Idem CC and SR	Wheat + P1	P2[5]	P3[5]	P4[5]
	Grazing	Idem CC and SR	Idem CC and SR	P1	P2	P3	P4
FR	Grazing	Fescue	Fescue	Fescue	Fescue	Fescue	Fescue

[1] CC: Continuous Cropping; SR: Short Rotation; LR: Long Rotation; FR: Forage Rotation. [2] Note that primary data from these years has not yet been collected, but due to the rotational nature within each system on an annual basis, the full six-year cycle can be represented from primary data collected during years one-to-three. [3] The same rotation of CC to first and second year. [4] Pasture follows by the age of the pasture (1 to 2 in SR and 1 to 4 in LR).[5] Pastures in crop area (in SR and LR) are grazed.

The continuous cropping system (CC, 12 ha) is represented by a rotation with two crops per year. CC does not rotate with pastures, but it is complemented with an external area (6 ha) of a permanent improved pasture composed of tall fescue (*Festuca arundinacea Schreb*.), birdsfoot trefoil (*Lotus* *corniculatus L*.), and white clover (*Trifolium repens L*.) re-seeded every five years with the same species to ensure sustained establishment. The short rotation system (SR, 24 ha) alternates in the same land for two years of crops identical to CC with another two years of grass-legume pastures based on Yorkshire fog (*Holcus lanatus L*.) and/or Italian ryegrass (*Lolium multiflorum L*.) interspersed with red clover (*Trifolium pratense L*.). The long rotation system (LR, 36 ha) alternates in the same land area with two years of crops identical to CC and SR followed by four years of grass-legume pastures composed of tall fescue, birdsfoot trefoil and white clover. The forage rotation system (FR, 24 ha) is seeded with tall fescue and does not rotate with grain crops.

Each pasture crop rotation (CC, SR and LR) was split into two halves: one half for grain production (defined as 'crop area'), which were seeded with oats (*Avena byzantina L*.), black oat (*Avena strigose Schreb*.) and wheat (*Triticum aestivum L*.) in winter, and soybean (*Glycine max L*.) and sorghum (*Sorghum bicolor L*.) in summer. The remaining area were oriented to grazing animals (livestock component, defined as 'grazing area') which were seeded as follows: annual ryegrass (*Lolium multiflorum Lam*.) and oats in winter, and sorghum and moha (*Setaria italica*) in summer. Winter crops and pastures were sown from March to June and were usually harvested in November. Summer crops were sown from October to November and harvested in April. Cover crops (black oat) were harvested for hay in October.

As part of the experimental platform redesign implemented in 2019 (Rovira et al., 2020), each system included a bespoke livestock strategy. In CC, SR and LR, animals enter their respective experimental paddocks in April – May each year and remain for one year (rearing animals) or, in the case of finishing animals, until reaching target weights to the slaughterhouse. In FR, animals enter in November- December each year. As reported by Pereyra Goday et al. (2022), in CC the objective was rearing male calves for one year; 32 calves were reared in 2019, 34 in 2020 and 35 in 2021. The average initial liveweight (LW) was 191 ± 16.2 kg LW, 179 ± 17.5 kg LW and 200 ± 30.3 kg LW for the same three years, respectively. In SR, the objective was rearing heifers, achieving 44 (148 ± 17.1 kg LW), 49 (153 ± 16.1 kg LW) and 46 (167 ± 21.4 kg LW) in 2019, 2020 and 2021, respectively. Heifer rearing was complemented with finishing culled cows during May and September, achieving 15 (484 ± 72.2 kg LW), 10 (446 ± 19.3 kg LW) and 10 (483 ± 24.1 kg LW) in 2019, 2020 and 2021, respectively. In LR, the objective was rearing male calves and finishing steers over a period of 18 months. A total of 50 male calves were allocated to LR in 2019 (190 ± 14.3 kg LW), 2020 (185 ± 15.4 kg LW) and 2021 (199 ± 31.2 kg LW). Steer's weight in LR was 393 ± 28.2 kg LW in 2019, 347 ± 26.7 kg LW in 2020 and 369 ± 32.3 kg LW in 2021. FR is the only system that begins at the end of the spring (Nov-Dec) with yearling steers instead of weaned calves. The objective of the livestock strategy in FR was to produce a finished steer ready for slaughter in 12-15 months. A total of 47 (318 ± 28.2 kg LW), 30 (250 ± 12.8 kg LW), 35 (253 ± 32.1 kg LW) and 41 (263 ± 65.3 kg LW) Aberdeen Angus steers entered the system in December 2018, December 2019, November 2020, and November 2021, respectively. British early-maturing beef cattle were used in the four systems (Hereford, Aberdeen Angus, and Hereford – Angus cross).

The experiment lacks synchronic replications, but all phases of the rotations are present each year, represented by paddocks of 3 ha in CC, SR, and LR. In FR, the 24 ha were divided into 5 paddocks of 4.8 ha each corresponding to fescue seeded in 2013 (9.6 ha), 2014 (9.6 ha), and 2020 (4.8 ha). Each system has a support area of natural grassland (NG) to handle the animals, when necessary (i.e., during periods with low forage availability in the seeded area), keeping the animals independently within each system. The proportion of NG surface is: 33%, 29%, 26%, and 33% of the total area for CC, SR, LR, and FR, respectively. Animals were handled to graze annual forage crops (ryegrass, oat, sorghum and moha), permanent pastures (in grazing and crop area of SR and LR), permanent improved pasture (in CC) and natural grasslands. Detailed information about experimental design, management, and productive performance can be found in Pereyra Goday et al. (2022) and Rovira et al. (2020).

4.4.3. Data analysis and scope of the study

N balance (NBAL) was calculated based on N inputs minus N outputs and NUE calculated from food N outputs relative to all N inputs, according to Erisman et al. (2018). NSURP was calculated as all the N inputs minus N removed in food products (EU Nitrogen Expert Panel, 2015). We assessed NBAL, NUE and NSURP at component level (crop and livestock) and at system level. The study boundary was the farm gate. Crop component was defined as the area where exclusively grain was produced (6 ha in each system), pasture phase was not included in the crop component. Livestock component included the total grazing area (permanent pastures, annual pastures, permanent improved and NG) which was 18 ha in CC, 22 ha in SR, 50 ha in LR and 36 ha in FR. For all systems, the main N inputs considered were N in synthetic fertilizers (di ammonium phosphate and urea), biological N fixation (pasture legumes), atmospheric N depositions and N in animal feed, while outputs were related to all N removed in food products and N losses (Figure 1). As N atmospheric and leached losses were not directly assessed these (N_2O, N_2 and NH_3 gas and leached NO_3-) were modeled using the DNDC software. In this regard, we leveraged calibrated and validated coefficients of a prior study including N fluxes in comparable crop-pasture-livestock rotations (Castillo et al., 2023).

N exported from the crop component by grains (soybean, oat, and wheat) was estimated according to our data from analysis: 12.6% crude protein content in wheat, 36% crude protein content in soybean and 10.8% crude protein content in oat. Straw from crops were left on the field. N exported by meat and animal-by-products were estimated in 2.6% of liveweight according to FAO (2018). N fixation from pasture legumes was obtained from a previous study (Pereyra Goday et al., 2022), whereas N fixation from soybean was estimated according to Salvagiotti et al. (2015).

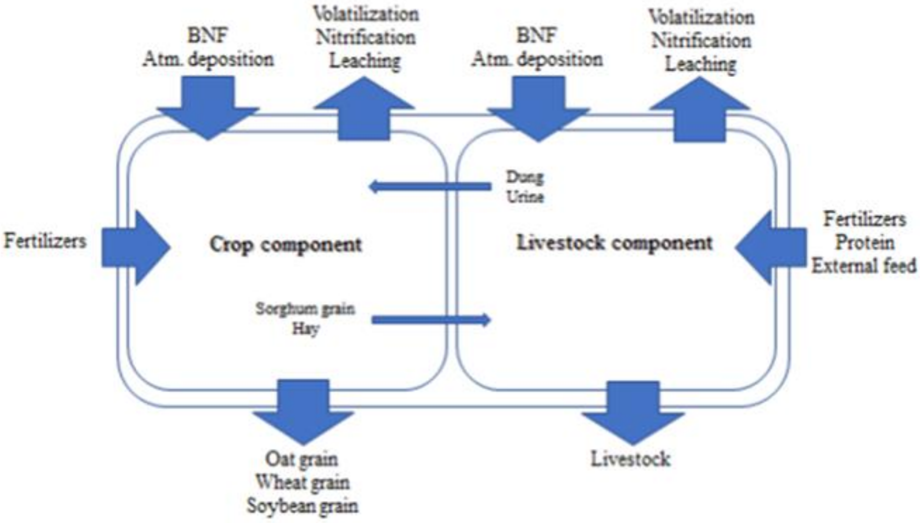


Figure 1. Scheme of N fluxes in pasture crop rotation at 'Palo a Pique' long term experiment in Treinta y Tres, Uruguay.

A low (≥ 50%) and high (≤ 90%) threshold was set up to evaluate NUE, according to EU Nitrogen Expert Panel (2015), for the livestock component the threshold was ≥ 10% and ≤ 25%, according to Gerber et al., (2014). For the crop component NUE values greater than 90% could be associated with soil N mining, values below 50% could be associated with low NUE. Animal systems on the other hand, reach a lower NUE, therefore both efficiency thresholds were set accordingly. For the whole system we propose a low (< 18%) and high (> 45%) threshold to evaluate NUE, according to a prorated average taking into consideration the crop and the livestock area in the experiment. The maximum NSURP was defined at 80 kg N ha^{-1} year^{-1} for crop component and 110 kg N ha^{-1} year^{-1} for livestock component following EU Nitrogen Expert Panel (2015), whereas the system NSURP was defined at 104 kg N ha^{-1} year^{-1}. For the livestock component, the minimum N desirable productivity was set at 10.4 kg N ha^{-1} year^{-1} according to an experimental hypothesis of 400 kg LW production ha^{-1} year^{-1} (Pereyra Goday et al., 2022). For the crop component, the threshold was defined as 80 N ha^{-1} year^{-1}, following EU Nitrogen Expert Panel (2015). The component effect on NUE and NSURP was tested using least difference significative (LSD) model, considering the year as a replica.

4.5. Results

4.5.1. N inputs and N outputs

N inputs and outputs are presented in Table 2 and 3. Although 'complementary outputs' were not part of the NUE calculations, they were accounted as a transference from the crop component to the livestock component (e.g., hay and grain to feed animals). Differences in 'feed production' for animals and 'feed to animals' value in crop vs livestock component are explained by differences in area considered and the inclusion of commercial feed. Animal deposition (feces and urine) were not considered for the calculation given that the grazing periods in the pastures of the agricultural phase were short and far from the beginning of cultivation period.

Table 2. Total N inputs and outputs at crop component of the 'Palo a Pique' long term experiment.

	Systems[1]			
	CC	SR	LR	FR
N Inputs (kg ha^{-1} $year^{-1}$)[2]				
Synthetic fertilizers	50.5/70.5/44.1	47.9/33.9/44.1	37.7/40.1/58.2	-
Biological N fixation	76.9/77.5/85.3	97.2/80.1/74.4	87.9/108.2/90.8	-
Atmospheric deposition	5/5/5	5/5/5	5/5/5	-
Food N outputs (kg ha^{-1} $year^{-1}$)[2]				
Wheat	7.7/36.9/23.7	12.7/27.1/28.3	11.7/26.1/27.2	-
Oat	0/0/0	12.8/21/21	10.6/18.6/18.6	-
Soybean	65.5/66/72.7	82.8/68.3/63.4	74.9/87/77.3	-
Complementary N outputs (kg ha^{-1} $year^{-1}$)[2]				
Feed production[3]	16.8/19.1/5.1	17.7/17.9/3.8	16.8/18.6/4.2	-

[1]CC: Continuous cropping; SR: Short rotation; LR: Long rotation; FR: Forage rotation. [2] 2019-2020/2020-2021/2021-2022. [3]To feed livestock in livestock area (include hay and sorghum grain).

Table 3. N inputs and N outputs at livestock component of the "Palo a Pique' long term experiment.

	Systems[1]			
	CC	SR	LR	FR
Inputs (kg N ha^{-1} $year^{-1}$)[2]				
Synthetic fertilizers	21.1/43.2/30.4	65.2/65.2/47	26.4/26.4/28.3	112.7/107.8/122.7
Biological N fixation	6.4/0/2.2	13.5/18/46.3	11.4/10.8/9.4	0/0/0
Atmospheric deposition	5/5/5	5/5/5	5/5/5	5/5/5
Feed to animals[3]	12.8/12.7/3.4	30.4/26.7/5.7	20.6/21.2/13.5	0/21.2/0
Food outputs (kg N ha^{-1} $year^{-1}$)[2]				
Livestock	11.1/12.3/11.6	14.5/16.3/14.5	9.5/11.8/11.8	6.8/7.4/5.9

[1]CC: Continuous cropping; SR: Short rotation; LR: Long rotation; FR: Forage rotation. [2] 2019-2020/2020-2021/2021-2022. [3]To feed livestock (include hay, sorghum grain, protein, and commercial feed).

4.5.2. N losses

On average, N losses accounted for 38 ± 3.1, 22 ± 2.6 and 24 ± 1.1 kg N ha^{-1} $year^{-1}$ for CC, SR and LR, respectively, in the crop component. Approximately 71% of total N losses occurred during winter crops and the remaining occurred during summer crops. The main N loss source in this component was volatilization (45%) followed by lixiviation (35%).

In the livestock component N losses were 32 ± 4.0, 40 ± 4.3, 18 ± 0.9 and 52 ± 3.1 kg N ha^{-1} $year^{-1}$ for CC, SR, LR and FR, respectively. The highest values of N losses were observed in tall fescue in FR (81 kg N ha^{-1} $year^{-1}$), which had inputs of 184 kg N ha^{-1} $year^{-1}$ as fertilizer in tall fescue area. In this rotation, the main gas losses were volatilization (56%). For natural grassland (NG), N losses were 9.6 kg N ha^{-1} $year^{-1}$ when grazing was included, whereas N losses dropped up to 3.6 kg N ha^{-1} $year^{-1}$ when grazing was excluded. On average, N losses associated with the permanence of grazing animals in the experiment accounted for 29%, 31%, 29% and 26% of the total N losses to CC, SR, LR and FR, respectively.

4.5.3. N balance

N inputs and outputs for each component are shown in Table 2 and 3. N losses were detailed in section 3.2. N balance at crop component were 2.2 ± 9.21, -14.9 ± 20.43 and -3.5 ± 12.88 kg N ha^{-1} $year^{-1}$ for CC, SR and LR, respectively. Figure 2 shows N balance for crop component. On average, of all inputs, fertilizers represented 38%, 30.9% and 30.1%, whereas biological N fixation (BNF) from soybean accounted for 55.1%, 61.8% and 63.3% for CC, SR and LR, respectively. The remaining input was atmospheric deposition.

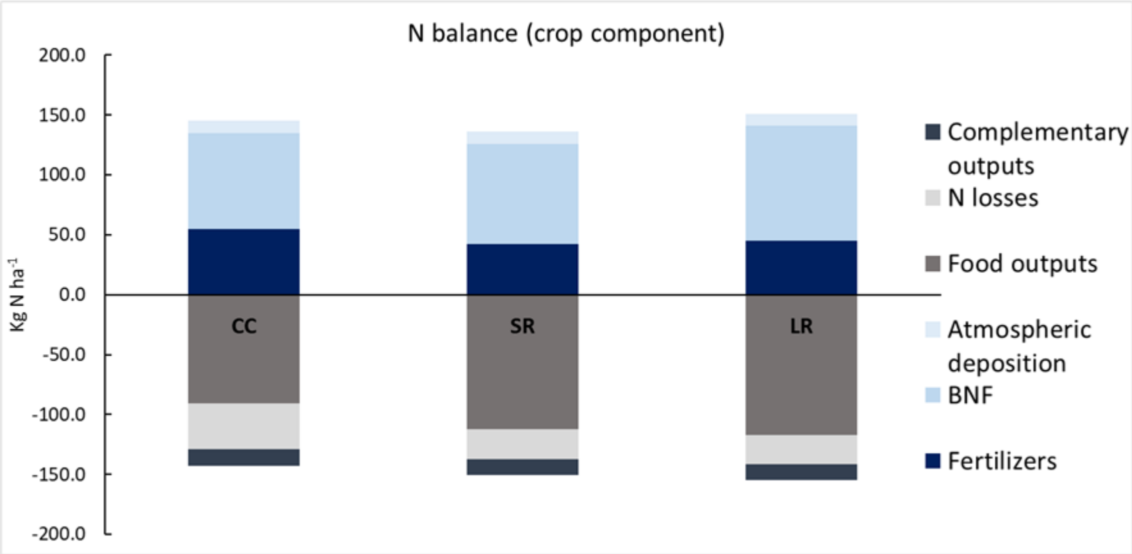


Figure 2. Components of N balance of crop component at 'Palo a Pique long term experiment. BNF: biological N fixation.

Figure 3 shows N balance for livestock component, including rotation area and NG area. For the experimental period (2019-2022) N balance was 6.1 ± 4.42, 59 ± 4.3, 29 ± 1.2 and 64 ± 3.5 kg N ha^{-1} $year^{-1}$, for CC, SR, LR and FR, respectively. On average, of all inputs, synthetic fertilizers accounted for 64.3%, 53.3%, 44.3% and 90.5%, external feed represented 19.6%, 18.9%, 30.2% and 5.6%, and BFN was 5.8%, 23.4%, 17.3% and 0%, for CC, SR, LR and FR, respectively. The remaining percentage was explained by atmospheric deposition.

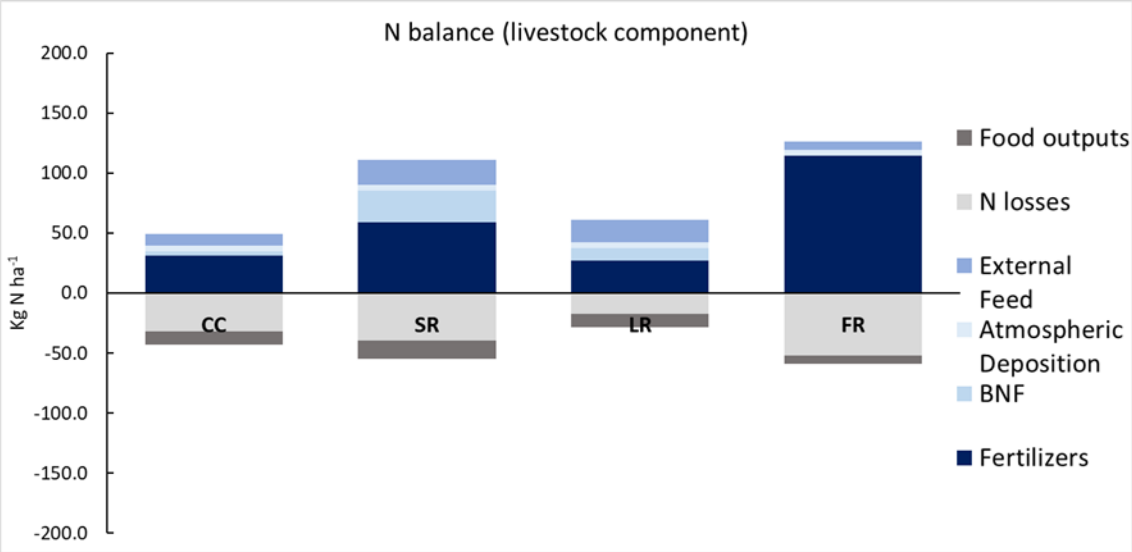


Figure 3. Components of N balance of livestock component at 'Palo a Pique long term experiment. BNF: biological N fixation.

Given that crop and livestock component are part of a single system that combines agricultural and livestock production, a whole system balance was calculated (Figure 4). N balance was positive in all cases being: 4.9 ± 1.5x, 41 ± 3.1, 29 ± 2.2, 64 ± 3.5 kg N ha^{-1} $year^{-1}$ for CC, SR, LR and FR, respectively.

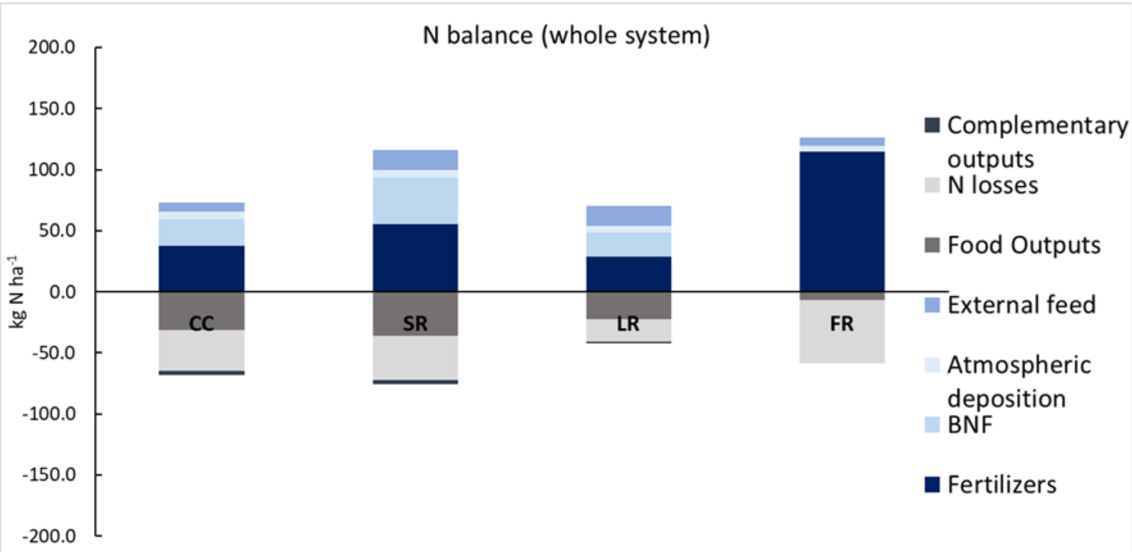


Figure 4. Components of N balance (whole system) at 'Palo a Pique long term experiment. BNF: biological N fixation.

4.5.4. Soil N

Soil N concentration (0-15 cm) for each pasture crop rotation (CC, SR, LR and FR) and component (crop from 2013 to 2021 and livestock from 2006 to 2021) ranged between 0.142 g kg^{-1} (CC) and 0.208 g kg^{-1} (LR) in the crop area, whereas in the grazing area (livestock component) values ranged between 0.132 g kg^{-1} (CC) and 0.211g kg^{-1} (FR). For the entire historical data series, increase of N content in crop component was on average 0.023 g kg^{-1} and differences were not significant between CC, SR and LR. However, systems that included pastures in their rotation had higher values of N content in soils, than the CC. For the livestock component, the increase in N content in soil (2006 – 2021) was 0.024 g kg^{-1} and differences were not significant between SR, LR and FR.

An upward trend for all systems and components was observed when comparing 2019 vs. 2021. Soil N content increased by 4.3, 4.4, and 3.1% per year in the crop component in LR, SR and CC, respectively, with differences between systems ($p = 0.015$). For the livestock component, N content in soil increased on average 6.8, 3.6, 6.9 and 8.7% in LR, SR, CC and FR, respectively. Differences were not detected between systems.

4.5.5. Crop NUE and NSURP

On average, crop NUE (NUE_C) values did not differ ($p = 0.07$) among systems, being 67 ± 8.2, 84 ± 12.5 and 78 ± 5.8 % for CC, SR and LR, respectively, meaning that the achieved NUE_C values were within the defined target NUE zone (Figure 5). The highest variability among years was observed for N inputs in SR, whereas the lowest variability was observed in N outputs in the same system. Highest NUE corresponded to SR, reaching 94%, and the lowest NUE was observed in CC (53.3%). The NSURP (kg N ha^{-1} $year^{-1}$) were 54 ± 10.6, 23 ± 18.5 and 34 ± 6.2 kg N ha^{-1} $year^{-1}$, for CC, SR and LR, respectively, where all the calculated values were below the defined threshold (80 kg N ha^{-1} $year^{-1}$).

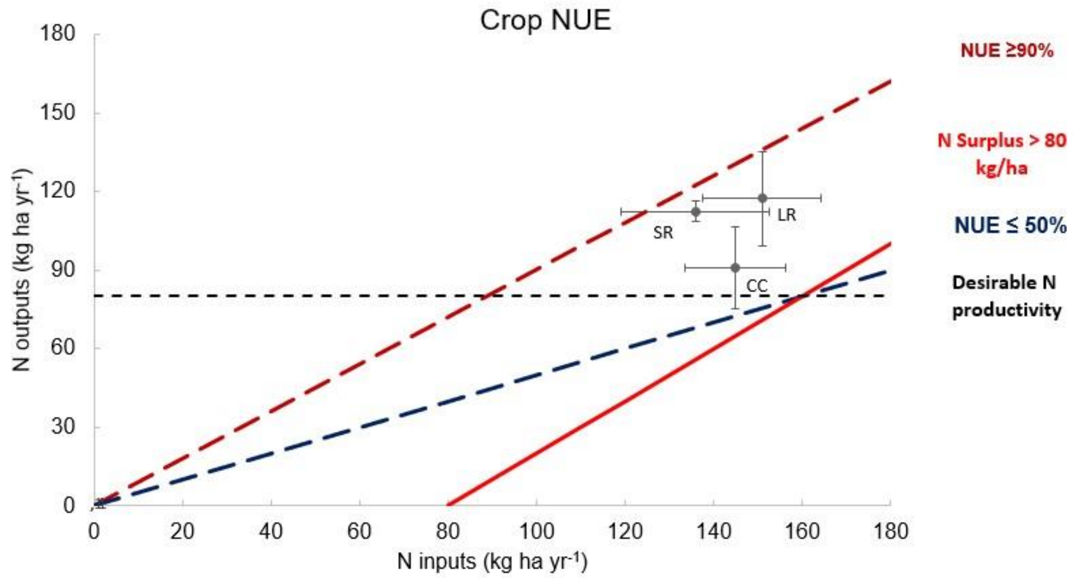


Figure 5. N outputs and N inputs average (2019-2022) at crop component. Dashed orange and blue lines indicate NUE (= outputs/inputs X 100) of 90% and 50% respectively for crops. Dashed black line indicate the expected N output for a desirable level of production. We kept the theoretical productivity given the low potential for crop yield in the area of study, according with Terra et al., (2006).

4.5.6. Livestock NUE and NSURP

The livestock NUE (NUE_L) showed differences among systems where CC was five-, three- and one-fold greater than FR, SR and LR, respectively. NUE_L was 24.4±4.14, 9.9±1.23, 14.3±1.92 and 5.5±0.73 to CC, SR, LR and FR respectively. While LR reached an intermediate NUE_L value when compared with CC and FR, the achieved efficiency on SR was similar to LR and FR. Significant differences were detected between FR and LR, FR and CC, SR and CC, LR and CC ($p = 0.0001$). The pasture length did not influence the achieved NUE_L when analyzing both SR and LR, being both closer to FR than CC.

Once N inputs were plotted against N outputs, differences in NUE_L and the deviation values for this parameter among systems were observed. Except for FR, the remaining systems achieved NUE values between the defined thresholds, with just a few records out of that, as well as reaching the defined minimum N productivity (10.4 kg N ha^{-1} yr^{-1}) (Figure 6). The NSURP was 37 ± 10.1, 50 ± 2.3, 100 ± 11.4, and 116 ± 5.4 kg N ha^{-1} $year^{-1}$ for CC, LR, SR, and FR, respectively. Values for SR and FR approached or exceeded the defined threshold.

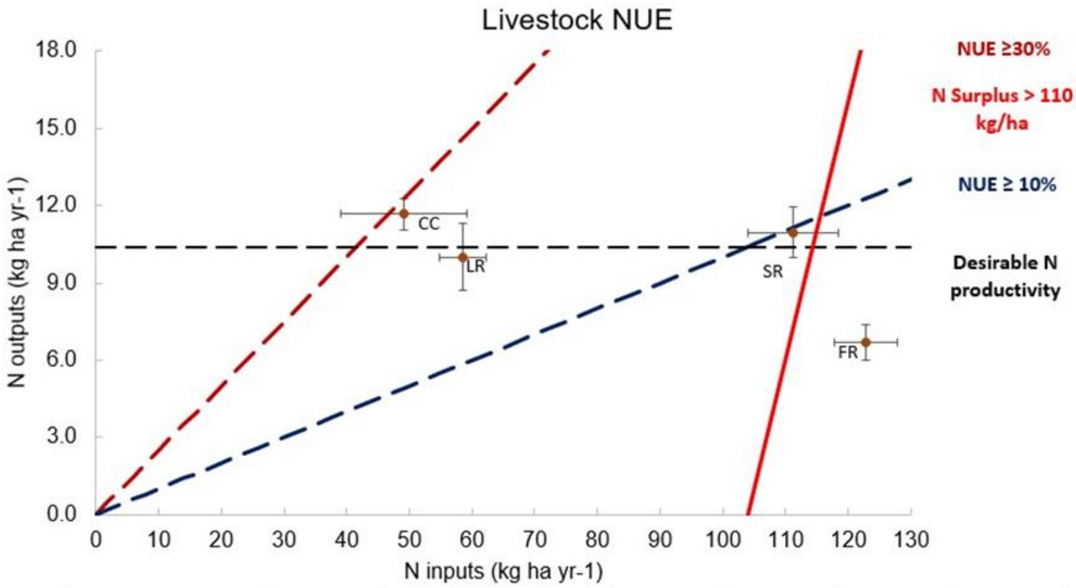


Figure 6. N outputs and N inputs average (2019-2022) for the livestock component. Dashed orange and blue lines indicate NUE (= outputs/inputs X 100) of 25% and 10% respectively for crops. Dashed black line indicate the expected N output for a desirable level of production (10.4 kg N ha^{-1} $year^{-1}$= 400 kg LWG ha^{-1} $year^{-1}$).

4.5.7. <u>System NUE and NSURP</u>

At system level (Table 7), NUE was significantly different between CC, SR and LR ($p<0.0001$). NUE values were 43.4±5.93, 28.1±2.03, 29.3±3.83 and 5.5±0.73 to CC, SR, LR and FR respectively. N surpluses at system level were 42 ± 8.8 kg N ha^{-1} $year^{-1}$, 84 ± 6.9 kg N ha^{-1} $year^{-1}$, 48 ± 2.2 kg N ha^{-1} $year^{-1}$ for CC, SR and LR, respectively.

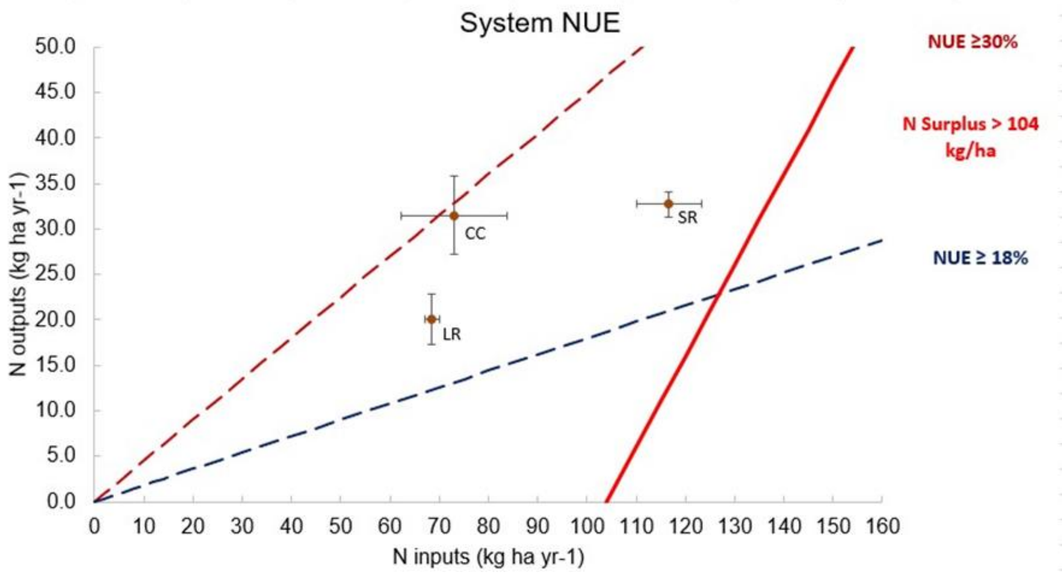


Figure 7. N outputs and N inputs average (2019-2022) at crop livestock system.

4.6. Discussion

The upward worldwide trend in N consumption shows the dependence agricultural systems have for N (Tilman et al., 2002). Mixed crop – livestock systems play a crucial role in the production of high-quality food as well as in system sustainability, which is gaining interest in recent years (Ryschawy et al., 2012). Given the dependence of these systems on N fertilizers, understanding the dynamics and processes associated with this nutrient, as well as management strategies aimed at improving the efficiency of use and reducing losses is critical. The inclusion of NUE calculations and N balance at farm level allows the interaction between crops and livestock to be investigated to help improve resource use and reduce environmental losses at system level. As stated by Oliveira et al., (2022), NUE is an indicator of sustainability, the evolution of this indicator is crucial to evaluate the intervention implemented in long-term experiments and, at the same time, provide information to farmers and policy makers with respect to sustainable food production systems.

4.6.1. NUE of crop component

For all crop rotations, NUE_C was within the defined thresholds. Differences observed in NUE_C could be explained by differences in crop yield and fertilization strategies in each year (i.e., CC had lower crop yields as was stated in a previous study by Pereyra Goday et al., 2022). NUE_C values reported by other studies had high variability; Hutchings et al., (2020) reported NUE_C values of 65 - 92% for arable crops in Northern and Southern Europe, Shen et al. (2023) found NUE_C values of 49% for wheat production in China, whilst Gu et al. (2017), reported NUEC values of 39% for croplands in China. According to EU Nitrogen Expert Panel (2015), the reported values in this study for SR and LR fell in the 'balanced N fertilization' category, whereas CC seemed to indicate a 'risk of N losses', which is aligned with the higher value of N losses estimated in CC compared to SR and LR. On the other hand, N balance tended to be neutral with variability between years. For CC, we observed higher N losses compared to the literature which could be attributed to the source of N used and the way it was applied (urea without inhibitors and applied directly without incorporation into the soil), which is consistent with the N internal fluxes reported by Pravia et al., (2019).

4.6.2. NUE of livestock component

The livestock component of the four systems had higher productivity values than the average for rearing and fattening systems (200 kg LWG ha^{-1} $year^{-1}$; Plan Agropecuario, 2021). However, the evaluated systems had a dependence on external sources of N (synthetic fertilizers and external feed), which determined medium-low values of NUE. For the livestock component NUE_L values were lower than NUE_C, due to biological differences at the trophic level (Godinot et al., 2015).

CC presented the highest value of NUE_L, close to the upper threshold, explained by the highest productivity in terms of kg LWG ha^{-1} $year^{-1}$ (426 kg ha^{-1} $year^{-1}$) due to the higher efficiency to gain liveweight of rearing male calves (CSIRO, 2007). LR and SR had similar values of NUE_L, even though SR had higher values of productivity than LR (418 kg LWG ha^{-1} $year^{-1}$ in SR vs 369 kg LWG ha^{-1} $year^{-1}$ in LR). This could be explained by the higher levels of inputs, such as productivity of red clover in SR, which accounted for 31 ± 12.3% of total dry matter (DM) production of the permanent pastures. Therefore, there was high values of biological N fixation, with high variability among years and N fertilization.

The system with the lowest NUE_L was FR, which was below the lower reference threshold. This was explained by high N inputs (184 kg N ha^{-1} $year^{-1}$ as synthetic fertilizer in tall fescue area) and a lower productivity (310 kg LWG ha^{-1} $year^{-1}$), since this system is focused on finishing animals to slaughterhouse, which is a low-efficiency process compared to the other categories. Although DM production was high (6867 kg DM ha^{-1} $year^{-1}$), conversion efficiency and forage utilization were low in this system (19.2 kg DM kg LWG^{-1} and 37% utilization, respectively). Also, gas N losses were highest in this system associated with the greater amounts of N fertilizer applied.

NUE_L values were higher than those reported by Jin et al., (2021) for Chinese livestock systems (3% – 4%), whilst Castillo et al., (2021) reported similar values (13.2%) for extensive livestock systems in rotation with rice in Uruguay. Although the latter value was close to that of our study, the LWG differed considerably in favor of our study (+400 %), mainly due to the livestock intensification level (extensive vs. intensive, respectively).

The positive N balance observed at the livestock component agrees with Ryschawy et al. (2012). A positive balance could induce lower demand for N inputs in livestock compared with crop production (Oliveira et al., 2022; Powell & Rotz, 2015), given that grazing animals contribute positively with soil N content through manure and urine deposited in the field (Segura et al., 2023). Furthermore, livestock production plays a critical role in food security, by supplying high-quality, nutrient-dense food (Rivero et al., 2021).

Livestock in mixed crop-livestock systems play an important role in non-arable areas, where cropping is unsustainable (Wilkinson & Lee, 2018), using non-human edible crops/pastures or crop residues as feed (Sekaran et al., 2021). As was summarized by de Faccio Carvalho et al. (2021), livestock grazing provides several benefits such as system stability, resilience, profitability, soil health (more microbial activity) and biomass production. This wider view of sustainability delivers to the concept of circularity of livestock production (Lemaire et al., 2023; Ward et al., 2016), where livestock play a beneficial role in nutrient cycling and ecosystem services; thus, maintaining soil fertility in agroecosystems could outweigh the adverse effects of a higher carbon footprint (driven by enteric fermentation derived biogenic methane).

4.6.3. System NUE and prospects for improvement

Based on the findings of this study, we can categorize the assessed systems according to their results. Systems incorporating pasture in rotation with crops (SR and LR) consistently demonstrated, on average, lower N losses, higher soil N gains, and higher efficiency values within the optimal desirable range. Conversely, rotation excluding pastures (CC) led to elevated atmospheric N losses across both crop and livestock components, along with suboptimal efficiencies in the crop component but higher efficiency in the livestock component (attributable to the specific livestock management approach). On the other hand, rotation exclusively comprising pastures (FR) with substantial N synthetic fertilizer inputs exhibited the highest estimated atmospheric N losses and notably low NUE, thereby raising significant concerns regarding potential environmental risks. As will be seen later, it is possible to reduce the use of synthetic fertilizers and replace them with organic sources.

Although the four systems exhibited soil N gains during the experimental period, this observation must be put into perspective. It is important to note that a period of N accumulation in the soil is often followed by a subsequent decline. This pattern is frequently observed in annual or cash crops that are rotated with pastures (Grahmann et al., 2020). These increases in N stored in soil during the experimental period represented 26.3 kg N ha-1 year-1, 18.3 kg N ha-1 year-1, 30.1 kg N ha-1 year-1 and 41.4 kg N ha-1 year-1 for CC, SR, LR and FR, respectively, which is close to the results obtained in the balance.

To improve all the assessed N parameters in this study, we believe it is possible to consider system-level improvements with a focus on the livestock component. This is because the livestock component offers greater opportunities for improvement, as stated by Castillo et al (2023), through agronomy management to achieve a 'safe space' in terms of NUE. In general terms, greater utilization of pastures, i.e. improving grazing management (mainly in those systems that include perennial pastures) would allow an increase in N output from the system and therefore would have implications for NUE. On the other hand, an adequate quantification of all N flows, including those derived from animal production (feces and urine) would allow to a rational use of external N sources.

To improve NUE in CC system, we must 'avoid soil degradation' through improving biomass production, which could have implications for N and C cycles and reduce N losses (Bilotta et al., 2007; de Faccio Carvalho et al., 2010). Also, redesigning the rotation with the inclusion of high productivity legumes could improve the quality of the diet for animals (Soussana & Lemaire, 2014), and at the same time increase the organic N inputs. On the other hand, an increase in the use of supplements could help to improve in terms of efficiency, but it is important to consider the economic implication into the whole system and the balance between N and C efficiency.

In the SR system the strategy was 'extensification' given that it is close to the lower threshold (10%) for the livestock component. An alternative could be considering N fixed by legumes, and reducing N fertilization accordingly (Carswell et al., 2022). On the other hand, a redesign of the livestock strategy (i.e., exclusively rearing males) could lead to improved productivity. In this system, N inputs cannot be increased; however, it is important to consider that these results could be influenced by weather conditions during the experimental period, and probably in the longer term this system would be more aligned with the LR, given the similarities it presents in the composition of the rotation and the combination of livestock strategies (rearing, fattening).

The LR system had on average lower productivity than the desirable level for livestock production. It could therefore be improved through ‘intensification’ of production to reach 400 kg LWG ha^{-1} $year^{-1}$, for example by supplementing the reared calves (given the high efficiency of conversion of this category) which would represent an increase of 0.010 kg LWG $calf^{-1}$ day^{-1}, according to annual performance reported by a previous study (Pereyra Goday et al., 2022). Another option could be increasing the proportion of rearing animals in the herd. However, this could have negative effect on the economic results of the system, given the differences in sales prices between rearing and fattening animals.

Fertilization strategy used in FR could be revised (e.g., N source, placement, objective N fertilization method) to maximize tall fescue DM production and fertilization response, reduce N losses and reduce nitrous oxide (N2O) emissions (Pereyra Goday et al., 2024) thereby ‘increasing efficiency’. According to the performance obtained, animals should increase around 35% LWG per animal to reach the target of 400 kg LWG ha^{-1} $year^{-1}$. Hence, the inclusion of legumes could be beneficial (Soussana & Lemaire, 2014) by improving forage quality, and, at the same time, reducing cost of production. Also, grazing management could be adjusted, considering an optimal balance between animal production and soil-plant carbon balance to improve conversion efficiency and utilization of forage (Szymczak et al., 2023). On the other hand, the livestock strategy could be revised (i.e., exclusively finishing animals which would enter the system with higher LW).

Finally, adequately quantifying the N contribution from all components combining with modelling tools, will reduce production costs, environmental risks, improve productivity, and move towards more efficient and sustainable crop-livestock systems (Castillo et al., 2023).

4.6.4. Limitations of the study

Given that our study draws information regarding the management and operation of a long-term semi-commercial scale experiment (Rovira et al., 2020), there are limitations in obtaining data for some nitrogen flows. For instance, nitrogen contributed to the soil by feces and urine (since the animals rotate grazing in different paddocks and natural grassland), and the nitrogen contribution from crop residues left in the field, which also help to increase nitrogen content in the soil. Additionally, the management of nitrogen fertilization, which does not differ between years except for rare exceptions, does not allow for the observation of the potential effect of crop residues and the animal component on NUE. As previously explained, these limitations can be addressed with adequate quantification of these flows.

4.7. Conclusions

By evaluating NUE and N balance values, we gain insights into the underlying processes within mixed crop-livestock rotations and can assess strategies for enhancing these indicators. Given the escalating concerns regarding the sustainability of food production processes, our findings contribute to advancing farm-level knowledge through high-resolution data and offer valuable insights for policymakers and farmers alike.

The sustainability of mixed crop-livestock systems hinges on the complementarity and interconnectivity between their components and the appropriate combination of crops and pastures. While both components -crop and livestock- have room for improvement in terms of management and outcomes (as discussed), the greatest potential for enhancement lies within the livestock component. The utilization of software to model processes within mixed crop-livestock systems warrants adjustments to better serve as a tool for enhancing our understanding of these systems.

The long-term nature of these studies presents an opportunity to explore further into the dynamics, outcomes, and processes involved. Therefore, extending this study over additional years is imperative for gaining comprehensive insights.

Acknowledgements

We thank INIA's field and laboratory staff and students who participated in the collection and processing of data. INIA, Rothamsted Research and Harper Adams University are all members of the Global Farm Platform initiative (www.globalfarmplatform.org) collaboratively working towards sustainable ruminant livestock production systems.

Funding

This research was funded by: Long Term Experimental Platforms project (INIA), F.P-G was funded by posgraduatepostgraduate fellowship: National Institute of Agricultural Research doctoral fellowship and National Agency of Research and Innovation (ANII) code MOV_CA_2021_1_171482 (mobility fellowship). Support in writing up the work was greatly received by the Biotechnology and Biological Sciences Research Council (BBSRC) through the strategic program Soil to Nutrition (S2N; BBS/E/C/000I0320) and Growing Health (BB/X010953/1) at Rothamsted Research. The contributions by MJR were also funded by the Natural Environment Research Council (NERC) under research Programme NE/W005050/1 AgZero+: Towards sustainable, climate-neutral farming. AgZero+ is an initiative jointly supported by NERC and BBSRC.

References

Bilotta, G.S., Brazier, R.E., Haygarth, P.M., 2007. The Impacts of Grazing Animals on the Quality of Soils, Vegetation, and Surface Waters in Intensively Managed Grasslands. pp. 237–280. https://doi.org/10.1016/S0065-2113(06)94006-1

Bowles, T.M., Mooshammer, M., Socolar, Y., Calderón, F., Cavigelli, M.A., Culman, S.W., Deen, W., Drury, C.F., Garcia y Garcia, A., Gaudin, A.C.M., Harkcom, W.S., Lehman, R.M., Osborne, S.L., Robertson, G.P., Salerno, J., Schmer, M.R., Strock, J., Grandy, A.S., 2020. Long-Term Evidence Shows that Crop-Rotation Diversification Increases Agricultural Resilience to Adverse Growing Conditions in North America. One Earth 2, 284–293. https://doi.org/10.1016/j.oneear.2020.02.007

Bratti, F., Luiz Locatelli, J., Henrique Ribeiro, R., Renan Besen, M., Dieckow, J., Bayer, C., Thiago Piva, J., 2022. Nitrous oxide and methane emissions affected by grazing and nitrogen fertilization in an integrated crop-livestock system. Geoderma 425, 116027. https://doi.org/10.1016/j.geoderma.2022.116027

Carswell, A., Sánchez-Rodríguez, A.R., Saunders, K., le Cocq, K., Shaw, R., Cotton, J., Zhang, Y., Evans, J., Chadwick, D.R., Jones, D.L., Misselbrook, T., 2022. Combining targeted grass traits with red clover improves grassland performance and reduces need for nitrogen fertilisation. European Journal of Agronomy 133, 126433. https://doi.org/10.1016/j.eja.2021.126433

Cassman, K.G., Dobermann, A.R., Walters, D.T., 2002. Agroecosystems, Nitrogen-use Efficiency, and Nitrogen Agroecosystems, Nitrogen-use Efficiency, and Nitrogen Management Management.

Cassman, K.G., Grassini, P., 2020. A global perspective on sustainable intensification research. Nat Sustain 3, 262–268. https://doi.org/10.1038/s41893-020-0507-8

Castillo, J., Kirk, G.J.D., Rivero, M.J., Dobermann, A., Haefele, S.M., 2021. The nitrogen economy of rice-livestock systems in Uruguay. Glob Food Sec 30. https://doi.org/10.1016/j.gfs.2021.100566

Castillo, J., Kirk, G.J.D., Rivero, M.J., Haefele, S.M., 2023. Regional differences in nitrogen balance and nitrogen use efficiency in the rice–livestock system of Uruguay. Front Sustain Food Syst 7. https://doi.org/10.3389/fsufs.2023.1104229

Chukalla, A.D., Reidsma, P., van Vliet, M.T.H., Silva, J.V., van Ittersum, M.K., Jomaa, S., Rode, M., Merbach, I., van Oel, P.R., 2020. Balancing indicators for sustainable intensification of crop production at field and river basin levels. Science of The Total Environment 705, 135925. https://doi.org/10.1016/j.scitotenv.2019.135925

CSIRO, 2007. Nutrient requirements of domesticated ruminants. CSIRO Pub.

de Faccio Carvalho, P.C., Anghinoni, I., de Moraes, A., de Souza, E.D., Sulc, R.M., Lang, C.R., Flores, J.P.C., Terra Lopes, M.L., da Silva, J.L.S., Conte, O., de Lima Wesp, C., Levien, R., Fontaneli, R.S., Bayer, C., 2010. Managing grazing animals to achieve nutrient cycling and soil improvement in no-till integrated systems. Nutr Cycl Agroecosyst 88, 259–273. https://doi.org/10.1007/s10705-010-9360-x

de Faccio Carvalho, P.C., de Albuquerque Nunes, P.A., Pontes-Prates, A., Szymczak, L.S., de Souza Filho, W., Moojen, F.G., Lemaire, G., 2021. Reconnecting Grazing Livestock to Crop Landscapes: Reversing Specialization Trends to Restore Landscape Multifunctionality. Front Sustain Food Syst. https://doi.org/10.3389/fsufs.2021.750765

DIEA - MGAP, 2022. ANUARIO ESTADÍSTICO AGROPECUARIO.

Erisman, J., Leach, A., Bleeker, A., Atwell, B., Cattaneo, L., Galloway, J., 2018. An Integrated Approach to a Nitrogen Use Efficiency (NUE) Indicator for the Food Production–Consumption Chain. Sustainability 10, 925. https://doi.org/10.3390/su10040925

EU Nitrogen Expert Panel, 2015. Nitrogen Use Efficiency (NUE) - an indicator for the utilization of nitrogen in agriculture and food systems. Wageningen, Netherlands.

FAO, 2018. Nutrient flows and associated environmental impacts in livestock supply chains: Guidelines for assessment (Version 1). FAO, Rome.

Fowler, D., Coyle, M., Skiba, U., Sutton, M.A., Cape, J.N., Reis, S., Sheppard, L.J., Jenkins, A., Grizzetti, B., Galloway, J.N., Vitousek, P., Leach, A., Bouwman, A.F., Butterbach-Bahl, K., Dentener, F., Stevenson, D., Amann, M., Voss, M., 2013. The global nitrogen cycle in the twenty-first century. Philosophical Transactions of the Royal Society B: Biological Sciences 368, 20130164. https://doi.org/10.1098/rstb.2013.0164

Franzluebbers, A.J., Martin, G., 2022. Farming with forages can reconnect crop and livestock operations to enhance circularity and foster ecosystem services. Grass and Forage Science 77, 270–281. https://doi.org/10.1111/gfs.12592

García-Préchac, F., Ernst, O., Siri-Prieto, G., Terra, J.A., 2004. Integrating no-till into crop–pasture rotations in Uruguay. Soil Tillage Res 77, 1–13. https://doi.org/10.1016/j.still.2003.12.002

Gerber, P., Uwizeye, A., Schulte, R., Opio, C., de Boer, I., 2014. Nutrient use efficiency: a valuable approach to benchmark the sustainability of nutrient use in global livestock production? Curr Opin Environ Sustain 9–10, 122–130. https://doi.org/10.1016/j.cosust.2014.09.007

Godinot, O., Leterme, P., Vertès, F., Faverdin, P., Carof, M., 2015. Relative nitrogen efficiency, a new indicator to assess crop livestock farming systems. Agron Sustain Dev 35, 857–868. https://doi.org/10.1007/s13593-015-0281-6

Goulding, K., Jarvis, S., Whitmore, A., 2008. Optimizing nutrient management for farm systems. Philosophical Transactions of the Royal Society B: Biological Sciences 363, 667–680. https://doi.org/10.1098/rstb.2007.2177

Grahmann, K., Rubio Dellepiane, V., Terra, J.A., Quincke, J.A., 2020. Long-term observations in contrasting crop-pasture rotations over half a century:

Statistical analysis of chemical soil properties and implications for soil sampling frequency. Agric Ecosyst Environ 287, 106710. https://doi.org/10.1016/j.agee.2019.106710

Groenestein, C.M., Hutchings, N.J., Haenel, H.D., Amon, B., Menzi, H., Mikkelsen, M.H., Misselbrook, T.H., van Bruggen, C., Kupper, T., Webb, J., 2019. Comparison of ammonia emissions related to nitrogen use efficiency of livestock production in Europe. J Clean Prod 211, 1162–1170. https://doi.org/10.1016/j.jclepro.2018.11.143

Gu, B., Ju, X., Chang, S.X., Ge, Y., Chang, J., 2017. Nitrogen use efficiencies in Chinese agricultural systems and implications for food security and environmental protection. Reg Environ Change 17, 1217–1227. https://doi.org/10.1007/s10113-016-1101-5

Haughey, E., Neogi, S., Portugal-Pereira, J., van Diemen, R., Slade, R.B., 2023. Sustainable intensification and carbon sequestration research in agricultural systems: A systematic review. Environ Sci Policy 143, 14–23. https://doi.org/10.1016/j.envsci.2023.02.018

Hutchings, N.J., Sørensen, P., Cordovil, C.M. d. S., Leip, A., Amon, B., 2020. Measures to increase the nitrogen use efficiency of European agricultural production. Glob Food Sec 26, 100381. https://doi.org/10.1016/j.gfs.2020.100381

International Fertilizer Industry Association., 2007. Fertilizer best management practices: general principles, strategy for their adoption and voluntary initiatives vs regulations : papers presented at the IFA international workshop on fertilizer best management practices, 7-9 March 2007, Brussels, Belgium. International Fertilizer Industry Association.

Jin, X., Zhang, N., Zhao, Z., Bai, Z., Ma, L., 2021. Nitrogen budgets of contrasting crop-livestock systems in China. Environmental Pollution 288, 117633. https://doi.org/10.1016/j.envpol.2021.117633

Lemaire, G., Garnier, J., da Silveira Pontes, L., de Faccio Carvalho, P.C., Billen, G., Simioni Assmann, T., 2023. Domestic Herbivores, the Crucial Trophic Level for Sustainable Agriculture: Avenues for Reconnecting Livestock to Cropping Systems. Agronomy 13, 982. https://doi.org/10.3390/agronomy13040982

Löw, P., Nadi Karatay, Y., Osterburg, B., 2020. Erratum: Nitrogen use efficiency on dairy farms with different grazing systems in northwestern Germany (2020 Environ. Res. Commun. 2 105002). Environ Res Commun 2, 119601. https://doi.org/10.1088/2515-7620/abccbc

MGAP, 2020. Normativa de Suelos y Aguas. https://www.gub.uy/ministerio-ganaderia-agricultura-pesca/politicas-y-gestion/normativa-suelos-aguas Uruguay.

Milroy, S.P., Wang, P., Sadras, V.O., 2019. Defining upper limits of nitrogen uptake and nitrogen use efficiency of potato in response to crop N supply. Field Crops Res 239, 38–46. https://doi.org/10.1016/j.fcr.2019.05.011

OECD/FAO, 2022. OECD-FAO Agricultural Outlook 2022-2031. OECD Publishing, Paris. https://doi.org/10.1787/f1b0b29c-en

Oliveira, J.G., Luiz Santana Júnior, M., Jaqueline Costa Maia, N., Batista Dubeux Junior, J.C., Hauber Gameiro, A., Kunrath, T.R., Geraldi Mendonça, G., Fernanda Simili, F., 2022. Nitrogen balance and efficiency as indicators for monitoring the proper use of fertilizers in agricultural and livestock systems. Sci Rep 12, 12021. https://doi.org/10.1038/s41598-022-15615-7

Paruelo, J.M., Sierra, M., 2023. Sustainable intensification and ecosystem services: how to connect them in agricultural systems of southern South America. J Environ Stud Sci 13, 198–206. https://doi.org/10.1007/s13412-022-00791-9

Pereyra Goday, F., Jebari, A., Takahashi, T., Rovira, P., Ayala, W., Lee, M. R. F., Rivero, M. J., McAuliffe, G. A., 2024. Carbon footprint of mixed farming crop-livestock rotational-based grazing beef systems using long term experimental data. Agron Sustain Dev 44(4), 41. https://doi.org/10.1007/s13593-024-00977-1

Pereyra Goday, F., Rovira, P., Ayala, W., Rivero, M.J., 2022. Management and Productivity of Key Integrated Crop–Livestock Systems in Uruguay: The Palo a Pique Long-Term Experiment's Third Phase. Agronomy 12, 3023. https://doi.org/10.3390/agronomy12123023

Plan Agropecuario, 2021. La invernada en sistemas pastoriles. Una mirada conceptual desde los productores.

Powell, J.M., Rotz, C.A., 2015. Measures of Nitrogen Use Efficiency and Nitrogen Loss from Dairy Production Systems. J Environ Qual 44, 336–344. https://doi.org/10.2134/jeq2014.07.0299

Pravia, M.V., Kemanian, A.R., Terra, J.A., Shi, Y., Macedo, I., Goslee, S., 2019. Soil carbon saturation, productivity, and carbon and nitrogen cycling in crop-pasture rotations. Agric Syst 171, 13–22. https://doi.org/10.1016/J.AGSY.2018.11.001

Quemada, M., Lassaletta, L., Jensen, L.S., Godinot, O., Brentrup, F., Buckley, C., Foray, S., Hvid, S.K., Oenema, J., Richards, K.G., Oenema, O., 2020. Exploring nitrogen indicators of farm performance among farm types across several European case studies. Agric Syst 177, 102689. https://doi.org/10.1016/j.agsy.2019.102689

Rawtani, D., Gupta, G., Khatri, N., Rao, P.K., Hussain, C.M., 2022. Environmental damages due to war in Ukraine: A perspective. Science of The Total Environment 850, 157932. https://doi.org/10.1016/j.scitotenv.2022.157932

Rivero, M.J., Lopez-Villalobos, N., Evans, A., Berndt, A., Cartmill, A., Neal, A.L., McLaren, A., Farruggia, A., Mignolet, C., Chadwick, D., Styles, D., McCracken, D., Busch, D., Martin, G.B., Fleming, H., Sheridan, H., Gibbons, J., Merbold, L., Eisler, M., Lambe, N., Rovira, P., Harris, P., Murphy, P., Vercoe, P.E., Williams, P., Machado, R., Takahashi, T., Puech, T., Boland, T., Ayala, W., Lee, M.R.F., 2021. Key traits for ruminant livestock across diverse production systems in the context of climate change: perspectives from a global platform of research farms. Reprod Fertil Dev 33, 1. https://doi.org/10.1071/RD20205

Rose, M., Pahlmann, I., Kage, H., 2023. Modified crop rotations for a sustainable intensification? A case study in a high-yielding environment with recurrent nitrogen surplus. European Journal of Agronomy 142, 126644. https://doi.org/10.1016/j.eja.2022.126644

Rovira, P., Ayala, W., Terra, J., García-Préchac, F., Harris, P., Lee, M.R.F., Rivero, M.J., 2020. The 'Palo a Pique' long-term research platform: First 25 years of a crop–livestock experiment in Uruguay. Agronomy 10. https://doi.org/10.3390/agronomy10030441

Ryschawy, J., Choisis, N., Choisis, J.P., Joannon, A., Gibon, A., 2012. Mixed crop-livestock systems: an economic and environmental-friendly way of farming? animal 6, 1722–1730. https://doi.org/10.1017/S1751731112000675

Salvagiotti, F., Collino, D.J., Perticari, A., Piccinetti, C., Ovando, G., Urquiaga, S., Racca, R.W., 2015. El aporte de la fijación biológica de nitrógeno en el cultivo de soja en Argentina. IAH.

Segura, C., Horrocks, C., Lopez-Aizpun, M., Blackwell, M.S.A., Darch, T., Hood, J., Le Cocq, K., McAuliffe, G.A., Lee, M.R.F., Cardenas, L., 2023. Response of soil health indicators to dung, urine and mineral fertilizer application in temperate pastures. J Environ Manage 330, 117096. https://doi.org/10.1016/j.jenvman.2022.117096

Sekaran, U., Lai, L., Ussiri, D.A.N., Kumar, S., Clay, S., 2021. Role of integrated crop-livestock systems in improving agriculture production and addressing food security – A review. J Agric Food Res. https://doi.org/10.1016/j.jafr.2021.100190

Shen, H., Gao, Y., Sun, K., Gu, Y., Ma, X., 2023. Effects of differential irrigation and nitrogen reduction replacement on winter wheat yield and water productivity and nitrogen-use efficiency. Agric Water Manag 282, 108289. https://doi.org/10.1016/j.agwat.2023.108289

Soria-Lopez, A., Garcia-Perez, P., Carpena, M., Garcia-Oliveira, P., Otero, P., Fraga-Corral, M., Cao, H., Prieto, M.A., Simal-Gandara, J., 2023. Challenges for future food systems: From the Green Revolution to food supply chains with a special focus on sustainability. Food Front 4, 9–20. https://doi.org/10.1002/fft2.173

Spiertz, J.H.J., 2010. Nitrogen, sustainable agriculture and food security. A review. Agron Sustain Dev 30, 43–55. https://doi.org/10.1051/agro:2008064

Sung, J., Kim, W., Oh, T.-K., So, Y.-S., 2023. Nitrogen (N) use efficiency and yield in rice under varying types and rates of N source: chemical fertilizer, livestock manure compost and food waste-livestock manure compost. Appl Biol Chem 66, 4. https://doi.org/10.1186/s13765-022-00766-y

Szymczak, L.S., de Moraes, A., Sulc, R.M., Barker, D., Monteiro, A.L.G., Lang, C.R., Moraes, R.F., Lemaire, G., de Faccio Carvalho, P.C., 2023. Convergence points of optimal herbage accumulation and intake rate by sheep grazing tall fescue. Grass and Forage Science 78, 578–589. https://doi.org/10.1111/gfs.12630

Terra, J., García-Préchac, F., Salvo, L., Hernández, J., 2006. Soil use intensity impact on total and particulate soil organic matter under no till crop pasture rotations under direct grazing. Advances in GeoEcology 38, 233–241.

Tilman, D., Cassman, K.G., Matson, P.A., Naylor, R., Polasky, S., 2002. Agricultural sustainability and intensive production practices. Nature 418, 671–677. https://doi.org/10.1038/nature01014

Uruguay XXI, 2021. Informe Anual Comercio Exterior.

USDA-NRCS, 1996. NSSC-SSL Report Uruguay. Soil characterization Data. . Washington DC.

Uwizeye, A., de Boer, I.J.M., Opio, C.I., Schulte, R.P.O., Falcucci, A., Tempio, G., Teillard, F., Casu, F., Rulli, M., Galloway, J.N., Leip, A., Erisman, J.W., Robinson, T.P., Steinfeld, H., Gerber, P.J., 2020. Nitrogen emissions along global livestock supply chains. Nat Food 1, 437–446. https://doi.org/10.1038/s43016-020-0113-y

van Eerdt, M.M., Fong, P.K.N., 1998. The monitoring of nitrogen surpluses from agriculture. Environmental Pollution 102, 227–233. https://doi.org/10.1016/S0269-7491(98)80037-7

Vingerhoets, R., Spiller, M., De Backer, J., Adriaens, A., Vlaeminck, S.E., Meers, E., 2023. Detailed nitrogen and phosphorus flow analysis, nutrient use efficiency and circularity in the agri-food system of a livestock-intensive region. J Clean Prod 410, 137278. https://doi.org/10.1016/j.jclepro.2023.137278

Ward, S.M., Holden, N.M., White, E.P., Oldfield, T.L., 2016. The “circular economy” applied to the agriculture (livestock production) sector-discussion paper.

Wilkinson, J.M., Lee, M.R.F., 2018. Review: Use of human-edible animal feeds by ruminant livestock. Animal 12, 1735–1743. https://doi.org/10.1017/S175173111700218X

Xue, H., Mainville, D., You, W., Nayga, R.M., 2010. Consumer preferences and willingness to pay for grass-fed beef: Empirical evidence from in-store experiments. Food Qual Prefer 21, 857–866. https://doi.org/10.1016/j.foodqual.2010.05.004

Yan, X., Xia, L., Ti, C., 2022. Temporal and spatial variations in nitrogen use efficiency of crop production in China. Environmental Pollution 293, 118496. https://doi.org/10.1016/j.envpol.2021.118496

5. Discusión general

El presente trabajo aborda la intensificación sostenible de los sistemas de producción a través del análisis de cuatro sistemas que combinan agricultura y ganadería con un nivel de intensificación de uso de suelo variable. A partir de la información obtenida en el experimento se calcularon diversos indicadores que fueron presentados oportunamente en cada uno de los capítulos que anteceden.

A modo resumen, en el primer capítulo se presentaron indicadores referidos a la productividad de los sistemas, en el segundo capítulo se resumen indicadores asociados con las emisiones de gases de efecto invernadero de los sistemas (perfiles de emisiones, contribución de cada proceso y huella de carbono parcial) y el tercer capítulo presenta resultados referidos a la eficiencia de uso de nitrógeno para cada uno de los sistemas y sus componentes.

En esta discusión se plantea abordar las dimensiones económica y social de la sustentabilidad a través de la contribución de estos sistemas a la seguridad alimentaria, así como también integrar los resultados de la dimensión ambiental.

5.1. Análisis económico

Este análisis fue elaborado en conjunto con el Ing. Agr. Enrique Fernández (Unidad de Economía Aplicada de INIA), para su presentación en el Seminario Claves para la Intensificación Sostenible de la Ganadería y la Agricultura en la Región Este, realizado el 28 de julio de 2023 en INIA Treinta y Tres.

5.1.1. Supuestos utilizados

Se utilizó la base de datos del experimento considerando la totalidad de los insumos utilizados (semillas, agroquímicos, fertilizantes, suplemento animal). Los precios de los insumos fueron obtenidos directamente de las facturas de compra. Las labores agrícolas también fueron consideradas dentro de los costos de producción. Para esto se consultó la página web de CUSA (Cámara Uruguaya de Servicios Agropecuarios).

Para el caso de los granos y carne vendidos, se tomaron los valores de facturación. En aquellos sistemas en que los animales pasaban de un año para el otro (RL, RF) se consideraron los pesos de ese momento y se valorizaron considerando los precios de los remates por pantalla de ese mes, incluyendo impuestos y comisión por venta. De la misma forma se procedió en aquellos sistemas donde el producto final no son animales a faena (RA, RC). La producción de fardos en el componente agrícola se consideró vendida al componente ganadero, mientras que el sorgo grano húmedo se valorizó de acuerdo con el precio de planilla sumando además el embolsado. Se consideró también el costo correspondiente a la sanidad, fijado en US$ por animal, de acuerdo con la categoría animal presente en los sistemas (recría, engorde o recría + engorde).

Para este análisis se consideró como superficie agrícola el primer año de la fase de pastura en RC y RL, ya que las praderas se siembran consociadas con trigo. Por lo tanto, el área agrícola fue de 6, 9 y 9 ha para RA, RC y RL, respectivamente.

5.1.1.1. Resultados subsistema agrícola

La tabla 1 resume el resultado económico obtenido para las rotaciones que incluyen producción agrícola.

Tabla 1

Costos, ingresos y margen bruto agrícola, por año y rotación.

Sistema Año de evaluación	Rotación agrícola			Rotación corta			Rotación larga		
	2019-2020	2020-2021	2021-2022	2019-2020	2020-2021	2021-2022	2019-2020	2020-2021	2021-2022
Área agrícola (ha)	6	6	6	9	9	9	9	9	9
	Costos agrícolas								
Insumos	1998	2686	3037	2440	3256	3508	2394	3416	3802
Maquinaria	1762	1752	1323	2152	2136	1790	2248	2508	1933
Reservas	191	539	632	1128	823	829	738	427	755
Total de costos	3951	4976	4993	5719	6215	6127	5380	6350	6490
	Ingresos agrícolas								
Venta de sorgo GH	1919	3119	0	2102	3778	0	2223	3236	0
Venta de fardos	0	275	930	1320	975	1650	800	725	1380
Venta de granos	2623	5738	3557	3453	7625	3822	3131	8404	5106
Total ingresos	4542	9132	4487	6875	12378	5472	6154	12365	6486
MB/ha	99	693	-84	128	685	-73	86	668	0
Promedio (2019-2022)		236			247			251	

Los tres años estudiados fueron distintos entre sí por diferentes razones, lo cual determinó diferencias en el margen bruto obtenido por año. Sin embargo, al promediar los valores (2019-2022), estos son similares entre rotaciones, con un año de muy buenos valores (2020-2021), un año de márgenes negativos (2021-2022), asociado a rendimientos de grano muy bajos por condiciones climáticas, y, por último, un año con rendimientos promedio y márgenes intermedios (2019-2020).

5.1.1.2. Resultados subsistema ganadero

Los resultados económicos correspondientes al subsistema ganadero se presentan en la tabla 2.

Tabla 2

Costos, ingresos y margen bruto ganadero, por año y rotación.

Sistema/año	Rotación agrícola			Rotación corta			Rotación larga			Rotación forrajera		
	2019-2020	2020-2021	2021-2022	2019-2020	2020-2021	2021-2022	2019-2020	2020-2021	2021-2022	2019-2020	2020-2021	2021-2022
Ha ganadería	15	15	15	22	22	22	44	44	44	31	31	31
	Costos ganadería											
Insumos	2368	3485	3839	3241	3779	4433	3557	4977	4879	6797	6210	8543
Maquinaria	1276	1354	1675	1699	1538	1569	1730	1598	1642	1075	1051	1075
Sanidad	163	154	168	245	265	245	477	502	561	221	180	235
Suplementos	722	41	1968	2462	1338	1995	5386	10582	10856	0	5218	0
Total costos	4528	5034	7650	7646	6921	8242	11150	17659	17938	8093	12659	9853
	Ingresos ganadería											
PB Ganadero	4993	6420	14978	4922	10771	20827	18980	20311	47718	9102	14324	28433
MB/ha	31	92	489	-124	175	572	178	60	677	33	54	599
Promedio (2019-2022)	204			208			305			229		

En contraposición a lo que sucedió en el subsistema agrícola, los mejores resultados en términos económicos se obtuvieron durante el tercer año, lo que se explica fundamentalmente por los altos precios de venta de ganado. Con respecto al año anterior (2020-2021), el margen bruto se incrementó 5 veces, 3 veces, 11 veces y 11 veces para RA, RC, RL y RF respectivamente. Esta diferencia entre años es proporcionalmente mayor en aquellos sistemas con mayor peso del engorde con respecto a la recría.

5.1.1.3. Resultado global de los sistemas

En promedio, el margen bruto por hectárea total de sistema fue 285, 200, 181 y 197 US$/ha para RL, RC, RA y RF, respectivamente. Aquellos sistemas con fase de pastura y componente agrícola que combinaron recría con engorde obtuvieron mejor resultado económico que la rotación que exclusivamente engordó animales sin componente agrícola y aquella que recrió animales sin fase de pasturas. La mayor variabilidad medida a través del coeficiente de variación (CV) fue observada en RF (140,6 %), mientras que el menor CV se observó en RA (68,4 %). RL y RC tuvieron valores intermedios de 75 % y 107,5 %, respectivamente. Estos valores de CV se explican por las variaciones importantes en los precios tanto de venta de animales y granos como de compra de insumos.

De acuerdo con los márgenes obtenidos y la intensidad de emisiones (kg de CO_2 eq/ha), presentadas en el capítulo 2, en promedio, para obtener un dólar de margen bruto, los sistemas emitieron 15,4, 13,7, 9,6 y 13,2 kg CO_2 eq, para RA, RC, RL y RF, respectivamente. En términos generales, los valores no presentaron grandes diferencias, lo que se explica por las compensaciones (*trade-offs*) entre los componentes del sistema (rotación-estrategia ganadera). Sin embargo, se observó una tendencia a que aquellos sistemas con un mayor largo de fase de pastura emitieran menos kg de CO_2 por dólar de margen bruto, es decir, fueron más eficientes ambientalmente, aun cuando la estrategia ganadera utilizada fue menos eficiente.

Asociado al manejo de la fertilización nitrogenada de los sistemas y su eficiencia, presentados en el capítulo 3, para generar un dólar de margen bruto, se utilizaron 0,21, 0,26, 0,09 y 0,58 kg de N sintético en RA, RC, RL y RF, respectivamente. Dado que este indicador involucra el resultado económico de los sistemas y que las diferencias en resultado económico agrícola fueron muy bajas, una proporción importante del resultado obtenido puede explicarse por el subsistema ganadero y por la estrategia ganadera elegida para cada sistema. En este sentido, el resultado observado en RF se explica por un elevado uso de fertilizantes nitrogenados en el área sembrada con festuca (184 kg N/ha/año), asociado a un manejo posterior que no logra capitalizar en su totalidad la producción de biomasa en peso vivo (producción de 300 kg PV/ha promedio, eficiencia de conversión promedio 19,2 kg MS/kg PV). En el otro extremo, RL explica sus resultados por niveles de producción más altos (369,4 kg PV/ha promedio) y un menor uso de fertilizantes nitrogenados (25,9 kg/ha promedio) y una fase de pastura de cuatro años de gramínea y leguminosas. En este sistema, el aporte de N por fijación biológica de las leguminosas se estimó en 27,8 kg N/ha/año. RC y RA tuvieron un comportamiento intermedio, lo cual era esperable, dado que estos sistemas se componen de una fase corta de pasturas y cultivo continuo, respectivamente, con mayor uso de fertilizantes por hectárea (51 y 37 kg/ha). En RC, el componente leguminosa aportó 52,6 kg/ha, lo que se explica fundamentalmente por la alta productividad observada en trébol rojo, mientras que en RA el aporte fue de 4,6 kg/ha.

5.2. Contribución a la seguridad alimentaria

Se evaluó la contribución a la seguridad alimentaria a través de la producción de proteína consumible (HEP, *human edible protein*) y la producción de energía consumible (HEE, *human edible energy*) de acuerdo con la metodología propuesta por Mosnier et al. (2021). Este tipo de análisis permite evaluar los sistemas de producción desde el punto de vista de la demanda de productos.

5.2.1. Supuestos utilizados y cálculos

Para los cálculos detallados en la tabla 3, se utilizaron los valores de producción de carne y granos reportados en el capítulo 1. Se asumió que las categorías de recría salían directo a faena.

Tabla 3

Método de cálculo para proteína consumible (HEP) y energía consumible (HEE) en carne y granos.

Fórmulas utilizadas para el cálculo de HEP[1] y HEE[2]	
Tipo de producto	Fórmula
Carne	HEP producida = kg de carne producidos × contenido de proteína bruta[3] × proporción consumible de la proteína total[4] HEE producida = kg de carne producida × contenido de energía bruta[5] × proporción consumible de la energía total[6]
Granos (soja, avena, trigo)	HEP producida = kg de grano producidos × contenido de proteína bruta[7]× proporción consumible de la proteína total[8] HEE producida = kg de grano producidos × contenido de energía[9] × proporción consumible de la energía total[10]

Nota. [1]HEP: proteína consumible humana; [2]HEE: energía consumible humana. [3]Contenido estimado de proteína bruta por kg de carne. [4]Proporción de la proteína total en carne que es consumible Laisse et al. (2019). [5]Contenido estimado de energía por kg de carne. [6]Proporción de la energía total en carne que es consumible (Mosnier et al., 2021). [7]Contenido estimado de proteína por kg de grano para soja, avena y trigo. [8]Proporción de la proteína total en grano que es consumible Laisse et al. (2019). [9]Contenido estimado de energía por kg de grano para soja, avena y trigo. [10]Proporción de la energía total en grano que es consumible (Mosnier et al., 2021).

La tabla 4 detalla los parámetros y coeficientes técnicos utilizados para cada uno de los cálculos, así como también la fuente de cada uno de los datos.

Tabla 4

Parámetros utilizados para los cálculos de proteína consumible (HEP) y energía consumible (HEE).

Parámetro	Rotación agrícola	Rotación corta	Rotación larga	Rotación forrajera	Fuente
kg carne/ kg PV[1]	0,53	0,53	0,53	0,53	Base de datos UEPP
kg PB/kg de carne[2]	0,158	0,158	0,158	0,158	Laisse et al. (2019)
EB/kg de carne[3]	10,9	10,9	10,9	10,9	Laisse et al. (2019)
Proporción consumible PB en carne	0,60	0,60	0,56	0,56	Mosnier et al. (2021)
Proporción consumible EB en carne	0,34	0,34	0,325	0,325	Mosnier et al. (2021)
kg PB/kg de soja	0,36	0,36	0,36	-	Base de datos UEPP
kg PB/kg de avena	0,108	0,108	0,108	-	Base de datos UEPP
kg PB/kg de trigo	0,126	0,126	0,126	-	Base de datos UEPP
EB/kg de soja	29,8	29,8	29,8	-	Mosnier et al. (2021)
EB/kg de avena	19,5	19,5	19,5	-	Mosnier et al. (2021)
EB/kg de trigo	18,3	18,3	18,3	-	Mosnier et al. (2021)
Proporción consumible PB en soja	0,60	0,60	0,60	-	Mosnier et al. (2021)
Proporción consumible PB en avena	0,84	0,84	0,84	-	Mosnier et al. (2021)
Proporción consumible PB en trigo	0,66	0,66	0,66	-	Mosnier et al. (2021)
Proporción consumible EB en soja	0,38	0,38	0,38	-	Mosnier et al. (2021)
Proporción consumible EB en avena	0,79	0,79	0,79	-	Mosnier et al. (2021)
Proporción consumible EB en trigo	0,67	0,67	0,67	-	Mosnier et al. (2021)

Nota. [1]PV: peso vivo; [2]PB: proteína bruta; [3]EB: Energía bruta (medida en Megajules por kg de carne o grano).

5.2.2. Resultados obtenidos

La producción de HEP fue, en promedio, para los años evaluados (2019-2022) 123,4, 83,7, 60,4 y 12,1 kg/ha para RA, RC, RL y RF, respectivamente. El CV en cada uno de los cálculos fue de 19,8%, 27,3%, 26,3% y 6% para RA, RC, RL y RF, respectivamente. La producción de HEE fue 8173, 6595, 4557 y 486 MJ/ha para RA, RC, RL y RF, respectivamente. La variación entre años medida a través del CV fue de 46,5%, 53,8%, 45,3% y 6% para RA, RC, RL y RF, respectivamente.

La proporción de proteína y energía producida por cada uno de los componentes varió entre los sistemas. A mayor largo de la fase de pastura, el aporte del componente agrícola disminuye con respecto al componente ganadero. Las proporciones de aporte por componente (agrícola-ganadero) para HEP fueron 85%-15%, 83%-17% y 77%-23%, mientras que en HEE las proporciones fueron 90 %-10%, 91%-9% y 87%-13% para RA, RC y RL, respectivamente.

La mayor contribución en producción de proteína y energía por hectárea se observó en aquellos sistemas con un uso de suelo más intensivo, lo que se explica por una mayor proporción de área destinada a la producción de grano, lo cual presenta una mayor eficiencia comparado con la producción de carne, y por una mayor proporción de recría con respecto a engorde. Esto coincide con resultados obtenidos por Mosnier et al. (2021), quienes reportaron que los sistemas especializados en producción de carne producen significativamente menos proteína y energía por hectárea en comparación con aquellos que incluyen producción de granos. Los mismos autores reportan producciones de HEP variables entre 43 y 370 kg /ha y producciones de HEE de 1600 a 60000 MJ/ha para diferentes sistemas de producción europeos. Estos autores proponen como punto de mejora para estos indicadores evitar el uso de alimentos consumibles por humanos en la alimentación animal, lo cual se alinea con el manejo realizado en Palo a Pique, donde el uso de alimentos consumibles por humanos es nulo (Rovira et al., 2020). Otra estrategia de mejora propuesta es el incremento de la productividad de biomasa de los sistemas y el uso de razas más eficientes en términos de conversión de forraje a carne, lo cual tendría, además, implicancias en las emisiones de metano de los animales (Navajas et al., 2022).

Si bien los sistemas que incluyeron fase de pastura en la rotación presentaron niveles más bajos de producción de HEP y HEE por ha, es importante realizar otras consideraciones. Por un lado, se trata de sistemas que no utilizan alimentos consumibles por humanos para alimentación del ganado; dichos sistemas serían productores netos de proteína consumible por humanos, lo cual implica que no haya competencia por recursos (*feed-food competition*) (Wilkinson y Lee 2018). Por otro lado, como se mencionó en la introducción, la incorporación del componente ganadero provee diversos servicios ecosistémicos tales como reciclaje de nutrientes, conservación de biodiversidad y secuestro de carbono (Hennessy et al., 2021). Un tercer aspecto para tener en cuenta es la calidad nutricional del producto carne obtenido en estos sistemas, es decir, realizar una valoración desde el punto de vista de la composición nutricional y su contribución a la nutrición humana, además del total de proteína o energía. En este sentido, la aplicación del nLCA (análisis de ciclo de vida nutricional), donde el suministro de nutrientes se considera una de las funciones principales del alimento (McLaren et al., 2021), sería una alternativa para realizar esta evaluación (McAuliffe et al., 2023).

5.2.3. Implicancias para los sistemas de producción

La tabla 5 resume los principales indicadores calculados para cada uno de los sistemas evaluados, a lo largo de este trabajo. Los colores utilizados en cada celda se asocian a la lógica del semáforo, donde aquellos valores que se ubican por fuera de los límites establecidos como óptimos se encuentran en color rojo, valores intermedios en color amarillo y valores óptimos en color verde. Aquellos indicadores en los que no se detectaron diferencias significativas o no se realizó análisis estadístico se encuentran en color gris.

INDICADOR	ROTACIÓN AGRÍCOLA	ROTACIÓN CORTA	ROTACIÓN LARGA	ROTACIÓN FORRAJERA
Producción PV por ha (kg PV/ha)	426	418	369	310
Producción de grano (soja/ trigo/avena/ sorgo) (t/ha)	2,4/ 2,21/-/4,97	2,48/2,68/1,81/5,65	2,76/2,59/2,18/5,29	-
Producción de forraje (kg MS/ha)	5206	5763	5399	6867
Emisiones por kg PV (kg CO_2 eq/kg PV)	11,3	11,8	11,8	16,4
Emisiones por ha (kg CO_2 eq/ha)	2795	2734	2727	2607
Emisiones por kg de grano (kg CO_2 eq/ kg grano)	1,36/0,61/-	1,24/0,54/0,57	1,01/0,43/0,47	-
Eficiencia de uso de N (agrícola)	62,5	83,8	77,5	-
Eficiencia de uso de N (ganadería)	24,4	9,9	14,3	5,5
Margen Bruto por ha (sistema) (US$/ha)	181	200	285	197
Emisiones por US$ de margen bruto (kg CO_2 eq/US$)	15,4	13,7	9,6	13,2
Producción de HEP y HEE por ha (kg/ha)	Alta	Media	Media	Baja
Pérdidas de suelo por ha (t/ha)	7,16	4,11	3,17	2,49

Figura 1

Principales indicadores calculados para los sistemas evaluados.

Los resultados obtenidos a partir de los análisis realizados permiten caracterizar los sistemas evaluados a través de diferentes indicadores. En términos generales, el sistema que no incluyó pasturas, asociado a una estrategia ganadera de recría (RA), presentó como ventaja frente a los otros sistemas mayores niveles de producción por hectárea y una alta eficiencia de uso de nitrógeno, cerca de los límites máximos óptimos. La incorporación de una fase de pastura (corta o larga) y las estrategias ganaderas asociadas que combinan recría y engorde (RC y RL) permitieron mejorar los indicadores de resultado económico y disminuir la intensidad de emisiones (kg CO_2 eq/ha) de los sistemas, así como también mejorar contenido de nutrientes en suelo y capitalizar el aporte de nitrógeno de las leguminosas a partir de la fijación bilógica de nitrógeno. Por otro lado, el sistema que no incluyó componente agrícola (RF) tuvo la menor intensidad de emisiones por ha y altos niveles de producción de biomasa, lo cual permitió una evolución positiva en el contenido de carbono y nitrógeno de suelo.

Por otra parte, RA presentó altos costos de producción y bajo margen bruto con respecto a los otros sistemas. En los sistemas con mayor proporción de engorde respecto a la recría (RC y RL) la productividad por hectárea y la eficiencia de uso de nitrógeno disminuyó. Por otro lado, RF presentó bajos valores de eficiencia de uso de nitrógeno —lo cual incrementaría el riesgo de pérdidas, al obtener valores de nitrógeno excedente (NSURP) por encima de los umbrales recomendados (Gerber et al., 2014)—, así como también bajos niveles de producción de peso vivo por hectárea, comparado con el resto de los sistemas. En este sentido, en el capítulo 4 se plantean diferentes estrategias a implementar en cada uno de los sistemas con el objetivo de optimizar la eficiencia de uso de nitrógeno, así como también mejorar indicadores productivos.

Otro aspecto para considerar en los sistemas agrícolas es la reglamentación vigente para uso de suelo (Ministerio de Ganadería Agricultura y Pesca, 2020), la cual expresa que las rotaciones de cultivos deben tener pérdidas anuales menores al límite establecido para el tipo de suelo correspondiente (Estación Sanz; argisol subéutrico melánico Abr F hid.); en este caso, 7 Mg/ha. Es decir que la rotación que no incluyó pasturas, si bien tuvo mayores niveles de producción en términos de proteína y energía consumible (HEP, HEE), estaría por encima de las pérdidas de suelo toleradas para ese tipo de suelo. Por lo tanto, este sistema sería insostenible en el largo plazo desde el punto de vista de las pérdidas de suelos, lo cual haría inviable su implementación.

6. Conclusiones

La combinación de sistemas de producción agrícola-ganaderos que incluyen una fase de pastura (corta o larga) en la rotación es una opción válida de intensificación para la región este del país. Al mismo tiempo, dicha combinación permite mejorar diversos indicadores productivos.

Específicamente, el sistema en el que se observó un mejor desempeño del conjunto de indicadores fue RL. Además, en este sistema no se observaron valores de indicadores por fuera de los rangos estipulados como óptimos para eficiencia de uso de nitrógeno y pérdidas toleradas de suelo.

En RC y RF, si bien se observaron valores por fuera de los rangos óptimos, estos pueden ser ajustados a través de cambios en el manejo de los sistemas, lo cual tendría implicancias en otros indicadores, por ejemplo resultado económico.

La utilización de experimentos de largo plazo permite evaluar efectos que en experimentos de corta duración no serían observables; por ejemplo, parámetros de calidad de suelos, variabilidad climática, entre otros. Asimismo, por tratarse de un experimento de escala semicomercial, es posible observar efectos de manejo similares a los observables en predios comerciales; por ejemplo, uso de maquinaria o pisoteo animal.

Respecto a los resultados obtenidos en el estudio y al horizonte temporal utilizado (tres años), sería deseable incluir un mayor número de años en el análisis, dado que se observó una alta variabilidad en los resultados, explicada fundamentalmente por razones climáticas y de coyuntura económica.

A futuro, sería relevante profundizar en algunos aspectos planteados en este trabajo tales como aplicación de análisis de ciclo de vida nutricional con enfoque en la nutrición humana y optimización del manejo del nitrógeno, considerando el concepto de eficiencia de uso.

7. Bibliografía

Ajibade, S., Simon, B., Gulyas, M. y Balint, C. (2023). Sustainable intensification of agriculture as a tool to promote food security: A bibliometric analysis. *Frontiers in Sustainable Food Systems*, 7. https://doi.org/10.3389/fsufs.2023.1101528

Arístide, P., Cittadini, E., Blumetto, O., Giobellina, E., Ledesma, S., Ovalle, C., Marchao, R., Caballero, P. J., Osman, A. y Tittonell, P. (2020). *Variables claves para la evaluación de la sustentabilidad de los sistemas agropecuarios: hacia un sistema de indicadores de intensificación sostenible en el Cono Sur.* PROCISUR.

Campanhola, C. y Pandey, S. (eds.). (2019). Context for Sustainable Intensification of Agriculture. *Sustainable Food and Agriculture* (pp. 171-172). Elsevier. https://doi.org/10.1016/B978-0-12-812134-4.00010-8

Cazzulli, F. y Paruelo, J (eds.). (2023). *Indicadores de desempeño ambiental para sistemas agropecuarios del Uruguay*. INIA.

Cortner, O., Garrett, R. D., Valentim, J. F., Ferreira, J., Niles, M. T., Reis, J. y Gil, J. (2019). Perceptions of integrated crop-livestock systems for sustainable intensification in the Brazilian Amazon. *Land Use Policy*, *82*, 841-853. https://doi.org/10.1016/j.landusepol.2019.01.006

De Faccio Carvalho, P. C., Anghinoni, I., de Moraes, A., De Souza, E. D., Sulc, R. M., Lang, C. R., Flores, J. P. C., Terra Lopes, M. L., Da Silva, J. L. S., Conte, O., De Lima Wesp, C., Levien, R., Fontaneli, R. S. y Bayer, C. (2010). Managing grazing animals to achieve nutrient cycling and soil improvement in no-till integrated systems. *Nutrient Cycling in Agroecosystems*, *88*(2), 259-273. https://doi.org/10.1007/s10705-010-9360-x

De Faccio Carvalho, P. C., Savian, J. V., Chiesa, T. Della, De Souza Filho, W., Terra, J. A., Pinto, P., Martins, A. P., Villarino, S., Da Trindade, J. K., De Albuquerque Nunes, P. A. y Pineiro, G. (2021). Land-Use Intensification

Trends in the Rio de la Plata Region of South America: Toward Specialization or Recoupling Crop and Livestock Production. *Frontiers of Agricultural Science and Engineering*, *8*(1), 97-110. https://doi.org/10.15302/J-FASE-2020380

Denardin, L. G. de O., Martins, A. P., Bastos, L. M., Ciampitti, I. A., Anghinoni, I., Moojen, F. G., Carvalho, P. C. de F., Huang, M. y Chabbi, A. (2020). Soybean Yield Does Not Rely on Mineral Fertilizer in Rotation with Flooded Rice under a No-Till Integrated Crop-Livestock System. *Agronomy*, *10*(9), 1371. https://doi.org/10.3390/agronomy10091371

Diaz Zorita, M., Duarte, G. A. y Grove, J. H. (2002). A review of no-till systems and soil management for sustainable crop production in the subhumid and semiarid Pampas of Argentina. *Soil & Tillage Research*, *65*, 1-18.

Dirección Estadísticas Agropecuarias-Ministerio de Ganadería, Agricultura y Pesca. (2022). Anuario estadístico agropecuario. Consultado 30 de julio de 2024. [En línea]. https://www.gub.uy/ministerio-ganaderia-agricultura-pesca/comunicacion/publicaciones/anuario-estadistico-agropecuario-2022

Eisler, M. C., Lee, M. R. F., Tarlton, J. F. y Martin, G. B. (2014). Steps to sustainable livestock. *Nature*, *507*(7490), 32-34. https://doi.org/10.1038/507032a

Flachowsky, G., Meyer, U. y Südekum, K.-H. (2017). Land Use for Edible Protein of Animal Origin—A Review. *Animals*, *7*(12), 25. https://doi.org/10.3390/ani7030025

García-Préchac, F., Ernst, O., Siri-Prieto, G. y Terra, J. A. (2004). Integrating no-till into crop–pasture rotations in Uruguay. *Soil and Tillage Research*, *77*(1), 1-13. https://doi.org/10.1016/j.still.2003.12.002

Garrett, R. D., Niles, M. T., Gil, J. D. B., Gaudin, A., Chaplin-Kramer, R., Assmann, A., Assmann, T. S., Brewer, K., De Faccio Carvalho, P. C., Cortner, O., Dynes, R., Garbach, K., Kebreab, E., Mueller, N., Peterson, C., Reis, J. C., Snow, V. y Valentim, J. (2017). Social and ecological analysis of commercial integrated crop livestock systems: Current knowledge and remaining uncertainty. *Agricultural Systems*, *155*, 136-146. https://doi.org/10.1016/j.agsy.2017.05.003

Gerber, P. J., Uwizeye, A., Schulte, R. P. O., Opio, C. I. y De Boer, I. J. M. (2014). Nutrient use efficiency: A valuable approach to benchmark the sustainability of nutrient use in global livestock production? *Current Opinion in Environmental Sustainability*, *9-10*, 122-130. https://doi.org/10.1016/j.cosust.2014.09.007

Haughey, E., Neogi, S., Portugal-Pereira, J., van Diemen, R. y Slade, R. B. (2023). Sustainable intensification and carbon sequestration research in agricultural systems: A systematic review. *Environmental Science & Policy*, *143*, 14-23. https://doi.org/10.1016/j.envsci.2023.02.018

Hennessy, D. P., Shalloo, L., van Zanten, H. H. E., Schop, M. y De Boer, I. J. M. (2021). The net contribution of livestock to the supply of human edible protein: the case of Ireland. *The Journal of Agricultural Science*, *159*(5-6), 463-471. https://doi.org/10.1017/S0021859621000642

Laisse, S., Baumont, R., Dusart, L., Gaudré, D., Rouillé, B., Benoit, M., Veysset, P., Rémond, D. y Peyraud, J.-L. (2019). L'efficience nette de conversion des aliments par les animaux d'élevage : une nouvelle approche pour évaluer la contribution de l'élevage à l'alimentation humaine. *INRA Productions Animales*, *31*(3), 269-288. https://doi.org/10.20870/productions-animales.2018.31.3.2355

Lee, M. R. F., Domingues, J. P., McAuliffe, G. A., Tichit, M., Accatino, F. y Takahashi, T. (2021). Nutrient provision capacity of alternative livestock farming systems per area of arable farmland required. *Scientific Reports*, *11*(1), 14975. https://doi.org/10.1038/s41598-021-93782-9

Mahon, N., Crute, I., Simmons, E. e Islam, Md. M. (2017). Sustainable intensification – "oxymoron" or "third-way"? A systematic review. *Ecological Indicators*, *74*, 73-97. https://doi.org/10.1016/j.ecolind.2016.11.001

McAuliffe, G. A., Takahashi, T., Lee, M. R. F., Jebari, A., Cardenas, L., Kumar, A., Pereyra Goday, F., Scalabrino, H. y Collins, A. L. (2023). A commentary on key methodological developments related to nutritional life cycle assessment generated throughout a 6-year strategic scientific programme. *Food and Energy Security*, *12*(4). https://doi.org/10.1002/fes3.480

McLaren, S., Berardy, A., Henderson, A., Holden, N., Huppertz, T., Jolliet, O., De Camillis, C., Renouf, M., Rugani, B., Saarinen, M., van der Pols, J., Vázquez-Rowe, I., Antón Vallejo, A., Bianchi, M., Chaudhary, A., Chen, C., CooremanAlgoed, M., Dong, H., Grant, T., … van Zanten, H. (2021) *Integration of environment and nutrition in life cycle assessment of food items: opportunities and challenges*. FAO.

Ministerio de Ganadería Agricultura y Pesca. (2020). *Normativa de suelos y aguas*. Consultado 30 de julio de 2024. [En linea] https://www.gub.uy/ministerio-ganaderia-agricultura-pesca/politicas-y-gestion/normativa-suelos-aguas.

Mosnier, C., Jarousse, A., Madrange, P., Balouzat, J., Guillier, M., Pirlo, G., Mertens, A., ORiordan, E., Pahmeyer, C., Hennart, S., Legein, L., Crosson, P., Kearney, M., Dimon, P., Bertozzi, C., Reding, E., Iacurto, M., Breen, J., Carè, S. y Veysset, P. (2021). Evaluation of the contribution of 16 European beef production systems to food security. *Agricultural Systems*, *190*, 103088. https://doi.org/10.1016/j.agsy.2021.103088

Navajas, E. A., Ravagnolo, O., De Barbieri, I., Pravia, M. I., Aguilar, I., Lema, M. O., Vera, B., Peraza, P., Marques, C. B., Velazco, J. I. y Ciappesoni, G. (2022). 29. Genetic selection of feed efficiency and methane emissions in sheep and cattle in Uruguay: progress and limitations. *Proceedings of 12th World Congress on Genetics Applied to Livestock Production* (WCGALP), 164-167. https://doi.org/10.3920/978-90-8686-940-4_29

Organisation for economic co-operation and development/Food and Agriculture Organisation. (2022). *OECD-FAO Agricultural Outlook 2022-2031*. OECD Publishing. https://doi.org/10.1787/f1b0b29c-en

Paruelo, J. M. y Sierra, M. (2023). Sustainable intensification and ecosystem services: how to connect them in agricultural systems of southern South America. *Journal of Environmental Studies and Sciences*, *13*(1), 198-206. https://doi.org/10.1007/s13412-022-00791-9

Peyraud, J. L., Taboada, M. y Delaby, L. (2014). Integrated crop and livestock systems in Western Europe and South America: A review. *European Journal of Agronomy*, *57*, 31-42. https://doi.org/10.1016/j.eja.2014.02.005

Poulton, P. R. (1996). The Rothamsted long-term experiments: Are they still relevant? *Canadian Journal of Plant Science*, 559-571. https://cdnsciencepub.com/doi/10.4141/cjps96-103

Ran, Y., Deutsch, L., Lannerstad, M. y Heinke, J. (2013). Rapidly Intensified Beef Production in Uruguay: Impacts on Water-related Ecosystem Services. *Aquatic Procedia*, *1*, 77-87. https://doi.org/10.1016/j.aqpro.2013.07.007

Rawtani, D., Gupta, G., Khatri, N., Rao, P. K. y Hussain, C. M. (2022). Environmental damages due to war in Ukraine: A perspective. *Science of the Total Environment*, *850*, 157932. https://doi.org/10.1016/j.scitotenv.2022.157932

Sanderson, M. A., Liebig, M. A., Hendrickson, J. R., Kronberg, S. L., Toledo, D., Derner, J. D. y Reeves, J. L. (2016). A century of grazing: The value of long-term research. *Journal of Soil and Water Conservation*, *71*(1), 5A-8A. https://doi.org/10.2489/jswc.71.1.5A

Sayre, N. F., deBuys, W., Bestelmeyer, B. T. y Havstad, K. M. (2012). “The Range Problem” After a Century of Rangeland Science: New Research Themes for Altered Landscapes. *Rangeland Ecology & Management, 65*(6), 545-552. https://doi.org/10.2111/REM-D-11-00113.1

Scott, J. M., Gaden, C. A., Edwards, C., Paull, D. R., Marchant, R., Hoad, J., Sutherland, H., Coventry, T. y Dutton, P. (2013). Selection of experimental treatments, methods used and evolution of management guidelines for comparing and measuring three grazed farmlet systems. *Animal Production Science*, *53*(7-8), 628-642. https://doi.org/10.1071/AN12265

Semmartin, M., Cosentino, D., Poggio, S. L., Benedit, B., Biganzoli, F. y Peper, A. (2023). Soil carbon accumulation in continuous cropping systems of the rolling Pampa (Argentina): The role of crop sequence, cover cropping and agronomic technology. *Agriculture, Ecosystems & Environment*, *347*, 108368. https://doi.org/10.1016/j.agee.2023.108368

Terra, J. (2017). Experimentos de largo plazo como plataforma agroambiental para la intensificación sostenible. *Revista INIA*, *48*, 67-72.

Terra, J. A., Garcia-Préchac, F. y Salvo, L. (2006). Soil use intensity impact on total and particulate soil organic matter in no till crop pasture rotations under direct grazing. *Advances in GeoEcology*, *38*, 233-241.

Terra, J. y Préchac, F. G. (2002). Soil organic carbon content of a typic argiudoll in Uruguay under forage crops and pastures for direct grazing: effect of tillage intensity and rotation system. En E. van Santen (ed.), *Making Conservation Tillage Conventional: Building a Future on 25 Years of Research*. Proc. of 25th Annual Southern Conservation Tillage Conference for Sustainable Agriculture. Auburn. https://www.researchgate.net/publication/242306530

Thornton, P. K. (2010). Livestock production: recent trends, future prospects. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *365*(1554), 2853-2867. https://doi.org/10.1098/rstb.2010.0134

United Nations. (2022). *World Population Prospects 2022 - Summary of Results*. https://www.un.org/development/desa/pd/sites/www.un.org.development.desa.pd/files/wpp2022_summary_of_results.pdf

United States Department of Agriculture-National Resources Conservation Services. (1996). NSSC-SSL Report Uruguay. Soil characterization Data. En *Primary Characterization Data (Uruguay)* (USDA-NRCS).

Ward, S. M., Holden, N. M., White, E. P. y Oldfield, T. L. (2016). *The «circular economy» applied to the agriculture (livestock production) sector-discussion paper*. http://www.agrocycle.eu

Wilkinson, J. M. y Lee, M. R. F. (2018). Review: Use of human-edible animal feeds by ruminant livestock. *Animal*, *12*(8), 1735-1743. https://doi.org/10.1017/S175173111700218X

Xue, H., Mainville, D., You, W. y Nayga, R. M. (2010). Consumer preferences and willingness to pay for grass-fed beef: Empirical evidence from in-store experiments. *Food Quality and Preference*, *21*(7), 857-866. https://doi.org/10.1016/j.foodqual.2010.05.004

8. Anexos

8.1. Anexo 1. Taking the steps toward sustainable livestock: our multidisciplinary global farm platform journey

M. Jordana Rivero,† Alex C. O. Evans,‡ Alexandre Berndt,|| Andrew Cartmill,$ Andrew Dowsey,¶ Anne Farruggia,** Catherine Mignolet,†† Daniel Enriquez-Hidalgo,†,¶ Dave Chadwick,‡‡ Davy I. McCracken,|||| Dennis Busch,$ Fabiana Pereyra,$$ Graeme B. Martin,¶¶ Gregg R. Sanford,*** Helen Sheridan,‡ Iain Wright,††† Laurent Brunet,†† Mark C. Eisler,¶ Nicolas Lopez-Villalobos,‡‡‡ Pablo Rovira,$$ Paul Harris,† Paul Murphy,‡ A. Prysor Williams,‡‡ Randall D. Jackson,*** Rui Machado,|| Suraj P.T.,|||||| Thomas Puech,†† Tommy M. Boland,‡ Walter Ayala,$$ and Michael R.F. Lee$$$

† Sustainable Agriculture Sciences, Rothamsted Research, North Wyke, Okehampton, Devon EX20 2SB, UK

‡ School of Agriculture & Food Science, University College Dublin, Belfield, Dublin 4, Ireland

|| Embrapa Southeast Livestock, São Carlos, São Paulo 13560-970, Brazil

$ School of Agriculture, University of Wisconsin–Platteville, Platteville, WI 53818, USA

¶ Bristol Veterinary School, University of Bristol, Langford, Somerset BS40 5DU, UK

** INRAE—ACT UE 0057 DSLP, 17450 Saint Laurent de la Prée, France

†† INRAE—ACT, UR 0055 ASTER, 88500 Mirecourt, France

‡‡ School of Natural Sciences, Bangor University, Bangor LL57 2UW, UK

|||| Hill & Mountain Research Centre, SRUC: Scotland's Rural College, Kirkton Farm, Crianlarich FK20 8RU, UK

$$ Instituto Nacional de Investigación Agropecuaria (INIA), Treinta y Tres 33000, Uruguay

¶¶ UWA Institute of Agriculture, The University of Western Australia, Crawley 6009, Australia

*** Department of Agronomy, University of Wisconsin–Madison, Madison, WI 53706, USA

††† International Livestock Research Institute (ILRI), Nairobi, Kenya

‡‡‡ School of Agriculture and Environment, Massey University, Palmerston North 4410, New Zealand

|||||| Livestock Research Station Thiruvazamkunnu, Kerala Veterinary and Animal Sciences University, Kerala-678601, India

$$$ Harper Adams University, Newport, Shropshire TF10 8NB, UK

Perspectives

Taking the steps toward sustainable livestock: our multidisciplinary global farm platform journey

M. Jordana Rivero,[†] Alex C. O. Evans,[‡] Alexandre Berndt,[||] Andrew Cartmill,[$] Andrew Dowsey,[¶] Anne Farruggia,[**] Catherine Mignolet,[††] Daniel Enriquez-Hidalgo,[†,¶] Dave Chadwick,[‡‡] Davy I. McCracken,[||||] Dennis Busch,[$] Fabiana Pereyra,[$$] Graeme B. Martin,[¶¶] Gregg R. Sanford,[***] Helen Sheridan,[‡] Iain Wright,[†††] Laurent Brunet,[††] Mark C. Eisler,[¶] Nicolas Lopez-Villalobos,[‡‡‡] Pablo Rovira,[$$] Paul Harris,[†] Paul Murphy,[‡] A. Prysor Williams,[‡‡] Randall D. Jackson,[***] Rui Machado,[||] Suraj P.T.,[||||||] Thomas Puech,[††] Tommy M. Boland,[‡] Walter Ayala,[$$] and Michael R.F. Lee[$$$]

[†]Sustainable Agriculture Sciences, Rothamsted Research, North Wyke, Okehampton, Devon EX20 2SB, UK
[‡]School of Agriculture & Food Science, University College Dublin, Belfield, Dublin 4, Ireland
[||]Embrapa Southeast Livestock, São Carlos, São Paulo 13560-970, Brazil
[$]School of Agriculture, University of Wisconsin–Platteville, Platteville, WI 53818, USA
[¶]Bristol Veterinary School, University of Bristol, Langford, Somerset BS40 5DU, UK
[**]INRAE—ACT UE 0057 DSLP, 17450 Saint Laurent de la Prée, France
[††]INRAE—ACT, UR 0055 ASTER, 88500 Mirecourt, France
[‡‡]School of Natural Sciences, Bangor University, Bangor LL57 2UW, UK
[||||]Hill & Mountain Research Centre, SRUC: Scotland's Rural College, Kirkton Farm, Crianlarich FK20 8RU, UK
[$$]Instituto Nacional de Investigación Agropecuaria (INIA), Treinta y Tres 33000, Uruguay
[¶¶]UWA Institute of Agriculture, The University of Western Australia, Crawley 6009, Australia
[***]Department of Agronomy, University of Wisconsin–Madison, Madison, WI 53706, USA
[†††]International Livestock Research Institute (ILRI), Nairobi, Kenya
[‡‡‡]School of Agriculture and Environment, Massey University, Palmerston North 4410, New Zealand
[||||||]Livestock Research Station Thiruvazamkunnu, Kerala Veterinary and Animal Sciences University, Kerala-678601, India
[$$$]Harper Adams University, Newport, Shropshire TF10 8NB, UK

Implications

- The Global Farm Platform was conceived and established to explore multidisciplinary strategies for optimising the sustainability of ruminant livestock systems around the world.
- International sustainability issues are common, but the solutions are often region-specific; therefore, our farms, situated across all major agroclimatic zones, are a unique resource worldwide.
- Each farm is following 'steps to sustainable livestock' to improve their production system(s), thereby developing robust metrics to progress economic, environmental and social viability.
- The consortium works collaboratively to improve the sustainability of ruminants, which we argue are a vital component of global food systems, delivering both human and planetary health.

https://doi.org/10.1093/af/vfab048

Key words: circularity, grazing systems, mixed farming, precision farming, research farms, ruminant livestock

Ruminant livestock are a vital global source of high-quality protein and bioavailable minerals and vitamins. They support healthy dietary choices by providing milk and meat produced from less productive land and food industry byproducts. However, despite the contribution of ruminants to food systems and the circular bioeconomy, ruminant production systems are increasingly questioned due to their environmental impact, particularly their significant contribution to greenhouse gas (GHG) emissions and associated global warming. There is a need, therefore, to identify a pathway to sustainable global ruminant production. In 2014, our group defined eight strategies or "steps" (Eisler et al., 2014), to mitigate the environmental impacts of ruminant production while optimizing the quantity and quality of the food they produce. To realize these goals, we established the "Global Farm Platform" initiative (www.globalfarmplatform.org), a network of "farm platforms" or research farms (RFs), to explore multidisciplinary strategies and evaluate different production systems around the globe (Table 1). Here, we provide a perspective on our approach and the steps we are taking to realize the ambition of supporting sustainable ruminant livestock production as a part of future food systems contributing to both human and planetary health.

Feed Animals Less Human Food (Step 1)

Most of our RFs are investigating ways to enhance the sustainability of forage-based systems, with no use, or only strategic use, of supplementary feeds for certain short periods of the production cycle (Rivero et al., 2021). INRAE-SLP decreased the percentage of arable lands dedicated to production of supplemental feed for animals from 48% in 2017 to 28% in 2020. All ruminants in the INRAE-AM system are fed exclusively on grass, while the annual crops are intended exclusively for human consumption. SVT has introduced a new cultivation strategy, the Kenyan "Tumbukiza" method, which uses cultivars of Hybrid Napier (*Pennisetum purpureum* L. × *Pennisetum glaucam* L.) planted in holes to improve soil fertility and moisture levels, thus increasing fodder biomass production for their cut and carry system. UCD-LTGP and HRC are testing grazing systems based on swards with increasing levels of plant diversity (perennial ryegrass monoculture, perennial ryegrass and white clover mixed sward, and a 6-species grass, legume, forage herb mixed sward) to enhance resilience to extreme weather events and deliver greater yield with reduced inputs.

Raise Regionally Appropriate Animals (Step 2)

We have identified the need for selecting animals adapted to local conditions that are able to cope with climate change challenges (Rivero et al., 2021). INRAE-AM is adapting its animals to low-input grazing systems (e.g., enhanced rusticity and reduction of cow size) via selection and crossbreeding. INRAE-SLP bases its research on a dual-purpose local rustic beef breed native to wetlands (Maraîchine), while SRUC-KA is crossbreeding Aberdeen Angus with Beef Shorthorn cattle in order to improve their ability to cope with extreme mountainside environments. Similarly, SVT is working with native breeds of cattle (Vechur), buffalo (Murrah), and goats (Malabari and Attapadi) plus *indicus* × *taurus* crossbred cattle, with the former having been shown to exhibit greater tolerance to heat stress (Elayadeth-Meethal et al., 2018).

Keep Animals Healthy (Step 3)

Most of our RFs are working in this area with different approaches. For instance, the use of sensors and additional technology allows SRUC-KA to monitor animal health and welfare in mountainous conditions, while JOC is using 64 video cameras to track cattle movements and social behavior for early disease detection and to assess infectious disease transmission and minimize antimicrobial resistance. KRS demonstrated that a vaccine against wildebeest-associated malignant catarrhal fever is highly effective against the disease in cattle with a vaccine efficacy of 80% (Cook et al., 2019). Through work at SVT, welfare challenges in subsistence dairy farms in India have been identified (Mullan et al., 2020). UCD-LTGP is showing that greater diversity in forage plants decreases animal parasite burdens.

Adopt Smart Supplements (Step 4)

In some of our RFs, spontaneous vegetation is being explored as feed, bedding (e.g., reed in INRAE-SLP; Durant et al., 2020), or smart supplements (e.g., *Azolla* spp.—a small aquatic fern that flows on the water surface and is nutritionally rich—in SVT and INRAE-SLP). ESL has developed the Guandu BRS Mandarim (*Cajanus cajan* cv. BRS Mandarim), an N-fixating legume suitable to enrich soil quality of degraded pasturelands while its aerial part serves as a protein supplement to cattle, particularly in the dry season (Figure 1). HAUF and UWA-FF are also testing dietary supplements or feed ingredients which act as methane suppressants at a farm system scale.

Eat Quality Not Quantity (Step 5)

Even though this step is mainly oriented toward the consumer, many of our RFs are working on improving the quality of the final food products. In addition to increasing system productivity, SRUC-KA is focusing on carcass conformation through the use of CT scanning (Lambe et al., 2017), the NWFP is investigating the nutritional value and the associated carbon footprint of forage-based beef systems (Lee et al., 2021), and UCD-LTGP has ongoing research on meat quality from multispecies forage leys.

Tailor Practices to Local Culture (Step 6)

Most of the researches undertaken by our RFs are agreed with and/or transferred to stakeholders, particularly the farming community. WICST seeks to transform agriculture of the North Central United States to perennial grassland dominance to restore the function of the original

Table 1. Global Farm Platform steps toward sustainable livestock systems in our network of 16 RFs

	Steps to sustainable livestock (see Eisler et al., 2014)							
Research farm (see Rivero et al., 2021)	1	2	3	4	5	6	7	Best Practice (8) and main aim
Dairy 1 (Palmerston North, New Zealand)	●	●	●	●	●	●	●	Temperate grazing dairy system—improve sustainability, farmer and animal welfare, and profitability through once a day milking and selection of dairy cows for feed conversion efficiency.
ESL - Embrapa Southeast Livestock (Sao Carlos, Brazil)	●	●	●	●	●	●	●	Subtropical sustainable beef and milk systems—explore net-zero C potential.
HAUF - Harper Adams University Farm (England, UK)	●		●			●	●	Conversion to circular farming—show how mixed-farming can deliver to net-zero C.
HRC - Henfaes Research Centre (Wales, UK)	●						●	Temperate uplands sheep—improve productivity with least environmental impact.
INIA-PAP - INIA Palo a Pique (Treinta y Tres, Uruguay)	●		●	●	●	●	●	No-till crop-livestock (beef) rotations—evaluate four ways of producing 400 kg LW/ha per yrs.
INRAE-AM - INRAE ASTER-Mirecourt (Mirecourt, France)	●	●	●				●	Organic crop-livestock (dairy) system—implement an agroecological transition.
INRAE-SLP - INRAE Saint-Laurent-de-la-Prée (La Rochelle, France)	●	●	●	●	●		●	Organic crop-livestock (beef) system in marshes—restore biodiversity, mitigate GHG, produce animal and vegetal human food for short circuit, enable adoption by farmers.
JOC - The John Oldacre Centre for Sustainability and Welfare in Dairy Production (England, UK)			●				●	Precision farming system for housed dairy cattle—monitoring animal health, behaviour and welfare, nutrition and GHG emissions.
KRS - Kapiti Research Station and Wildlife Conservancy (Nairobi, Kenya)	●	●	●				●	Semi-arid rangeland (livestock-wildlife)—improve livestock production sustainably, explore the ecological dynamics of savannahs and their interactions with humans, livestock and wildlife
NWFP - North Wyke Farm Platform (England, UK)	●	●	●	●	●		●	Temperate lowland sheep and beef systems—assess sustainability of production systems in its three dimensions.
UWP-PF - University of Wisconsin-Platteville Pioneer Farm (Wisconsin, US)	●			●		●	●	Dairy (housed or hybrid grazed systems)—investigate the effects of alternative dairy production systems on water quality and nutrient cycling.
SRUC-KA - SRUC Kirkton and Auchtertyre (Scotland, UK)	●	●	●		●	●	●	Temperate uplands livestock (sheep and beef)—understanding what may be practical or economically viable for upland land managers to implement to improve sustainability.
SVT - Silent Valley Thiruvazhamkunnu Livestock Research Station (Kerala, India)	●	●	●	●			●	Cut and carry livestock systems—assess sustainability of different fodder management strategies.
UCD-LTGP - UCD Lyons Farm Long Term Grazing Platform (Dublin, Ireland)	●		●		●		●	Temperate lowland dairy x beef systems—investigate the interrelationships between pasture type, animal production, the environment, product quality and farm economics.
UWA-FF - University of Western Australia Future Farm 2050 (Pingelly, Australia)	●	●	●		●		●	Drylands sheep system—define and implement the 'ideal' farm: profitable, 'clean, green and ethical', commitment to conservation of biodiversity; it must take into account the people's needs.
WICST - The Wisconsin Integrate Cropping Systems Trial (Wisconsin, US)	●		●		●	●	●	Midwestern cropping systems - evaluate productivity, profitability, and ecological performance.

●Step being addressed by the research farm (RF) as part of their research activities.

Figure 1. Canchim breed heifers in an *Urocloa brizantha* pasture enriched with *Cajanus cajan* cv. BRS Mandarim legume (Photo: Gisel Rosso).

prairie—water purification, flood mitigation, climate stabilization, and biodiversity—while revitalizing rural communities decimated by farm consolidation. INIA-PAP is testing four crop-livestock (beef) rotations, representative of the predominant commercial livestock strategies in Uruguay, with the aim of evaluating four ways of producing 400 kg LW/ha per year that is economically, environmentally, and operationally viable (Rovira et al., 2020). UWP-PF is investigating the effects of alternative dairy production systems on water quality and nutrient cycling. Dairy 1 is evaluating breeds and crossbreeding for once-a-day milking (Jiang et al., 2020) and the use of precision technology to feed cows more efficiently (Duranovich et al., 2021). HAUF is mapping the impact (economic, environmental, and social indicators) of conversion from separate crop and livestock enterprises to a mixed circular crop-livestock farming system. HRC has identified the cultural, practical, and economic barriers to better soil and nutrient management in ruminant systems (Gibbons et al., 2014; Rhymes et al., 2021).

Track Costs and Benefits (Step 7)

All our RFs are delivering to this step with various approaches. HRC found that urine patches deposited on hill and upland soils generate very small quantities of nitrous oxide, with implications for carbon footprinting (Marsden et al., 2018). UCD-LTGP is investigating the impact on above- and below-ground biodiversity, water quality, meat quality, economic, and other non-market benefits of sustainable grazing systems. ESL has demonstrated that crop-livestock and crop-livestock-forest integrated systems deliver less nitrous oxide into the atmosphere as compared with conventional crop practices (Sato et al., 2019). The NWFP is applying Life Cycle Assessment (LCA) approaches to compare its production systems (McAuliffe et al., 2018), while INIA-PAP is collating a database to apply LCA to its four crop-livestock systems.

Study Best Practice (Step 8)

Our vision is to identify better practices to optimize the use of livestock in various regions, using local resources, breeds, and feedstuffs—and produce tangible evidence of sustainability. The "Global Farm Platform" initiative started with three operational RFs in three continents in 2014 and has subsequently grown to 16 RFs in five continents covering a wide variety of social and agroclimatic conditions and production systems (Table 1). There are plans to continue establishing further platforms to test other relevant ruminant production systems, for example, two Chinese RFs and another Australian RF are in the process of joining.

Final Remarks

Our network of RFs traverses a wide variety of social and agroclimatic conditions and production systems, and also brings together researchers with expertise in most of the areas relevant to the multidisciplinary approach required to address the global issues contributing to sustainable animal production, such as animal health, welfare, nutrition and genetics, pasture management, agroecology, biodiversity, agroforestry, silvopastoralism, meat quality and safety, GHG emissions, hydrology, soil carbon, biogeochemistry, LCA, economics, knowledge exchange and extension, precision farming and sensors, informatics, statistics, modeling, and artificial intelligence.

Since our first paper on the steps to sustainable livestock was published (Eisler et al., 2014), there has been a major increase in recognition that livestock managers play a vital role in managing land, from the perspectives of carbon sequestration and biodiversity, among other benefits, such as wildfire control (FAO, 2020). Furthermore, the role of farmed livestock in the circular bioeconomy has been recognized (Van Zanten et al., 2019), as has the potential for Precision Livestock Farming, further strengthening the commitment of our RF network to the exploration of solutions needed for the next steps toward sustainable livestock. Despite their variation, our farms face the same challenges–reducing environmental impact, improving animal performance, and maintaining health and welfare–yet, the solutions to these challenges must be regional and applied under local conditions, verifying the value of our network across contrasting agroclimatic zones as a global resource.

Single metrics of sustainability, such as methane intensity/carbon footprint, seem to favor intensive solutions for ruminant production. However, in such solutions, there are tradeoffs in relation to, for example, the food/feed competition and the ability of the animals to express their natural behavior. Our team has acknowledged these tradeoffs as critical issues in choosing the major steps to sustainable livestock production, and we decided to favor forage-based solutions. Forage-based systems are inevitably complicated by the largely uncontrolled environment within which the animals and the forage plants need to survive and thrive. An obvious major limitation is the seasonal nature of rainfall and temperature, but successful responses of these challenges can be found by making visionary choices for both animal genotype and forage species. For example, by moving away from "traditional" forages, we have found species that offer nutritional advantages, drought resistance, shelter for neonates, and plant secondary compounds that combat helminths and methane emissions. Few if any of these alternative forages have been subjected to genetic selection, so there is an opportunity for improvement. Finally, increasing forage diversity, and thus offering dietary diversity, improves animal productivity and health.

Acknowledgments

This work was funded by several sources varying with the RF. ESL: PECUS project (Grant number 02.12.02.008.00.02). HAUF: School of Sustainable Food and Farming. HRC: Department for Environment, Food & Rural Affairs (DEFRA) through the Sustainable Intensification Platform (SIP), Integrated Farm Management (LM0201). INRAE-SLP: National Research Institute for Agriculture, Food and the Environment (INRAE) internal fund and Nouvelle-Aquitaine region grant. NWFP: Soil to Nutrition Institute Strategic Programme (grant number BBS/E/C/000I0320) and The North Wyke Farm Platform National Capability (grant number BBS/E/C/ 000J0100) funded by the UK Biotechnology and Biological Sciences Research Council (BBSRC). SRUC: The Global Food Security's 'Resilience of the UK Food System Program', supported by the BBSRC, the Economic and Social Research Council, the Natural Environment Research Council, and the Scottish Government. UCD-LTGP: Department of Agriculture, Food and the Marine, Ireland's Competitive Research Funding Programs. UWP-PF: United States Department of Agriculture (USDA) Agricultural Research Service Long-Term Agroecosystem Research Network and the USDA National Institute of Food and Agriculture (NIFA) through its Capacity Building Grant for Non-Land Grant Colleges of Agriculture.

Conflict of interest statement. The authors declare that they have no conflict of interest.

About the Authors

"Global Farm Platform" Initiative

The Global Farm Platform initiative (www.globalfarmplatform.org) is a network of research farms and institute members working collaboratively to enhance the sustainability of ruminant livestock systems through the development of transformational regional solutions to global challenges and promote their adoption. This multidisciplinary international network will provide a unique combination of research and practice for diverse ruminant production systems in a wide range of cultural, socioeconomic, and climatic zones.

Research farms:

- **Dairy 1**, Massey University, Palmerston North, New Zealand
- Embrapa Southeast Livestock, Embrapa, Sao Carlos, Brazil (ESL)
- Harper Adams University Farm, Harper Adams University, England, UK (HAUF)
- Henfaes Research Centre, Bangor University, Wales, UK (HRC)
- INIA Palo a Pique, INIA, Treinta y Tres, Uruguay (INIA-PAP)
- INRAE ASTER-Mirecourt, INRAE, Mirecourt, France (INRAE-AM)
- INRAE Saint-Laurent-de-la-Prée, INRAE, La Rochelle, France (INRAE-SLP)
- The John Oldacre Centre for Sustainability and Welfare in Dairy Production, University of Bristol, England, UK (JOC)
- Kapiti Research Station and Wildlife Conservancy, ILRI, Nairobi, Kenya (KRS)
- The North Wyke Farm Platform, Rothamsted Research, England, UK (NWFP)
- University of Wisconsin-Platteville Pioneer Farm, Wisconsin, US (UWP-PF)
- SRUC Kirkton & Auchtertyre, Scotland, UK (SRUC-KA)
- Silent Valley Thiruvazhamkunnu Livestock Research Station, KVASU, Kerala, India (SV)
- UCD Lyons Farm Long-Term Grazing Platform, Dublin, Ireland (UCD-LGP)
- University of Western Australia Future Farm 2050, Pingelly, Australia (UWA-FF2050)
- The Wisconsin Integrate Cropping Systems Trial, University of Wisconsin-Madison, Wisconsin, US (WICST)

In addition to the institutions hosting the research farms, there are other institute members that contribute to the network:

- Key Laboratory of Plant Resources, Chinese Academy of Science, China (CAS)
- International Centre for Tropical Agriculture, Kenya (CIAT-Kenya)
- China Agricultural University, China (CAU)
- Federal University of Agriculture, Abeokuta, Nigeria (FUNAAB)
- Department of Agronomy, Kansas State University, US (KSU)
- Small Scale Livestock and Livelihoods Program, Malawi (SSLLP)
- Soil Science Department, College of Agriculture and Natural Sciences, University of Cape Coast, Ghana (UCC)
- Faculty of Agricultural, Life and Environmental Sciences, University of Alberta, Canada (UoA)
- Institute of Dairy Science, Zhejiang University, China (ZU)

World map with the location of the Research Farms and Member Institutes that are part of the Global Farm Platform initiative

Literature Cited

Cook, E., G. Russell, D. Grant, C. Mutisya, L. Omoto, E. Dobson, F. Lankester, and V. Nene. 2019. A randomised vaccine field trial in Kenya demonstrates protection against wildebeest-associated malignant catarrhal fever in cattle. Vaccine. 37:5946–5953. doi:10.1016/j.vaccine.2019.08.040

Duranovich, F., N. López-Villalobos, N. Shadbolt, I. Draganova, I. Yule, and S. Morris. 2021. The deviation between dairy cow metabolizable energy requirements and pasture supply on a dairy farm using proximal hyperspectral sensing. Agric. 11:1–15. doi:10.3390/agriculture11030240

Durant, D., A. Farruggia, and A. Tricheur. 2020. Utilization of common reed (*Phragmites australis*) as bedding for housed suckler cows: practical and economic aspects for farmers. Resources. 9:1–13. doi:10.3390/resources9120140

Eisler, M.C., M.R.F. Lee, J.F. Tarlton, G.B. Martin, J. Beddington, J.A.J. Dungait, H. Greathead, J. Liu, S. Mathew, H. Miller, et al. 2014. Steps to sustainable livestock. Nature. 507:32–34. doi:10.1038/507032a

Elayadeth-Meethal, M., A. Thazhathu Veettil, S.K. Maloney, N. Hawkins, T.H. Misselbrook, V. Sejian, M.J. Rivero, and M.R.F. Lee. 2018. Size does matter: parallel evolution of adaptive thermal tolerance and body size facilitates adaptation to climate change in domestic cattle. Ecol. Evol. 8:10608–10620. doi:10.1002/ece3.4550

FAO. 2020. Biodiversity and the livestock sector – Guidelines for quantitative assessment – Version 1. Rome, Livestock Environmental Assessment and Performance Partnership (FAO LEAP). https://doi.org/10.4060/ca9295en

Gibbons, J.M., J.C. Williamson, A.P. Williams, P.J.A. Withers, N. Hockley, I.M. Harris, J.W. Hughes, R.L. Taylor, D.L. Jones, and J.R. Healey. 2014. Sustainable nutrient management at field, farm and regional level: soil testing, nutrient budgets and the trade-off between lime application and greenhouse gas emissions. Agric. Ecosyst. Environ. 188:48–56. doi:10.1016/j.agee.2014.02.016

Jiang, H., R.E. Hickson, O.T. Woods, M. Morandeau, J.L. Burke, M. Correa-Luna, D.J. Donaghy, and N. Lopez-Villalobos. 2020. Persistency and lactation curves modelled using nonlinear random regression in dairy. New Zeal. J. Anim. Sci. Prod. 80:131–136.

Lambe, N.R., K.A. McLean, J. Gordon, D. Evans, N. Clelland, and L. Bunger. 2017. Incorporating computed tomography based predictors of meat quality into a breeding program—breeding lambs for more taste less waste. In: D. Troy, C. McDonnell, L. Hinds and J. Kerry, editors. 63rd International Congress of Meat Science and Technology 3–4; 13-18 August 2017; Cork, Ireland. Wageningen Academic Publishers.

Lee, M.R.F., G.A. McAuliffe, J.K.S. Tweed, B.A. Griffith, S.A. Morgan, M.J. Rivero, P. Harris, T. Takahashi, and L. Cardenas. 2021. Nutritional value of suckler beef from temperate pasture systems. Animal. 15:100257. doi:10.1016/j.animal.2021.100257

Marsden, K.A., J.A. Holmberg, D.L. Jones, and D.R. Chadwick. 2018. Sheep urine patch N_2O emissions are lower from extensively-managed than intensively-managed grasslands. Agric. Ecosyst. Environ. 265:264–274. doi:10.1016/j.agee.2018.06.025

McAuliffe, G.A., T. Takahashi, R.J. Orr, P. Harris, and M.R.F. Lee. 2018. Distributions of emissions intensity for individual beef cattle reared on pasture-based production systems. J. Clean. Prod. 171:1672–1680. doi:10.1016/j.jclepro.2017.10.113

Mullan, S., S.J. Bunglavan, E. Rowe, D.C. Barrett, M.R.F. Lee, D. Ananth, and J. Tarlton. 2020. Welfare challenges of dairy cows in India identified through on-farm observations. Animals. 10:1–13. doi:10.3390/ani10040586

Rhymes, J.M., S. Wynne-Jones, A.P. Williams, I.M. Harris, D. Rose, D.R. Chadwick, and D.L. Jones. 2021. Identifying barriers to routine soil testing within beef and sheep farming systems. Geoderma. 404:115298. doi:10.1016/j.geoderma.2021.115298

Rivero, M.J., N. Lopez-Villalobos, A. Evans, A. Berndt, A. Cartmill, A. L. Neal, A. McLaren, A. Farruggia, C. Mignolet, D. Chadwick, et al.

2021. Key traits for ruminant livestock across diverse production systems in the context of climate change: perspectives from a global platform of research farms. Reprod. Fertil. Dev. 33:1–19. doi:10.1071/RD20205

Rovira, P., W. Ayala, J. Terra, F. García-Préchac, P. Harris, M.R.F. Lee, and M.J. Rivero. 2020. The 'Palo a Pique' long-term research platform: first 25 years of a crop–livestock experiment in Uruguay. Agronomy. 10(3):441. doi:10.3390/agronomy10030441

Sato, J.H., C.C. de Figueiredo, R.L. Marchão, A.D. de Oliveira, L. Vilela, F.M. Delvico, B.J.R. Alves, and A.M. de Carvalho. 2019. Understanding the relations between soil organic matter fractions and N_2O emissions in a long-term integrated crop–livestock system. Eur. J. Soil Sci. 70:1183–1196. doi:10.1111/ejss.12819

Van Zanten, H.H.E., M.K. Van Ittersum, and I.J.M. De Boer. 2019. The role of farm animals in a circular food system. Glob. Food Sec. 21:18–22. doi:10.1016/j.gfs.2019.06.003

8.2. Anexo 2. Feasibility of mitigation measures for agricultural greenhouse gas emissions in the UK. A systematic review

Asma Jebari[1] · Fabiana Pereyra Goday[2] · Atul Kumar[1] · Adrian L. Collins[1] · M. Jordana Rivero[1] · Graham A. McAuliffe[1]

[1] Net Zero and Resilient Farming, Rothamsted Research,

North Wyke, Okehampton EX20 2SB, Devon, UK

[2]Instituto Nacional de Investigación Agropecuaria (INIA),

Ruta 8 km 281, Treinta y Tres, postcode 33000 Montevideo,

Uruguay

Agronomy for Sustainable Development (2024) 44:2
https://doi.org/10.1007/s13593-023-00938-0

REVIEW ARTICLE

Feasibility of mitigation measures for agricultural greenhouse gas emissions in the UK. A systematic review

Asma Jebari[1] · Fabiana Pereyra-Goday[2] · Atul Kumar[1] · Adrian L. Collins[1] · M. Jordana Rivero[1] · Graham A. McAuliffe[1]

Accepted: 24 November 2023 / Published online: 28 December 2023

Abstract

The UK Government has set an ambitious target of achieving a national "net-zero" greenhouse gas economy by 2050. Agriculture is arguably placed at the heart of achieving net zero, as it plays a unique role as both a producer of GHG emissions and a sector that has the capacity via land use to capture carbon (C) when managed appropriately, thus reducing the concentration of carbon dioxide (CO_2) in the atmosphere. Agriculture's importance, particularly in a UK-specific perspective, which is also applicable to many other temperate climate nations globally, is that the majority of land use nationwide is allocated to farming. Here, we present a systematic review based on peer-reviewed literature and relevant "grey" reports to address the question "how can the agricultural sector in the UK reduce, or offset, its direct agricultural emissions at the farm level?" We considered the implications of mitigation measures in terms of food security and import reliance, energy, environmental degradation, and value for money. We identified 52 relevant studies covering major foods produced and consumed in the UK. Our findings indicate that many mitigation measures can indeed contribute to net zero through GHG emissions reduction, offsetting, and bioenergy production, pending their uptake by farmers. While the environmental impacts of mitigation measures were covered well within the reviewed literature, corresponding implications regarding energy, food security, and farmer attitudes towards adoption received scant attention. We also provide an open-access, informative, and comprehensive dataset for agri-environment stakeholders and policymakers to identify the most promising mitigation measures. This research is of critical value to researchers, land managers, and policymakers as an interim guideline resource while more quantitative evidence becomes available through the ongoing lab-, field-, and farm-scale trials which will improve the reliability of agricultural sustainability modelling in the future.

Keywords Net zero · Carbon footprint · Farming interventions · Arable farming · Livestock systems · Mixed farming

Contents

✉ Asma Jebari
asma.jebari@rothamsted.ac.uk

[1] Net Zero and Resilient Farming, Rothamsted Research, North Wyke, Okehampton EX20 2SB, Devon, UK

[2] Instituto Nacional de Investigacion Agropecuaria (INIA), Ruta 8 km 281, Treinta y Tres, postcode 33000 Montevideo, Uruguay

1 Introduction

While agriculture contributes less than 1% to the United Kingdom's (UK's) economy, it provides around three-quarters of domestic food consumption and utilizes around 71% of the land. Approximately 72% of the latter is used for grazing systems and 26% for arable crops including cereals, oilseeds, and potatoes, with the remaining land (~2%) being utilized for produce such as medicinal plants and herbs (Defra 2021). As a food-trading nation, the UK relies on both imports and a thriving domestic agricultural sector to feed itself and drive economic growth (ADAS 2019). In the most recent national inventory assessment of UK emissions, agriculture accounted for ~10% of total greenhouse gas (GHG) emissions (Brown et al. 2020). Despite the relatively low total emissions arising from primary food production compared to other sectors, such as energy and transport (BEIS 2022), the agricultural sector is the major source of both nitrous oxide (N_2O) and methane (CH_4) emissions in the UK, both of which are powerful and complex GHGs, accounting for nearly 69% of total N_2O emissions and 48% of total CH_4 emissions in the UK, respectively (Defra 2021). In contrast, agriculture only accounts for ~1.7% of total carbon dioxide (CO_2) emissions (Defra 2021). More specifically, nearly 90% of agricultural N_2O emissions originate from soils through microbial (de)nitrification of nitrogen-based fertilizers, farmyard manure (FYM), and deposition of urine and feces on grazing/foraging lands and indirectly through leaching/runoff and volatilization primarily from ammonia (NH_3). Most CH_4 emissions (~90%) arise from enteric fermentation (digestive processes, specifically eructation) in ruminant animals, with manure management practices accounting for the remainder.

The agricultural sector accounted for 88% of the UK's NH_3 emissions in 2021 (Defra 2021). NH_3 is generated from the application of synthetic (e.g., ammonium nitrate) and organic fertilizers (e.g., slurry and manure) to soils and during storage. Further, while rates of soil erosion in England are not excessively high by global standards, rates on agricultural land are elevated relative to those under natural land covers, resulting in elevated sediment delivery to rivers (Collins and Zhang 2016; Collins et al. 2021) leading to off-farm impacts including degradation of aquatic ecology (e.g., Kemp et al. 2011) and the siltation of drinking water reservoirs (Foster et al. 2011).

The Committee on Climate Change (CCC) has recommended a 64% reduction in GHG emissions from the agriculture and land use sector to meet the national 2050 net-zero GHG target in the UK (CCC 2020). The fact that this is not a 100% reduction reflects the natural biological baseline emissions associated with primary food production (e.g., even if the land was "rewilded," there would still be baseline emissions arising from unproductive land, due to microbial activity during natural decomposition cycles) (CIEL 2020). In line with the CCC, the National Farmers Union (NFU) of England and Wales established an ambitious goal of net zero by 2040, while assuring climate-friendly food production with high standards of food safety, animal welfare, and environmental stewardship. For instance, agriculture will need to reduce emissions from its production and increase its potential to sequester soil organic carbon (SOC) through land occupation optimization, with GHG offsetting strategies (Fig. 1) such as afforestation and silvopastoral systems being prime exemplars of mitigation pathways (Eory et al. 2020).

Mitigation measures for delivering the UK Government's net-zero target by 2050 must consider both the economic (e.g., food production and reliance on imports) and environmental sustainability of production systems going forward (CIEL 2022). Furthermore, the NFU highlighted the fact that the transition of agriculture to net-zero GHGs must ensure the economic, environmental, and social benefits of farming, such as supporting rural workforces and delivery of nutritious produce, are protected (NFU 2021a). Environmental scientists and engineers, social scientists, nutritional scientists, and economists are therefore tasked to seek ways to increase productivity while at the same time reducing

Fig. 1 Example of offsetting mitigation measure: planting hedgerows into sheep-grazed pasture in southwest England. Apart from GHG mitigation potential, through SOC sequestration, establishing hedgerows provides a range of co-benefits to livestock and the landscape. Trees can boost production, improve animal health and welfare, and provide wider environmental benefits (see Section 3.7).

environmental damage and maintaining the healthy function of agroecosystems (e.g., increasing biodiversity, often measured as species losses-gains per year, while simultaneously reducing GHGs) in the long term (Tilman et al. 2011; Tilman and Clark 2014).

An important part of working towards net zero includes the accurate accounting of GHG emissions. The national inventory accounting forms the basis of international climate change treaties (e.g., the Kyoto Protocol). Another, more holistic, approach to quantifying supply-chain-level environmental impacts is life cycle assessment (LCA), a deterministic modelling framework widely used in agricultural sustainability analyses. In contrast to national GHG inventories, detailed LCAs quantify losses of pollutants occurring in other countries for imported products associated with food production, such as animal feed (e.g., displaced protein-sources imported from the Americas), and fertilizer chemicals (CIEL 2020). As such, LCA provides a deeper, global view of the C footprint for any product or service (Müller et al. 2020). However, despite recent computational and mathematical improvements to LCA, data availability remains one of the major limiting factors when utilizing the framework to answer pressing societal concerns pertaining to environmental degradation (McAuliffe 2020a). In the absence of suitable life cycle inventory analyses (LCI) material flows, assessing the potential of GHG abatement efforts is challenging. For instance, predictions made by scenario-based LCA models in the context of net zero are currently liable to high degrees of uncertainty, despite numerous methodological capabilities to capture such data-based restrictions (ISO 2006; Cain et al. 2019; Müller et al. 2020; McAuliffe et al. 2020b).

Systematic reviews provide a rigorous, objective, and transparent means of creating a searchable database of relevant academic and grey literature (Kohl et al. 2018), while providing an opportunity to clarify the current evidence base and highlight important knowledge gaps. To the best of our knowledge, the most recent review on climate change mitigation in the UK was a literature review that focused only on cropping systems (i.e., food-crop production, particularly arable systems including root crops; Rial-Lovera et al. 2017). Other reviews related to broader sustainability assessments (e.g., exploration of environmental impacts including water pollution and terrestrial acidification, both of which indirectly produce GHGs and thus affect the achievement of net zero) have covered livestock in general (de Vries and de Boer 2010), beef production (de Vries et al. 2015), pig production (McAuliffe et al. 2016), the nutrition-environment nexus (McAuliffe et al. 2020a), and technical issues related to complexities such as how to allocate burdens arising from dairy systems which produce multiple (co)products such as milk and beef (Rice et al. 2017).

In this new systematic review, we synthesized a quantitative *and* qualitative dataset (see data in brief in Jebari et al. (2023)) of existing and potentially viable GHG mitigation measures and technologies which can be deployed on farms, regardless of whether they are arable, livestock, or mixed farms, including rotational systems. We refer to scientific literature and aggregated data that are key to the net-zero objectives, thus exploring environmental, economic, and societal perspectives for different mitigation measures.

2 Structure of the systematic review

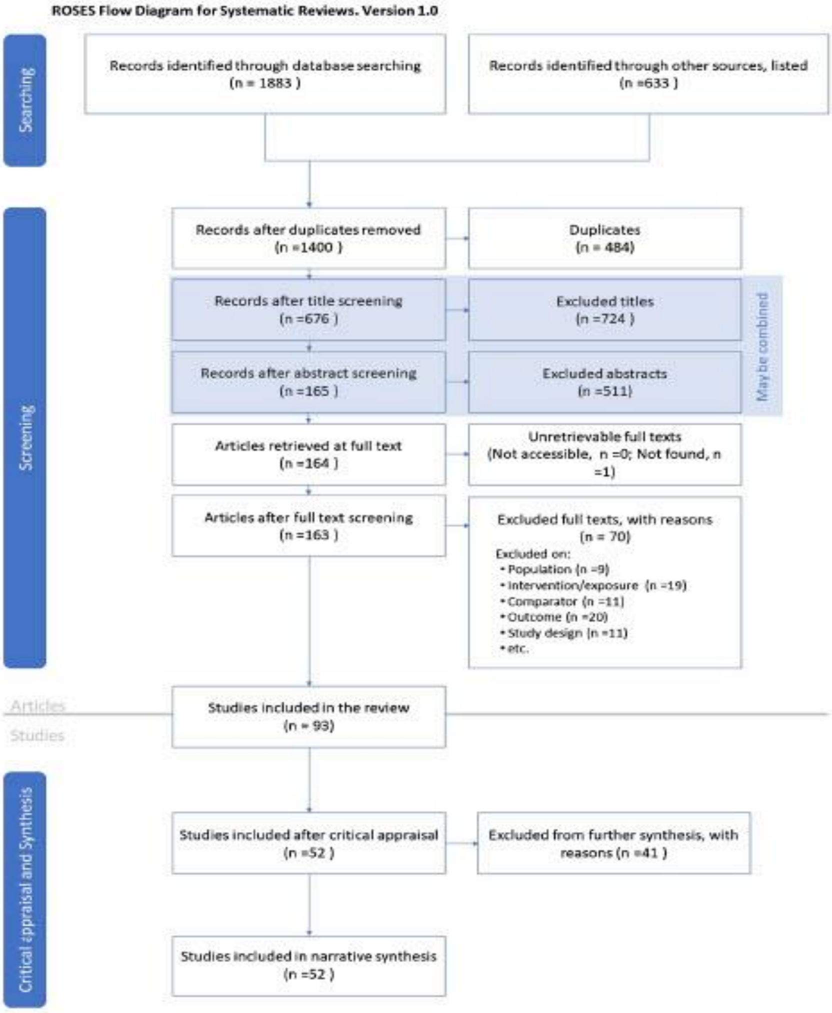
2.1 Search strategy

We followed the Collaboration for Environmental Evidence (CEE) guidelines and methodology therein to create our systematic review (CEE 2018) (Fig. 2). Only papers or reports published in English were considered for inclusion under the following structure:

Activity terms: "arable crops," "cereal," "wheat," "barley," "oilseed," "potato," "horticulture," "livestock," "dairy," "beef," "cattle," "pig," "sheep," "poultry," "chicken," "turkey," "mixed farm," "cow," "grassland," "pasture," "oat"
Intervention terms: "management," "practice measures," "alternative technology"
Outcome terms: "carbon footprint," "greenhouse gas emissions," "direct emissions," "indirect emissions," "methane," "nitrous oxide," "carbon dioxide," "ammonia," and "nitrate."

The search terms within each of the three categories (activity, intervention, and outcome) were combined using the Boolean operator "OR." We combined the three categories into a search string using the Boolean operator "AND." The search string was modified depending on the functionality of different databases (e.g., looking for keywords or topics), specialist sustainability-related websites, and search engines (e.g., *Scopus*). The temporal boundary of the literature search applied included recent relevant information and data published during the last 5 years (i.e., between 2017 and 2022). The purpose was to update the most recent literature and available technological advances in the agricultural sector of the UK. All the searches were performed in English in June 2022. The geographic boundary focused as far as feasibly possible on UK-specific literature; however, studies which covered multiple nations, including the UK, were also assessed. Despite focusing primarily on the aforementioned temporal boundary, older material sourced via "snowball" searching (i.e., identifying relevant sources of information via reference lists within the retrieved papers and reports) was also assessed to target novel, updated research streams. Recorded references were imported into *Mendeley* library and *Rayan* (online systematic review software) (Ouzzani

Fig. 2 RepOrting standards for Systematic Evidence Syntheses (ROSES) flow diagram (Haddaway et al. 2018) showing literature sources and inclusion/exclusion process.



et al. 2016). All duplicates were removed, and their numbers were recorded (Jebari et al. 2023).

2.2 Publication databases

The search included the following online scientific databases:

1. Web of Science Core Collection (https://mjl.clarivate.com/home)
2. Scopus (https://www.scopus.com/)
3. Rothamsted Repository (https://repository.rothamsted.ac.uk/)
4. British Library (ETHOS) (https://ethos.bl.uk/)
5. Formerly American Doctoral Dissertations (EBSCO) (http://search.ebscohost.com/login.aspx?authtype=ip,athens&custid=ns010809&group=main&profile=ehost)

Specialist websites of relevant UK organizations listed below were also searched in June 2022 for links or references to relevant articles and data (i.e., "snowball sampling," as mentioned previously), including grey literature:

1. Department for Environment and Rural affairs (Defra) (http://defra.gov.uk/)
2. National Farmers' Union (NFU) (https://www.nfuonline.com/)
3. Bangor University (http://www.bangor.ac.uk/)
4. North Wyke Publications Platform (https://www.rothamsted.ac.uk/north-wyke-farm-platform

2.3 Article screening and study eligibility criteria

Article screening was evaluated for relevance based on the eligibility criteria at three levels, title, abstract, and full text, using the systematic review software *Rayan*. Articles were first evaluated for eligibility based on their titles. The primary strategy was to be as inclusive as possible within the boundaries described in Section 2.1. Each article found to be relevant based on its abstract was judged for eligibility by screening the full text. The excluded articles dealt with keywords related to health or food industry (either upstream or downstream from the farmgate), and coastal and marine ecosystems, rather than agricultural systems. Additionally, phosphorus pollution was omitted due to its negligible impacts on GHG emissions (interactions of nutrients within soils and the influence of nutrient ratios, a complex topic, were beyond the scope of the current study). Moreover, experiments conducted outside the UK or under arid or Mediterranean climate conditions were also eliminated.

2.4 Study validity assessment

Eligible studies were subject to a critical appraisal. We assessed study validity and categorized relevant studies as "validated," "not validated," and "unclear validity" (the latter could also be considered "inconclusive"). Validity criteria included both susceptibilities to bias (internal validity: study design, strength of evidence, and reliability/replicability) and relevance of the study for our review questions (i.e., external validity). A study was excluded from the narrative synthesis due to internal validity if any of the following factors applied:

1. It does not have replicates (i.e., less than two independent experimental/observational units), in the case of experimental studies.
2. It does not include any uncertainty or sensitivity analysis or assessment of the predicted output against measured data, in the case of modelling studies.

If none of the above factors applied, the study was validated, as it complied with both external and internal validity (as explained above), whereas studies considered to possess unclear validity were subject to internal yet independent revision to judge whether the study is validated or not. A study was categorized to be "unclear" if it did not report sufficient details to judge its validity, for instance, if there is a vague methodological description or if it is difficult to interpret the efficacy of the mitigation measure discussed.

The final validated studies were included in the narrative synthesis. It is worth noting that we considered different agricultural systems and both modelling and experimental studies (Fig. 3a, b). The final list of included papers, which cover several mitigation measures with various impacts and objectively defined win-win strategies (i.e., reducing GHG emissions while improving agricultural productivity), was reported with recommendations for future research. Studies at the global scale were assessed in terms of the mitigation potential related to the UK.

2.5 Data extraction strategy

We extracted data (and metadata, where applicable) on study characteristics (e.g., whether the study deals with experimental or modelling approaches, or both), description of exposure, outcomes, and study findings. In the case of missing or vague quantitative values pertaining to GHG mitigation measures from the main manuscript, data from available supplementary material, as well as graphs using WebPlotDigitiser (https://automeris.io/WebPlotDigitizer/), were used. We also contacted authors for missing data. All extracted data were quality controlled. Quality control was conducted to identify the value of mitigation and implications of each mitigation measure.

To ensure that the extraction of data and metadata was replicable, entries were subsequently extracted by one author and cross-checked by another author as part of the quality control process. All disagreements amongst team members were discussed and the coding scheme was subsequently adjusted and clarified. Missing data were simply defined as "not stated."

2.6 Data synthesis and presentation

A qualitative synthesis of a semi-quantitative dataset was conducted as the primary goal to initiate a strategic pathway to net zero through the interpretation of state-of-the-art sustainability literature with a specific focus on GHG mitigation. The coding of the data presented in the synthesis is illustrated in supplementary Table S1 (see Supplementary Material). Our coding process refers to the different questions addressed in the introduction section, regarding the mitigation measure, and its implications in terms of economy, environmental sustainability (particularly GHG emissions), food security, and energy.

3 Mitigation measures

The outputs derived from the systematic review are summarized in Tables S1, S2, and S3, according to the primary mitigation measures under three pillars: reducing

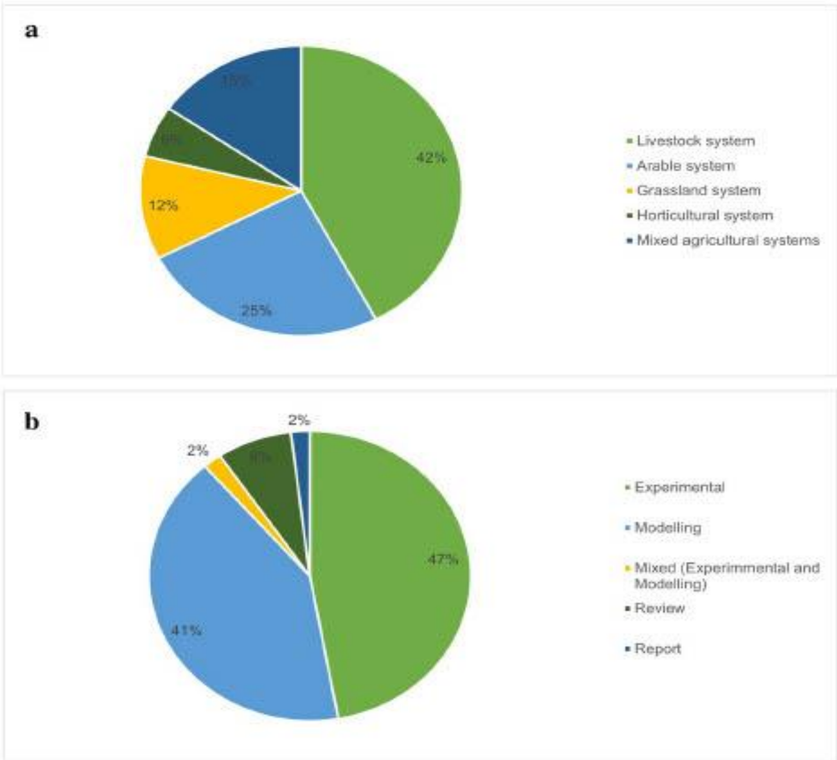


Fig. 3 Number of studies per agricultural system (**a**) and study category (**b**).

emissions and/or production efficiency (Table S1), offsetting, and bioenergy production (Table S3).

3.1 Arable systems' mitigation measures

Regenerative and soil conservation practices, such as cover cropping and reduced tillage, enhance SOC stocks while providing important ecosystem services such as enhancing water retention and reducing soil erosion. As a result, cover cropping with legumes for different arable crops demonstrates sequestration up to 800 kg C ha^{-1} $year^{-1}$ (Glenk et al. 2017) and could potentially sequester up to 16% C up to the year 2050 (Jordon et al. 2022). Similarly, reduced tillage was estimated to sequester up to 100 kg C ha^{-1} $year^{-1}$ (Glenk et al. 2017) and reduce up to 25% of GHG emissions at 5 cm depth in arable cropland (Alskaf 2018). In this context, a global meta-analysis of 946 paired data from 116 peer-reviewed studies showed that, overall, no tillage reduced global warming potential by 14.4% (Li et al. 2023).

Although cover crops and reduced tillage may imply a reduction in operational costs related to energy, they might induce a reduction in crop yield in the short and medium term (Glenk et al. 2017). On the other hand, cover crops maintain soil fertility in the longer term (Sun et al. 2011), thus signifying the need for policy intervention including financial incentives for farmers during the early stages of the transition from ploughing to conservation agriculture to offset potential yield reductions (Alskaf 2018).

Management practices including cover cropping and reduced tillage, as mentioned above, align with conservation agriculture through the improvement of soil and water quality by reducing runoff and leaching, enhancing water retention, and preventing soil erosion (Alskaf 2018; Warner et al. 2017). Accordingly, these measures should be targeted to geographic areas with higher erosion risk (e.g., hilly terrain and certain soil types) and where arable farming is found to contribute significantly to diffuse water pollution (Glenk et al. 2017). Weed management and pests including slugs were, however, identified as considerable challenges for reduced tillage adopters (Alskaf 2018). In this context, ley integration in arable rotation systems offset 27% of British agricultural emissions through SOC sequestration (Jordon et al. 2022), while simultaneously being adopted as a tool to control weeds which evolved to gain herbicide resistance (e.g., blackgrass; Jordon et al. 2022). The mitigation potential is lower at the European level when including leys in rotations with annual crops (i.e., 4 to 10%), according to Englund et al. (2023).

Soil amendment under specific edaphoclimatic conditions is considered to be a CO_2 removal technique. Particularly, soil amendment in the form of spreading crushed silicate rocks such as basalt to croplands, known as enhanced rock weathering, has been shown to be an effective mitigation measure in acidic loamy soils (Kelland et al. 2020). This mitigation measure aims to accelerate natural geological processes of SOC sequestration (as it enhances SOC stocks by a factor of four) and reduces energy demands for milling (occasionally carried out on-farm; McAuliffe et al. 2017) and associated carbon emissions from the use of fossil fuels (Lefebvre et al. 2019). According to Kelland et al. (2020), this intervention is beneficial for both farmers and the environment since economic gains derived from yield improvement could offset the purchase and operational costs of enhanced rock weathering. Moreover, a supplemental source of silicon (Si), calcium (Ca), and potassium (K) can be provided without any increase in toxic trace elements. These elements, apart from improving crop production, increase protection from pests and diseases, and restore soil fertility and structure (Beerling et al. 2018). As a consequence, the well-managed soil amendment addresses multiple UN Sustainable Development Goals (Smith et al. 2019) and contributes to net-zero objectives.

Similarly, another important soil amendment is the replacement of ammonium sulfate with a different form of sulfur (S) (e.g., single superphosphate, potassium sulfate, magnesium sulfate, calcium sulfate dihydrate (gypsum), and polyhalite (polysulphate)), which are most notably observed on high pH soils (Powlson and Dawson 2022). With each of these S fertilizers, the content of phosphorus (P), K, magnesium (Mg), or Ca needs to be considered when deciding on other nutrient applications (Powlson and Dawson 2022). Elemental S can also be used, but it is more slowly available to crops than the other forms as it must first be oxidized to sulfate by soil bacteria and the rate of conversion is somewhat unpredictable (Malhi et al. 2005). This relatively easy measure would make a significant contribution to reducing NH_3 emissions (i.e., by 90%; Powlson and Dawson 2022). Biochar application to soils has also been recommended as an important component of the pathway to "climate-smart soil" management practices in modern agriculture (Purakayastha et al. 2019). It has been shown to improve soil quality (soil bulk density, porosity, water retention, soil aggregation, and hydraulic conductivity; Purakayastha et al. 2019). Moreover, the increase in soil pH with biochar addition would result in a greater availability of primary and secondary nutrients like K, P, Ca, and Mg, as reported by Purakayastha et al. (2019).

Regardless of the pedoclimatic conditions, biosolid application to croplands provided valuable evidence in support of maintaining a sustainable agricultural landbank for biosolid recycling in the UK (Water UK 2010). Indeed, the mitigation measure helped to reduce up to 17% of GHG emissions (through SOC sequestration) in established experimental platforms at four sites in England with contrasting soil types and agroclimatic conditions (Nicholson et al. 2018). The mitigation potential through SOC accumulation in the latter study is comparable to 19% in Canadian croplands after biochar application (Gross et al. 2022). Moreover, biosolids amongst other environmentally positive impacts related to increasing water infiltration rate may improve soil quality and fertility. Biosolids contain valuable quantities of crop-available N, which can replace some of the required mineral fertilizer N together with increasing soil extractable P and total S for the plants (Rigby et al. 2016).

Acidification of digestate has been shown to be an effective mitigation measure for the utilization of food waste because it contributes to the mitigation of N losses (with around 95% reduction of cumulative NH_3 losses, which indirectly produce N_2O through microbial nitrification) following application to croplands (Sánchez-Rodríguez et al. 2018). This mitigation measure provides an environmentally sound option for N management and higher yields, as well as the production of renewable energy via anaerobic digestion (Kataki et al. 2017).

It is worth noting the importance of appropriate crop nitrogen management to avoid unnecessary trade-offs (e.g., potential increase in ammonia volatilization and nitrate leaching and ensure optimal crop production). In this context, Cammarano et al. (2021), for example, established an optimal N fertilizer rate of 120–140 kg N ha^{-1}, in malting barley production in order to maximize the economic return, maintain acceptable grain N%, and minimize environmental impacts including marine and terrestrial eutrophication.

3.2 Livestock systems

3.2.1 Manure management

Introducing anaerobic digestion to grassland-based livestock systems has demonstrated mitigation of the C footprint of livestock production (Webb 2017). For instance, the anaerobic treatment of dairy processing effluents showed a mitigation potential of 15.1 kg CO_2-eq according to Stanchev et al. (2020). Likewise, via predictive modelling based on the IPCC refined methodology, Scott and Blanchard (2021) simulated up to 44% reduction of total commercial dairy farm emissions through the adoption of anaerobic digestion. This is in line with Battini et al. (2014), as anaerobic digestion can lead to an over 30% reduction in GHG emissions, compared to traditional manure treatment. Although its implementation can be challenging, especially for small farms (due to the cost) or those located with insufficient access to water (Smith et al. 2021), anaerobic digestion provides diverse positive environmental impacts. For instance, high bioavailable N from digestate enables lower inorganic fertilizer requirements per hectare (Walsh et al. 2018). In addition, the application of bio-slurry as an organic fertilizer increases SOC sequestration (Walsh et al. 2018). Moreover, it enables pollution control by removing waste from the environment and reducing N and P discharge to the waterbodies (Scott and Blanchard 2021), and reduces land occupation and ozone depletion (Stanchev et al. 2020).

As briefly mentioned above, anaerobic digestion can be expensive and requires improvements in the maintenance of digesters to avoid increased emissions (Smith et al. 2021). However, energy savings from anaerobic digestion are important (NFU 2021b). Such savings are estimated to reduce 715 t CO_2-eq year^{-1} (41%) for commercial dairy farms in Northern Ireland (Scott and Blanchard 2021). Exploiting the CO_2 component of biogas and the ability to use CH_4 to power farm vehicles are seen as routes to achieve a reduction of 50% GHG emissions via offsetting (Scott and Blanchard 2021). Accordingly, government support could be instrumental in overcoming the costs of investment either using capital grants targeting the pollution reduction potential of systems or tax breaks and profitable tariffs to encourage the uptake of anaerobic digestion, thus providing renewable energy to the national grid (Scott and Blanchard 2021).

Applying additives to slurry (e.g., acidifiers alum, calcium chloride, and sulfuric acid) has shown abatements of NH_3 emissions up to 76% from confined dairy production (McIlroy et al. 2019). However, the technologies for the application of these additives in livestock housing need to be further developed (McIlroy et al. 2019). It is important to note that abatement techniques for manure management involve a holistic approach and should be implemented at both the storage and land spreading stages (Montes et al. 2013).

3.2.2 Grassland management: fertilization and extensification

Several mitigation measures related to N fertilization have proved to be efficient in terms of GHG mitigation. For instance, organic amendment scheduling compared to a traditional one-time application per season may be a useful on-farm mitigation measure for minimizing N_2O emissions (Shah et al. 2020). The use of high-frequency, low-dose organic fertilizer applications was predicted to reduce N_2O peak fluxes (up to 17%) for cattle slurry during the autumn and spring seasons (Shah et al. 2020). Furthermore, the optimal use of organic fertilizers has potential benefits compared to synthetic fertilizers, as it enhances forage yield and livestock productivity and soil quality (through SOC storage) and provides high-value organic food production with a suitable source of bioavailable soil nutrient replenishment (Zheng et al. 2010; Wang 2014; FAO 2017).

The application of nitrification inhibitors during fertilization has been shown to mitigate soil emissions (Chadwick et al. 2018). For instance, dicyandiamide (DCD) reduced N_2O emissions by ~13% under trampled grasslands and 53% under tractor compaction (Hargreaves et al. 2021). The reduction in N_2O emissions is accompanied by a decrease in NO_3 leaching and runoff, and NH_3 volatilization, all of which are indirect sources of N_2O (Cardenas et al. 2022). However, caution should be taken as issues have been raised when using nitrification inhibitors, as traces of DCD were found in milk when DCD was directly fed to animals (Welten et al. 2014). Further, swards from grasslands which received DCD have been reported to contain traces of DCD (Pal et al. 2016). Despite this concern, there is no defined threshold concentration for DCD in human-edible produce related to food safety as the compound has been reported to be non-toxic under typical application rates (OECD 2004).

Similarly, sodium chlorate ($NaClO_3$) amendment showed substantial mitigation potential with more than 60% reduction in the net nitrification rate under agricultural soils (Fu et al. 2018). Likewise, inhibited urea with *N*-(*n*-butyl)thiophosphoric triamide (NBPT) was shown to decrease NH_3 emissions within a range of 48–65% under grasslands in England and Wales (Carswell et al. 2019a). However, with no apparent yield differences compared to other N fertilizer sources (e.g., ammonium nitrate and urea), there is no economic incentive for the farmer to use the more environmentally acceptable option, unless externality costs are incorporated into fertilizer prices at the point of sale (Carswell et al. 2019a).

N fertilizer should be applied optimally through soil testing prior to applications when increasing yield potential. Perhaps, the most promising outcome of reduced N fertilizer input is the reduction associated with N leaching into waterbodies (which subsequently produces *indirect* N_2O) and *direct* GHG emissions during manufacture, transport, and application (Harris and Ratnieks 2022). The substitution of fertilizer nitrogen with symbiotically fixed nitrogen from legumes (e.g., white clover, *Trifolium repens*) within the range of 30–50% enables mitigation up to 58% g N_2O-N kg^{-1} DM yield compared to a baseline with a high fertilizer rate of 200 kg N ha^{-1} year^{-1} (Fuchs et al. 2020). This specific mitigation measure seems beneficial with respect to multiple outputs such as yields, N yields, and feeding values (Lüscher et al. 2014; McAuliffe et al. 2018), thereby improving animal health and welfare, through enhanced nutritional benefits (Carswell et al. 2019b). Indeed, root-node fixed N provides a supply of N for plants that is more bioavailable than occasional fertilizer applications and increases N use efficiency (Barneze et al. 2022) while improving diet-level sustainability (Costa et al. 2021). The biologically fixed N reduces energy costs associated with producing synthetic fertilizer with no reduction in productivity (Harris and Ratnieks 2022). Moreover, introducing local legumes has shown feasibility for replacing imported soy-based feeds, as reported by Costa et al. (2021). However, a potential limitation of this mitigation measure can be the challenge of achieving high and persistent legume proportions, particularly under grasslands receiving low sunlight or excessively cold growth periods (Barneze et al. 2022).

Moving towards extensification by reducing the livestock density and N fertilization has been underscored as a reliable mitigation measure (Sándor et al. 2018). The latter demonstrated a reduction of 78% in soil N_2O emissions for the mown and grazed site of Easter Bush (Edinburgh; Sándor et al. 2018). The reduction in soil N_2O emissions is within average estimations (~70%) for grasslands under similar conditions (e.g., France and Switzerland) (Sándor et al. 2018). The mitigation was accompanied by positive implications such as decreases in NH_3 losses and NO_3 leaching, thereby simultaneously reducing indirect N_2O emissions (Sándor et al. 2018). On the other hand, intensification, and the specialization in livestock production, for example, dairy systems, results in both an increase in C footprint, which relies on feed importation, and burdens such as eutrophication and acidification (Soteriades et al. 2019). The effect of ongoing trends in dairy farms can be mitigated by (i) increasing beef output per unit of milk achievable without a large change in a dairy farm's management and (ii) sustainable intensification of displaced beef-breeds production on suckler-beef farms (Soteriades et al. 2019). These measures can spare larger areas of land for forest (regionally or in major beef-exporting countries such as Brazil; Styles et al. 2018). Although this may reduce by up to 11–56% of burdens (i.e., GWP, eutrophication potential, acidification potential, and land occupation) per liter of milk (Soteriades et al. 2019), the investment in technology to maintain production levels and improve environmental efficiencies can be financially restrictive due to initial capital investment requirements (Dumont et al. 2013). Moreover, the positive environmental impacts of lower eutrophication and acidification potential could be negated by an increase in indirect land occupation related to animal feed cultivation (Gonzalez-Mejia et al. 2018).

Finally, regarding pork production, partly outdoor organic production where pigs spend part of the year outside and the rest indoors (seasonal housing) showed lower acidification, and thereby fewer indirect GHG emissions, than indoor systems. Conversely, traditional or "hardy" pig breeds which spend their lives outdoors yearly produce higher eutrophication potentials than semi-outdoor systems (Rudolph et al. 2018).

3.3 Livestock diets

3.3.1 Supplements to inhibit greenhouse gas production

While the use of biotechnological interventions can be challenging on a practical basis, feed additive supplementation appears to be the most researched and therefore the most "ready-to-use" mitigation measure to mitigate enteric CH_4 emissions and/or N_2O emissions for ruminants (Prathap et al. 2021). For instance, dietary nitrate and increased lipids included together could reduce enteric CH_4 emissions by 45% for finishing beef cattle (Duthie et al. 2018). This measure is achievable through the utilization of by-product feed such as rapeseed cake (Duthie et al. 2018). Potential adverse effects such as toxicity and impaired animal performance can be avoided by feeding low amounts of nitrate (Lee and Beauchemin. 2014). On the contrary, feeding nitrate to animals may increase N in excreta and therefore the trade-off between CH_4 and N_2O emissions reductions requires further research (Beauchemin et al. 2020). Similarly, supplementing dairy cow diets with oilseed-based preparations (e.g., extruded linseed or calcium salts of palm or linseed oil) as 22 g oil kg^{-1} DM showed a reduction of 10% of CH_4 emissions per kilogram of DM (Kliem et al. 2019). In a meta-analysis, Arndt et al. (2022) showed that feeding oils or fats versus oilseeds had comparable mitigation effects on total daily CH_4 production, with an average of 21% (ranging from 12 to 35%). This specific oilseed-based dietary mitigation measure is commercially practical with no negative effect on DM intake or milk fatty acid concentration (Kliem et al. 2019). However, it should be noted that feeding higher levels of oil supplements ($\geq$50 g oil kg^{-1} DM) can have a negative impact on ruminal and total tract organic matter and therefore neutral detergent fiber (NDF) digestion (Firkins and Eastridge 1994). Furthermore, using 2 g of liquorice extract for feeding animals (rich in prenylated isoflavonoids and particularly glabridin) might potentially improve the efficiency of N utilization and reduce CH_4 production in the rumen (Ramos-Morales et al. 2018). In this context, Ramos-Morales et al. (2018) conducted experiments which showed a reduction of 77% NH_3 emissions and 27% CH_4 emissions following the inclusion of 2 g extract of liquorice for sheep diets. The mitigation effect was accompanied with an improvement in feed conversion efficiencies by ruminants which subsequently increased their productivity (e.g., kg average daily gains). The invention of feed composition for ruminants comprising bis esters of hederagenin or ivy sapogenins (saponins are naturally occurring compounds that are widely distributed in all cells of legume plants) helped to mitigate ruminant emissions (Al Dulayymi et al. 2017). The synthetic molecule derives its name from its ability to form stable, soap-like foams in aqueous solutions and constitutes a complex and chemically diverse group of compounds including glycoside. The hederagenin bis esters have a persistent effect against ciliate protozoa in the rumen, without affecting the bacterial microflora, and feeding ruminants with doses of 50 mg to 1 g per kg per feed demonstrates a mitigation potential of up to 23% for enteric CH_4 emissions and up to 16% for NH_3 emissions (Al Dulayymi et al 2017). This dietary mitigation measure also helped to improve milk production and ruminant growth performance was observed to be more efficient (Al Dulayymi et al. 2017). In this context, several studies with saponins reported reduced CH_4 production from ~6 to 27% by reducing the protozoa population (Goel and Makkar 2012).

Other effective supplements for reducing enteric CH_4 emissions include concentrate supplementation with ground corn, essential oils, or acidic supplements (e.g., *encapsulated fumaric acid*) as well as certain plant secondary metabolites (e.g., grape marc; Prathap et al. 2021). Notably, a potential CH_4 inhibitor known as 3-nitrooxypropanol (3-NOP) is receiving much attention. 3-NOP has been shown to be effective in long-term studies with dairy and beef cattle (Melgar et al. 2020). 3-NOP decreases CH_4 production by 30% (Dijkstra et al. 2018; Kebreab et al. 2023). In general, the reduction of CH_4 emissions derived from enteric fermentation is within the large range of mitigation reported by UNEP (2021) at a global scale (i.e., 15 to 45%). However, farmers should be selective regarding this feeding practice, as some of the feed additives might be expensive (e.g., propionate precursors) or have side effects such as reduced calorie intake (e.g., halogenated compounds; Smith et al. 2021).

3.3.2 Modifying feeding regimes

Replacing a moderate proportion of total mixed ration-based diets with freshly cut and delivered grass or grass grazed at pasture for dairy cows showed a reduction in CH_4 emissions of up to 17% for the animals fed fresh cut grass and up to 39% for the grazing animals (Cameron et al. 2018). Within this mitigation measure, the costs of any longer-term reductions in milk yields may be outweighed by the benefits of improved farm profitability and reduced GHG emissions (Cameron et al. 2018). High-sugar grasses are thought to provide a better balance of N and carbohydrates to rumen microbes, thereby improving N and feed efficiency (Soteriades et al. 2018). In this context, re-seeding conventional permanent pastures (which occupy ~70% of UK-based agricultural land) with high-sugar grass varieties is seen as an attractive short-term measure for farmers by improving productivity, and reducing acidification and eutrophication impacts. However, it is important to note that primary data (e.g., digestibility and crude protein measurements as well as animal growth rates during grazing)–based assessments of high-sugar grass introduction suggest that the cultivar may produce more N_2O emissions and poorer animal performance compared to other swards such as those including legumes (e.g., white clover; *Trifolium repens*) under clayey soil types and temperate climatic conditions (McAuliffe et al. 2018; Mcauliffe et al. 2020b).

Grazing of dairy cows has also been shown to be effective with respect to SOC sequestration (Wilkinson et al. 2021). Pasture access benefits milk quality (i.e., milk produced on grass has higher levels of digestible protein as well as vitamin E and carotene; Wilkinson et al. 2021). Grazing dairy cows display behaviors including improved lying/resting times, lower levels of aggression, more normal estrous behaviors, and better synchronicity of behaviors compared to housed cows (Mee and Boyle 2020). Farmers are thereby encouraged to provide pasture access to dairy cows whenever weather conditions permit. Nevertheless, ruminant welfare is complex and there are different schools of thought about benefits and risks related to year-round housing, but when managed appropriately, improved welfare through grassland access has been shown to improve productivity and therefore reduce GHG emissions via fewer CH_4 and N_2O emissions (Rivero and Lee 2022). It is also worth mentioning that improving welfare (e.g., reducing lameness occurrences, preventing liver fluke, reducing stocking densities, and minimizing tuberculosis outbreaks) can actually marginally increase GHG emissions in certain livestock systems such as poultry while reducing water and soil pollution (Leinonen et al. 2014). In the case of ruminants, unintended consequences of improved animal welfare include reduced gross margins due to increased management costs (Rivero and Lee 2022). These complexities require further investigation to determine (a) whether the observed trade-offs can be balanced through mitigation measures and management practices (e.g., cell-grazing for ruminants) or (b) if one aspect of sustainability (i.e., environmental benefits, animal welfare improvements, or increased profitability) should be prioritized over the others. To add to the aforementioned complexities, other studies have demonstrated that improved profitability via high-quality management practices (e.g., high levels of feed conversion ratios) can in fact improve environmental health and economic performance simultaneously in intensive pig production systems (McAuliffe et al. 2017); despite this encouraging finding, implications for animal welfare require further exploration in the context of achieving net zero (see Section 3.4 for more information).

In terms of point (a) in the previous paragraph, feeding Ericaceous species (e.g., plants which thrive in low pH soils) to grazing sheep and red deer on heathlands is an effective mitigation measure to mitigate GHG emissions (Pérez-Barbería et al. 2020). Indeed, the mitigation measure balanced multiple trade-offs through improved cost-effectiveness, reduced the C footprint, and demonstrated biodiversity gains compared to other systems of animal production such as intensive farming (when animals are indoors, and fed on imported food and silage) (Gordon and Prins 2008). Ericaceous species also help to maintain traditional grazing culture and improve animal welfare (Pérez-Barbería et al. 2020). On the other hand, introducing high concentrate (e.g., barley or maize based) diets fed to different breeds of beef cattle during the finishing period helped to reduce up to 45% of CH_4 emissions, while increasing feed efficiency and propionate (a main precursor of glucose for ruminants) production, thereby decreasing CH_4 production in the rumen (Snelling et al. 2019).

Nevertheless, under grazing systems, larger areas of pasture may be needed to produce the same amount of throughput (Wilkinson et al. 2021). In this context, diets for livestock could be formulated to reduce the total feed-related C footprint and reduce the proportion of human-edible feed in the total diet (Wilkinson and Garnsworthy 2017). For instance, dairy cow diets formulated to include high proportions of by-product feeds such as dried distillers' grains can support high levels of milk output and are environmentally attractive compared with those based on grazed pasture or silage with concentrates (Wilkinson and Garnsworthy 2017). By-product utilization contributes to a circular economy via waste avoidance and reduction of "empty" (i.e., agricultural produce which ends up in landfills, incinerators, or slightly less burdensome, recycling centers) GHG emissions.

3.4 Livestock health and genetic performance

Although highly complex in terms of sustainability trade-offs, as introduced in Section 3.3, improving livestock health has been shown to have positive environmental and societal benefits in certain livestock systems; for instance, the reduction of GHG emissions arising from livestock production can be delivered by reducing the maintenance of poorly performing animals through genetic selection (Llonch et al. 2017; McAuliffe et al. 2018). Improving health can lead to improvements in the parameters that ruminants' emission intensities are sensitive to, e.g., maternal fertility, abortion rates, and cow mortality rates, while calf, ewe, and lamb mortality rates and growth rates, milk yields, and feed conversion rates are also important factors to improve (MacLeod and Moran 2017). Regarding C "credits," the marginal cost for livestock health improvement was higher than -100 £ t^{-1} CO_2-eq for cattle and lower than 50 £ t^{-1} CO_2-eq for sheep production (MacLeod and Moran 2017). Similarly, performance recording technology showed livestock production's potential to be C efficient, thus adhering to growing public demands on climate change and animal welfare simultaneously (Morgan-Davies et al. 2021). For instance, using performance recording on sheep farms in order to achieve higher genetic merit mitigated up to 18% of GHG emissions (3.5 CO_2-eq kg liveweight^{-1}) and increased economic margins by £6 ewe^{-1}, thereby ensuring enhanced food security and lower climate-related impacts; however, this management practice incurred 10% extra labor with ramifications for profit-loss margins (Morgan-Davies et al. 2021). Moreover, future animal breeding schemes may include a wider range of traits linked to environmental emissions apart from production and health traits (Gill et al. 2021). Wallace et al. (2019) reported that a heritable subset of the core rumen microbiome dictates dairy cow productivity and CH_4 emissions. As alluded to above, in theory, it should then be possible to select ruminants with specific rumen microbiomes suited to different production systems, leading to higher feed efficiency (e.g., through increased digestible energy) and lower CH_4 emissions. This is a notable finding as improvements to the biological performance of ruminants fall behind the performance of monogastrics which are easier to increase feed conversion efficiencies due to the absence of rumen microbial communities.

Considering livestock bedding material, straw is commonly used and often transported long distances from arable to livestock regions (Copeland and Turley 2008). This process is becoming increasingly unsustainable and uneconomical as the demand and price for straw increase (Wonfor 2017). Alternative bedding materials (for instance, coppice willow and miscanthus) cultivated directly on livestock farms could potentially avoid transport-related emissions and competition for use (Glithero et al. 2013). In this context, the use of miscanthus bedding production on livestock farms and the substitution of fossil fuels with straw in electricity generation have been shown to provide environmental benefits (Yesufu et al. 2020). This mitigation measure is considered to be cost-effective and capable of reducing GHG emissions by ~9 million t CO_2-eq at a UK level and also minimizes both eutrophication and acidification burdens (Yesufu et al. 2020).

3.5 Horticultural systems on peatlands

Around 40% of UK peatlands have been drained for agricultural use, namely horticultural cultivation, which has caused serious peat wastage and associated GHG emissions (CO_2 and CH_4; Dixon et al. 2014). While peatland drainage increases CO_2 loss into the atmosphere, natural peatlands are sources of CH_4 due to methanogenic activity under their prevalent waterlogged anoxic soil conditions. To address GHG emissions and C losses, water tables should be raised (or lowered if applicable) to reduce GHG emissions from agricultural peatlands while simultaneously maintaining the current levels of horticultural productivity (Musarika et al. 2017). For instance, increasing the water table to −40 cm presented a possible compromise to decrease peat oxidation and maintain romaine lettuce production (Matysek et al. 2022). Similarly, raising the water table from −50 to −30 cm in lowland fen peatland used for radish production reduced GHG emissions (i.e., CO_2 by 89% and CH_4 by 58%), while maintaining the same yield production (Musarika et al. 2017). Likewise, maintaining a high-water table in different horticultural peatlands helped to reduce the global warming potential by approximately 30% (Taft et al. 2018). However, it is important to bear in mind that this mitigation measure may be impractical to implement within current horticultural systems. For instance, raising the water table to within 15 cm of the soil surface would not be implemented while a crop was in place, as it would likely result in high crop mortality and thus be unsuitable for field trafficking. Instead, this intervention would probably need to be implemented between summer crops, possibly over quite short fallow periods (Taft et al. 2018). Optimizing the water table in agricultural peatlands contributes significantly to economic development in many areas (Evans et al. 2021) and promotes food security (Taft et al. 2018).

3.6 Mixed farm systems and their role in sustainable agriculture

Integrated farming under horticultural and crop systems has demonstrated the capability to mitigate more than 100% of GHG emissions, while enhancing food health and promoting agricultural sustainability (Abdul-Salam et al. 2019). Integrated farming involves cover crops, legumes, conservation tillage, reduced mineral fertilizer, pesticide and herbicide applications, and soil amendments to increase SOC content. However, since the relative financial performance of conventional farm systems is better than many low-carbon integrated farm systems, price premiums of up to 20% for integrated farming would help to enhance their economic performance to be comparable with conventional farming (FWI 2017; Abdul-Salam et al. 2019). In this way, consumers are increasingly sourcing low-carbon produce and paying extra as a way of improving their food health and contributing to reductions in their C footprints (Abdul-Salam et al. 2019).

Under both croplands and grasslands, several practices could be implemented to maximize crop nutrient utilization and to minimize emissions to the environment. As an "environmentally benign" material, applying green/food composts (characterized by lower N content, compared to food digestate and slurry) reduced N_2O emissions by up to 54% while accumulating long-term soil organic N reserves and improving soil structure and nutrient composition (Nicholson et al. 2017). Farmers are also advised to apply food-based digestate, as a provider of renewable energy, in the spring where practically possible, or in autumn to an actively growing crop such as grass or oilseed rape (Nicholson et al. 2017). Under this management, the crop will take up available N from the soil which will not be lost via overwinter NO_3 leaching (Nicholson et al. 2017). Similarly, bandspreading is thought to be effective at reducing NH_3 emissions (up to ~70%) from slurry instead of surface broadcasting (Nicholson et al. 2017). Precision application (i.e., bandspreading) provides numerous other advantages over broadcast applications: for example, more accurate assessment of application rates, the ability to apply from tramlines, reduced odor and crop damage, and a cleaner sward can be achieved (Nicholson et al. 2017). However, the effectiveness of this technique is dependent on the prevailing soil conditions (Nicholson et al. 2017).

Within arable and livestock systems, when using the by-products of whisky production to replace alternative feed ingredients (such as imported soya meal) for livestock, notable reductions of GHG emissions were shown (associated with land use changes, and to a lesser extent with enteric fermentation, manure management, and the end use of manure and its potential to replace synthetic fertilizers) (Leinonen et al. 2018). As briefly discussed in Section 3.3, distillery by-products could also be used as anaerobic digester feedstock to generate renewable energy (heat and electricity), though the mitigation potential as animal feed is lower than using it as human-edible ingredients (0.703 to 0.759 kg CO_2-eq kg^{-1} DM of by-product used for human consumption, compared to 0.101 to 1.219 kg CO_2-eq kg^{-1} DM of by-product used for animal feed; Leinonen et al. 2018). When used as an organic fertilizer, digestate arising from the anaerobic digestion process is high in N and P, as well as C, thereby simultaneously accumulating SOC *and* reducing the need for synthetic fertilizers (Leinonen et al. 2018), which are a major source of agri-food related GHG emissions.

3.7 Offsetting greenhouse gas emissions on agricultural land

Agroforestry systems deliver environmental benefits through C uptake compared with grasslands or croplands without trees (Jordon et al. 2020). Agroforestry, including silvopasture systems, shelterbelts, windbreaks, riparian buffer strips, hedges, wood pasture, forest grazing, orchards, woody biofuel, and farm woodlands, is gaining considerable attention from the perspective of agricultural sustainability, particularly in terms of net-zero ambitions globally. For instance, in terms of GHG mitigation and SOC sequestration, forest regeneration on sheep pasture with natural regeneration or forest plantation showed a mitigation potential of up to 85 t CO_2-eq ha^{-1} and 147 t CO_2-eq ha^{-1}, respectively, over 25 years (O'Neill et al. 2020). Moreover, planting red alder trees into sheep-grazed pasture showed a CO_2 mitigation potential of 47.5 to 99 Mg C ha^{-1}, after 20 years, for different types of red alder trees (Nworji 2017). Likewise, land use change by either afforestation with species of broadleaf trees (planted at 800 or 1600 stems ha^{-1}), or reversion to rough grassland, showed both soil N and C accumulation increasing SOC up to 46% and 334%, respectively, for 21 years (Baddeley et al. 2017). When pragmatically feasible, establishing hedgerows and field margins in arable landscapes and agroforestry systems could provide up to 63 t C ha^{-1} (Dunn et al. 2021). The mitigation potential is comparable to the estimated 81.7 ± 28.8 t C ha^{-1} for hedgerows in Belgium (Van Den Berge et al. 2021). Similarly, Crous-Duran et al. (2020) using modelling showed that introducing trees in arable systems allowed the sequestration of up to ~400 t C ha^{-1} in high tree-density agroforestry systems. Likewise, Poulton et al. (2018) analyzed rates of SOC increase in the treatments on 16 long-term experiments in the southeast UK. The latter study showed that the conversion from cropland to grassland or woodland enhanced SOC sequestration exceeding *4 per 1000* SOC stocks per year in the case of woodlands and reaching 55% in the case of grasslands. More widely, under the European territory, agroforestry implementation in the priority areas (areas with the highest number of accumulated pressure), which made up 8.9% of total European farmland, would reduce between 1.4 and 43% of European agricultural GHG emissions, depending on the type of the agroforestry (Kay et al. 2019). In addition, several environmental impacts could be reduced under agroforestry systems due to microclimate amelioration through the windbreak effect of the trees, the conservation of soil and water, and wildlife habitats as well as the forest productivity and sustainability through C uptake, thereby GHG offsetting contributing to cross-sector net-zero targets (Nworji 2017; Jordon et al. 2020).

It is worth noting that the viability of land use conversion to agroforestry, without subsidies, depends on low farm performance, a strong likelihood of natural regeneration, and a high carbon-market price. For instance, Burgess and Rosati (2018) confirmed that silvopastoral systems are not financially profitable (compared to silvoarable systems) but they provide the greatest societal benefit if environmental externalities are included. Accordingly, imposing, e.g., carbon payments or penalties for nutrient or soil loss pollution, would make agroforestry a more financially profitable opportunity for sustainable food production and security (Kay et al. 2019). In other words, financial aid for woodland establishment, a strategy being deployed in the UK by the "Woodland Trust," makes planting trees to sequester C financially viable (O'Neill et al 2020). However, other studies, such as Crous-Duran et al. (2020), showed that introducing trees in different farming systems such as arable and pasture, as a solution for additional environmental benefits, maintained similar levels of productivity. Afforestation mitigation measures provide economic benefits in terms of monetary value (e.g., harvesting wood for paper pulp or heating fuel which would offset fossil fuel depletion and associated GHG emissions), job creation, and financial income for rural economies as well as contributing to the circular economy if managed appropriately (Dunn et al. 2021). Many of the "tree outputs" have different established markets such as timber, food, energy, recreation, and non-timber forest products (e.g., foliage, biochar, and Christmas trees), which offer a developing or niche opportunity for farm enterprises to enhance ecosystem services (Pagella and Whistance 2019). Decision support tools should be offered at the planning stage of farm woodland schemes to aid farmers in tree species selection and assessment of benefits and trade-offs (Wiik et al. 2019). It is also important to bear in mind that the rate of SOC increase slows as the new equilibrium value (i.e., reaching SOC saturation) is approached and that increases are reversed if the modified management practices are not continued (Smith 2014).

Widespread adoption, however, would have a negative impact on global food security, e.g., converting agricultural land to forest or grassland (Poulton et al. 2018). Conversion to grasslands and woodlands could be convenient in limited situations where soils are either of low productivity or are fragile and prone to erosion, to ensure food security (Albanito et al. 2016). Moreover, afforestation should be accompanied by a shift in diet away from meat and dairy products. This change is necessary because without it, it would be necessary to import additional meat and dairy products from overseas (Dunn et al. 2021).

3.8 Bioenergy production

The CCC identified that bioenergy coupled with carbon capture and storage (BECCS) could deliver a significant reduction of up to 53 Mt CO_2-eq by 2050 (BEIS 2021). Indeed, bioenergy crops help mitigate climate change through displacing fossil fuel energy generation while removing CO_2 from the atmosphere and storing it in soils. This is the case with willow and miscanthus which both offer biomass production and higher SOC sequestration rates (with up to 12% increase in soil depths of 0–0.3 m) when planted in arable soils (Gregory et al. 2018). Robertson et al. (2017) estimated that the miscanthus-derived soil C accumulated a rate of 860 kg C ha^{-1} $year^{-1}$ over the top 30 cm. Therefore, miscanthus cropping could be attributed as a CO_2-sink related to an additional credit from soil C sequestration in the soil during the cultivation period, as confirmed in the Felten et al. (2013) study in Western Germany. Harris et al. (2017) showed that the conversion of grassland to short rotation coppice bioenergy willow converted the system from a net C source of 119 g C m^{-2} $year^{-1}$ to a net sink, −620 g C m^{-2} $year^{-1}$.

However, in the UK, conversion of grassland to bioenergy cropping systems represents one of the most significant potential land use transitions, as grasslands are a considerable part of the UK landscape (4–5 10^6 ha; Defra et al. 2007) and management of grasslands can vary widely in the UK, particularly with respect to fertilizer input and grazing strategies (Harris et al. 2017). As a consequence, it is desirable that bioenergy crops are concentrated on less-productive "marginal" land to minimize conflict between food and bioenergy production on higher-quality soils (McCalmont et al. 2017).

Lastly, poultry litter has been shown to perform better than miscanthus for most of the impacts. In this sense, gasification of poultry litter to produce electricity and heat generation in the UK could save 1.7 Mt CO_2-eq year^{-1}, equivalent to around 0.4% of UK's GHG emissions (Jeswani et al. 2019). However, owing to high capital costs, the unsubsidized cost of generating heat and electricity from poultry litter is similar to that of natural gas heat and power but significantly cheaper than that from other fossil fuel alternatives within an abatement cost of £34 t^{-1} CO_2-eq. This signifies that animal waste (by-product) management is a critical research stream in the context of agriculture's contribution to a net-zero economy.

4 Limitations and critical gaps for future research

4.1 Limitations

Our findings on GHG mitigation measures applied in the UK are applicable to broader geographies under similar climatic conditions. Despite adhering to a standard operating procedure for systematic reviews, our synthesis of results did not apply streamlined effect size predictions of the benefits and risks surrounding individual (or combined) GHG mitigation measures as the data extracted was not consistent in terms of agricultural systems, mitigation measures, and edaphoclimatic conditions in the UK (Jebari et al. 2023); as a result, this made statistical analyses of these reviewed measures' potential to contribute to the UK's net-zero ambitions infeasible. Likewise, emission reductions were provided per area or per kilogram of product. However, emission reductions per area may imply a caveat associated with reductions in productivity. Further, although the resulting dataset provides novel information to guide future research in the context of agriculture's net-zero achievements, the results should be interpreted with caution as they could potentially be misleading within the study's geographic boundary due to the low UK-specific literature sample size ($n = 52$). Despite this limitation, the resultant dataset (Jebari et al. 2023) provides a simple, yet comprehensive progress to communicate cutting-edge sustainability research with the farming community, thereby enabling qualitative analyses to guide future scientific efforts which are economically (e.g., capital investment requirements) and socially feasible.

4.2 Critical gaps for future research

As touched upon throughout the examination of literature, knowledge gaps were highlighted in our findings related to the implications of various mitigation opportunities for the UK's agricultural systems. While the environmental impacts of different mitigation measures have been investigated extensively, other impacts remain poorly understood. For instance, barriers on the adoption of the mitigation measures for the farmer, in terms of ease of maintenance or installation and operational costs, have been overlooked by 49% of the reviewed literature (see dataset; Jebari et al. (2023)). In this context, information on the attitudes of farmers towards the different management practices is needed (Collins et al. 2016), as farmers make the management decisions for most agricultural land in the UK (Harris and Ratnieks 2022). Engaging farmers on the issue of climate change mitigation (e.g., via participatory extension programs, surveys, and workshops, where farmers are allowed to share their feedback) is one option to address this current important knowledge gap (Knook et al. 2020). This bridge between scientists and farmers has already been established as part of another complementary, collaborative, and nationwide research stream which aims to identify which mitigation measures should be explored more rigorously from the agricultural community's perspective (see Section 4.1).

Moreover, the energy implications of the mitigation measures (i.e., whether the mitigation measure implies energy consumption reductions or increases) were not considered in 52% of the studies reviewed herein, even though entire food supply chains are major energy users and contributors to climate change (Rosa et al. 2021). Similarly, food security provision was overlooked in 51% of the studies reviewed, despite the potential negative trade-offs between food security and climate mitigation (Fujimori et al. 2019). Particularly, the import requirement induced by the mitigation measure was stated in only 15% of the retained studies.

Although the financial viability (in monetary and/or productivity terms) of the mitigation measures was considered in most of the studies reviewed (> 77% of studies), the marginal abatement cost (i.e., the average cost of reducing 1 ton of CO_2 equivalent) was rarely considered. The latter was not mentioned in 90.6% of the studies, which could be considered a major knowledge gap for future research. The cost-effectiveness of mitigation measures can change in response to factors such as commodity prices and the indirect effects of non-GHG policy (MacLeod et al. 2010). Even though prices and/or costs are fluctuating with time (Tang et al. 2021), marginal abatement cost information of potential mitigation measures has been shown to help policymakers identify the most recent cost-effective GHG mitigation options (Eory et al. 2018). As a consequence, the generation of accurate information on the cost-effectiveness of the mitigation measures is needed for effective government policies.

5 Conclusions

We synthesized existing evidence for several agricultural management practices and technologies, which can be deployed on farms, in order to help mitigate climate change. In many cases, the mitigation measures provided co-benefits for farmers, including improving farm productivity and diversifying farm income through energy generation. Well-implemented measures also result in environmental co-benefits in addition to mitigating climate change, including biodiversity, soil health, and other ecosystem services related to human health and animal welfare. However, it is also important to look at the sustainability from the farmers' perspective. Uneconomic practices for farmers (e.g., bioenergy industrial plants, agroforestry establishment) could be potentially overcome by government changes in regulations and subsidies to ensure greater financial viability by compensating for initial high capital costs. We have synthesized the evidence base within existing literature (Jebari et al. 2023), primarily focusing on the relevance to the UK's GHG strategies up to 2050 and the identification of opportunities and risks which require further attention. Our open-access dataset (Jebari et al. 2023) can inform scientists and policymakers on state-of-the-art GHG-related studies and guide funding bodies to target areas, which need urgent attention. Finally, net-zero achievement and relevant government policies need to be examined more holistically (e.g., accounting for unintended consequences such as farmers' well-being and animal welfare) in the context of business resilience and broad sustainability. This is particularly pertinent to food security as there is an ever-increasing population, which only the agri-food sector as a whole can sustain.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s13593-023-00938-0.

Authors' contributions AJ: conceptualization; methodology; validation; formal analysis; investigation; data curation; writing original draft; visualization. FP: data curation. AK: data curation. ALC: conceptualization; writing—review and editing; supervision. MJR: funding acquisition. GAM: conceptualization; resources; writing—review and editing; supervision; project administration; funding acquisition.

Funding Asma Jebari was financially supported by the Science Initiative Catalyst Award (SICA) program, an internal UKRI (UK Research and Innovation) Biotechnology and Biological Sciences Research Council (UKRI-BBSRC) award. Fabiana Pereyra was supported by Beca de movilidad ANII (Agencia Nacional de Investigacion e Innovacion) MOV_CA_2021_1_171482. Atul Kumar acknowledges funding from the UK Government's Department for Environment, Food, and Rural Affairs (Defra). Graham McAuliffe and Adrian L. Collins acknowledge funding from Rothamsted Research's Institute Strategic Program "Soil to Nutrition" (S2N) supported by UKRI-BBSRC BBS/E/C/000I0320 & BBS/E/C/000I0330.

Data availability The datasets generated and/or analyzed during the current study are available in the Mendeley repository, Jebari et al. (2023) Dataset on agricultural greenhouse gas mitigation measures in the UK, Mendeley Data, V1, https://doi.org/10.17632/t9kynfj5jf.1.

Code availability Not applicable

Declarations

Ethics approval Not applicable

Consent to participate Not applicable

Consent for publication Not applicable

Competing interests The authors declare no competing interests.

References

Abdul-Salam Y, Hawes C, Roberts D, Young M (2019) The economics of alternative crop production systems in the context of farmer participation in carbon trading markets. Agroecol Sust Food 43(1):67–91. https://doi.org/10.1080/21683565.2018.1537986

ADAS (2019) Mitigation against GHG emissions: agricultural practices review. https://www.gov.wales/mitigation-against-greenhouse-gas-emissions-agricultural-practices-review. Accessed 07 August 2023

Al Dulayymi J R, Baird M, Bouillon M.E, Duval S, Ramos Morales E, New Bold C J, Preskett D, Radek B, Strawson S, Wehrli C, Lahmann M (2017) New bis esters of ivy sapogenins for ruminants. https://research.bangor.ac.uk/portal/en/researchoutputs/new-bis-esters-of-ivy-sapogenins-for-ruminants(307ea516-10a8-477a-8fbc-13e402a4157b).html. Accessed 25 June 2022

Albanito F, Beringer T, Corstanje R, Poulter B, Stephenson A, Zawadzka J, Smith P (2016) Carbon implications of converting cropland to bioenergy crops or forest for climate mitigation: a global assessment. GCB Bioenergy 8:81–95. https://doi.org/10.1111/gcbb.12242

Alskaf K (2018) Conservation agriculture for sustainable land use: the agronomic and environmental impacts of different tillage practices and plant residue retention: farmer uptake of reduced tillage in England. PhD thesis, University of Nottingham. http://eprints.nottingham.ac.uk/51902/. Accessed 25 June 2022

Arndt C, Hristov AN, Price WJ, McClelland SC, Pelaez AM, Cueva SF, Oh J, Dijkstra J, Bannink A, Bayat R, Crompton LA, Eugène MA, Enahoro D, Kebreab E, Kreuzer M, McGee M, Martin C, Newbold CJ, Reynolds CK, Schwarm A, Shingfield KJ, Veneman JB, Yáñez-Ruiz DR, Yu Z (2022) Full adoption of the most effective strategies to mitigate methane emissions by ruminants can help meet the 1.5 C target by 2030 but not 2050. Proc Natl Acad Sci USA 119:e2111294119. https://doi.org/10.1073/pnas.2111294119

Baddeley JA, Edwards AC, Watson CA (2017) Changes in soil C and N stocks and C: N stoichiometry 21 years after land use change on an arable mineral topsoil. Geoderma 303:19–26. https://doi.org/10.1016/j.geoderma.2017.05.002

Barneze AS, Whitaker J, McNamara NP, Ostle NJ (2022) Interactions between climate warming and land management regulate greenhouse gas fluxes in a temperate grassland ecosystem. Sci Total Environ 833:155212. https://doi.org/10.1016/j.scitotenv.2022.155212

Battini F, Agostini A, Boulamanti AK, Giuntoli J, Amaducci S (2014) Mitigating the environmental impacts of milk production via anaerobic digestion of manure: case study of a dairy farm in the Po Valley. Sci Total Environ 481:196–208. https://doi.org/10.1016/j.scitotenv.2014.02.038

Beauchemin KA, Ungerfeld EM, Eckard RJ, Wang M (2020) Review: fifty years of research on rumen methanogenesis: lessons learned and future challenges for mitigation. Animal 14:s2–s16. https://doi.org/10.1017/S1751731119003100

Beerling DJ, Leake JR, Long SP, Scholes JD, Ton J, Nelson PN, Bird M, Kantzas E, Taylor L, Sarkar B, Kelland M, DeLucia E, Kantola I, Müller C, Rau G, Hansen J (2018) Farming with crops and rocks to address global climate, food and soil security. Nat Plants 4:138–47. https://doi.org/10.1038/s41477-018-0108-y

BEIS (2021) The Role of Biomass in achieving net zero. https://assets.publishing.service.gov.uk/media/607d9fbc8fa8f51b96b126a0/role-of-biomass-achieving-net-zero-call-for-evidence.pdf. Accessed 20 June 2022

BEIS (2022) Final UK greenhouse gas emissions national statistics: 1990 to 2020. https://www.gov.uk/government/statistics/final-uk-greenhouse-gas-emissions-national-statistics-1990-to-2020. Accessed 20 June 2022

Brown P, Cardenas L, Choudrie S, Jones L, Karagianni E, MacCarthy J, Passant N, Richmond B, Smith H, Thistlethwait G, Thomson A, Turtle L, Wakeling D (2020) UK Greenhouse Gas Inventory, 1990 to 2018: annual report for submission under the framework convention on climate change. Report no. ED62689/0/CD9487/PB, Ricardo Energy & Environment, Department for Business, Energy & Industrial Strategy, London. https://uk-air.defra.gov.uk/assets/documents/reports/cat09/2004231028_ukghgi-90-18_Main_v02-00.pdf. Accessed 18 June 2022

Burgess PJ, Rosati A (2018) Advances in European agroforestry: results from the AGFORWARD project. Agroforest Syst 92:801–810. https://doi.org/10.1007/s10457-018-0261-3

Cain M, Lynch J, Allen MR, Fuglestvedt JS, Frame DJ, Macey AH (2019) Improved calculation of warming-equivalent emissions for short-lived climate pollutants. npj Clim AtmosSci 2:29. https://doi.org/10.1038/s41612-019-0086-4

Cameron L, Chagunda MGG, Roberts DJ, Lee MA (2018) A comparison of milk yields and methane production from three contrasting high-yielding dairy cattle feeding regimes: cut-and-carry, partial grazing and total mixed ration. Grass Forage Sci 73:789–797. https://doi.org/10.1111/gfs.12353

Cammarano D, Basso B, Holland J, Gianinetti A, Baronchelli M, Ronga D (2021) Modeling spatial and temporal optimal N fertilizer rates to reduce nitrate leaching while improving grain yield and quality in malting barley. Comput Electron Agric 182:105997. https://doi.org/10.1016/j.compag.2021.105997

Cardenas LM, Olde L, Loick N, Griffith B, Hill T, Evans J, Cowan N, Segura C, Sint H, Harris P, McCalmont J (2022) CO_2 fluxes from three different temperate grazed pastures using Eddy covariance measurements. Sci Total Environ 831:154819. https://doi.org/10.1016/j.scitotenv.2022.154819

Carswell A, Shaw R, Hunt J, Sánchez-Rodríguez AR, Saunders K, Cotton J, Hill PW, Chadwick DR, Jones DL, Misselbrook TH (2019) Assessing the benefits and wider costs of different N fertilisers for grassland agriculture. Arch Agron Soil Sci 65:625–639. https://doi.org/10.1080/03650340.2018.1519251

Carswell AM, Gongadze K, Misselbrook TH, Wu L (2019) Impact of transition from permanent pasture to new swards on the nitrogen use efficiency, nitrogen and carbon budgets of beef and sheep production. Agric Ecosyst Environ 283:106572. https://doi.org/10.1016/j.agee.2019.106572

CCC (2020) Land use: policies for a net zero UK land use: policies for a net zero UK. Climate Change Committee, UK. https://www.theccc.org.uk/publication/land-use-policies-for-a-net-zero-uk/. Accessed 26 June 2022

Chadwick DR, Cardenas LM, Dhanoa MS, Donovan N, Misselbrook T, Williams JR, Thorman RE, McGeough KL, Watson CJ, Bell M, Anthony SG (2018) The contribution of cattle urine and dung to nitrous oxide emissions: quantification of country specific emission factors and implications for national inventories. Sci Total Environ 635:607–617. https://doi.org/10.1016/j.scitotenv.2018.04.152

CIEL (2020) Centre for innovation excellence in livestock. Net Zero Carbon & UK Livestock Report. York 2020. https://cielivestock.co.uk/expertise/net-zero-carbon-uk-livestock/. Accessed 20 June 2022

CIEL (2022) Centre for innovation excellence in livestock. Net Zero Carbon & UK Livestock Report April 2022. https://cielivestock.co.uk/expertise/net-zero-carbon-uk-livestock/report-april-2022/. Accessed 30 June 2022

Collaboration for Environmental Evidence. Guidelines and standards for evidence synthesis in environmental management. Version 5.0. In: Pullin AS, Frampton GK, Livoreil B, Petrokofsky G, editors (2018) http://www.environmentalevidence.org/information-for-authors. Accessed 20 June 2022

Collins AL, Zhang Y (2016) Exceedance of modern "background" fine-grained sediment delivery to rivers due to current agricultural land use and uptake of water pollution mitigation options across England and Wales. Environ Sci Policy 61:61–73. https://doi.org/10.1016/j.envsci.2016.03.017

Collins AL, Zhang YS, Winter M, Inman A, Jones JI, Johnes PJ, Cleasby W, Vrain E, Lovett A, Noble L (2016) Tackling agricultural diffuse pollution: what might uptake of farmer-preferred measures deliver for emissions to water and air. Sci Total Environ 547:269–281. https://doi.org/10.1016/j.scitotenv.2015.12.130

Collins AL, Zhang Y, Upadhayay HR, Pulley S, Granger SJ, Harris P, Sint H, Griffith B (2021) Current advisory interventions for grazing ruminant farming cannot close exceedance of modern background sediment loss – assessment using an instrumented farm platform and modelled scaling out. Environ Sci Policy 116:114–127. https://doi.org/10.1016/j.envsci.2020.11.004

Copeland J, Turley D (2008) National and regional supply/demand balance for agricultural straw in Great Britain. York, UK: National Non-Food Crops Centre, pp. 1– 17. http://www.ruraldevelopment.org.uk/northwoods/files/2012/12/StrawAvailabilityinGreatBritain.pdf. Accessed 30 June 2022

Costa MP, Reckling M, Chadwick D, Rees RM, Saget S, Williams M, Styles D (2021) Legume-modified rotations deliver nutrition with lower environmental impact. Front Sustain Food Syst 5:656005. https://doi.org/10.3389/fsufs.2021.656005

Crous-Duran J, Graves AR, García de Jalón S, Kay S, Tomé M, Burgess PJ, Giannitsopoulos M, Palma JH (2020) Quantifying regulating ecosystem services with increased tree densities on European Farmland. Sustainability 12(16):6676. https://doi.org/10.3390/su12166676

Defra, SEERAD, DARD, DEPC (2007) Agriculture in the United Kingdom 2006. The Stationery Office, London

Defra, Department for Environment, Food and Rural Affairs (2021). Agriculture in the United Kingdom 2021. https://www.gov.uk/government/statistics/agriculture-in-the-united-kingdom-2021. Accessed 30 June 2022

Dijkstra J, Bannink A, France J, Kebreab E, van Gastelen S (2018) Short communication: anti-methanogenic effects of 3-nitrooxypropanol depend on supplementation dose, dietary fiber content, and cattle type. J. Dairy Sci 101:9041–9047. https://doi.org/10.3168/jds.2018-14456

Dixon SD, Qassim SM, Rowson JG, Worrall F, Evans MG, Boothroyd IM, Bonn A (2014) Restoration effects on water table depths and CO_2 fluxes from climatically marginal blanket bog. Biogeochemistry 118:159–176. https://doi.org/10.1007/s10533-013-9915-4

Dumont B, Fortun-Lamothe L, Jouven M, Thomas M, Tichit M (2013) Prospects from agroecology and industrial ecology for animal production in the 21st century. Animal 7:1028–1043. https://doi.org/10.1017/S1751731112002418

Dunn C, Burden A, Chamberlain B, Danek S, Evans C, Freeman C, Harvey R, Proctor S, Walker J (2021) Nature-based solutions for climate change in the UK-Peatlands Nature-based solutions for climate change in the UK. https://research.bangor.ac.uk/portal/files/38669482/NbS_Report_Final_Designed.pdf. Accessed 30 June 2022

Duthie CA, Troy SM, Hyslop JJ, Ross DW, Roehe R, Rooke JA (2018) The effect of dietary addition of nitrate or increase in lipid concentrations, alone or in combination, on performance and methane emissions of beef cattle. Animal 12:280–287. https://doi.org/10.1017/S175173111700146X

Englund O, Mola-Yudego B, Börjesson P, Cederberg C, Dimitriou I, Scarlat N, Berndes G (2023) Large-scale deployment of grass in crop rotations as a multifunctional climate mitigation strategy. GCB Bioenerg 15(2):166–84. https://doi.org/10.1111/gcbb.13015

Eory V, Pellerin S, Garcia GC, Lehtonen H, Licite I, Mattila H, Lund-Sørensen T, Muldowney J, Popluga D, Strandmark L, Schulte R (2018) Marginal abatement cost curves for agricultural climate policy: state-of-the art, lessons learnt and future potential. J Clean Prod 182:705–716. https://doi.org/10.1016/j.jclepro.2018.01.252

Eory V, Maire J, MacLeod M, Sykes A, Barnes A, Rees RM, Topp CFE, Wall, E (2020) Non-CO_2 abatement in the UK agricultural sector by 2050. https://pure.sruc.ac.uk/en/publications/non-co2-abatement-in-the-uk-agricultural-sector-by-2050-summary-r. Accessed 12 June 2022

Evans CD, Peacock M, Baird AJ, Artz RR, Burden A, Callaghan N, Chapman PJ, Cooper HM, Coyle M, Craig E, Cumming A (2021) Overriding water table control on managed peatland greenhouse gas emissions. Nature 593(7860):548–52. https://doi.org/10.1038/s41586-021-03523-1

FAO (2017) Global Livestock Environmental Assessment Model (GLEAM) http://www.fao.org/gleam/results/en/. Accessed 22 June 2022

Felten D, Fröba N, Fries J, Emmerling C (2013) Energy balances and greenhouse gas-mitigation potentials of bioenergy cropping systems (Miscanthus, rapeseed, and maize) based on farming conditions in Western Germany. Renew Energ 55:160–74. https://doi.org/10.1016/j.renene.2012.12.004

Firkins JL, Eastridge ML (1994) Assessment of the effects of iodine value on fatty acid digestibility, feed intake and milk production. J Dairy Sci 77:2357–2366. https://doi.org/10.3168/jds.S0022-0302(94)77178-2

Foster ID, Collins AL, Naden PS, Sear DA, Jones JI, Zhang Y (2011) The potential for paleolimnology to determine historic sediment delivery to rivers. J Paleolimnol 45:287–306. https://doi.org/10.1007/s10933-011-9498-9

Fu Q, Clark IM, Zhu J, Hu H, Hirsch PR (2018) The short-term effects of nitrification inhibitors on the abundance and expression of ammonia and nitrite oxidizers in a long-term field experiment comparing land management. Biol Fertil Soils 54:163–172. https://doi.org/10.1007/s00374-017-1249-2

Fuchs K, Merbold L, Buchmann N, Bellocchi G, Bindi M, Brilli L, Conant RT, Dorich CD, Ehrhardt F, Fitton N, Grace P (2020) Evaluating the potential of legumes to mitigate N_2O emissions from permanent grassland using process-based models. Global Bio Geochem Cycles 34(12):e2020GB006561. https://doi.org/10.1029/2020GB006561

Fujimori S, Hasegawa T, Krey V, Riahi K, Bertram C, Bodirsky BL, Bosetti V, Callen J, Després J, Doelman J, Drouet LA (2019) A multi-model assessment of food security implications of climate change mitigation. Nat Sustain 2(5):386–396. https://doi.org/10.1038/s41893-019-0286-2

FWI (2017) Farm income taxes. https://www.fwi.co.uk/business/guide-to-taxation. Accessed 28 June 2022

Gill M, Garnsworthy PC, Wilkinson JM (2021) More effective linkages between science and policy are needed to minimize the negative environmental impacts of livestock production. Animal 100291. https://doi.org/10.1016/j.animal.2021.100291

Glenk K, Shrestha S, Topp CF, Sánchez B, Iglesias A, Dibari C, Merante P (2017) A farm level approach to explore farm gross margin effects of soil organic carbon management. Agric Syst 151:33–46. https://doi.org/10.1016/j.agsy.2016.11.002

Glithero NJ, Wilson P, Ramsden SJ (2013) Prospects for arable farm uptake of short rotation coppice willow and miscanthus in England. Appl Energ 107:209–218. https://doi.org/10.1016/j.apenergy.2013.02.032

Goel G, Makkar H (2012) Methane mitigation from ruminants using tannins and saponins. Trop Anim Health Prod 44:729–39. https://doi.org/10.3390/ani10091531

Gonzalez-Mejia A, Styles D, Wilson P, Gibbons J (2018) Metrics and methods for characterizing dairy farm intensification using farm survey data. PLoS One 13:e0195286. https://doi.org/10.1371/journal.pone.0195286

Gordon IJ, Prins HHT (2008) Introduction: grazers and browsers in a changing world. In: Gordon IJ, Prins HHT (eds) The Ecology of Browsing and Grazing, Ecological Studies. ISBN : 978-3-540-72421-6

Gregory AS, Dungait JA, Shield IF, Macalpine WJ, Cunniff J, Durenkamp M, White RP, Joynes A, Richter GM (2018) Species and genotype effects of bioenergy crops on root production, carbon and nitrogen in temperate agricultural soil. Bioenergy Res 11:382–397. https://doi.org/10.1007/s12155-018-9903-6

Gross CD, Bork EW, Carlyle CN, Chang SX (2022) Biochar and its manure-based feedstock have divergent effects on soil organic carbon and greenhouse gas emissions in croplands. Sci Total Environ. 806:151337. https://doi.org/10.1016/j.scitotenv.2021.151337

Haddaway NR, Macura B, Whaley P, Pullin AS (2018) ROSES Reporting standards for systematic evidence syntheses: pro forma, flow-diagram and descriptive summary of the plan and conduct of environmental systematic reviews and systematic maps. Environ Evid 7:1–8. https://doi.org/10.1186/s13750-018-0121-7

Hargreaves PR, Baker KL, Graceson A, Bonnett SA, Ball BC, Cloy JM (2021) Use of a nitrification inhibitor reduces nitrous oxide (N_2O) emissions from compacted grassland with different soil textures and climatic conditions. Agric Ecosyst Environ 310:107307. https://doi.org/10.1016/j.agee.2021.107307

Harris C, Ratnieks FLW (2022) Clover in agriculture: combined benefits for bees, environment, and farmer. J Insect Conserv 26:339–357. https://doi.org/10.1007/s10841-021-00358-z

Harris ZM, Alberti G, Viger M, Jenkins JR, Rowe R, McNamara NP, Taylor G (2017) Land-use change to bioenergy: grassland to short rotation coppice willow has an improved carbon balance. GCB Bioenerg 9:469–484. https://doi.org/10.1111/gcbb.12347

ISO (2006) ISO 14040: Environmental management -- life cycle assessment -- principles and framework. Geneva: International Organization for Standardization. https://www.iso.org/standard/37456.html. Accessed 30 June 2022

Jebari A, Pereyra-Goday F, Kumar A, Collins A, McAuliffe G (2023) Dataset on agricultural greenhouse gas mitigation measures in the UK, Mendeley Data, V1. https://doi.org/10.17632/t9kynfj5jf.1

Jeswani HK, Whiting A, Martin A, Azapagic A (2019) Environmental and economic sustainability of poultry litter gasification for electricity and heat generation. Waste Manage 95:182–191. https://doi.org/10.1016/j.wasman.2019.05.053

Jordon MW, Willis KJ, Harvey WJ, Petrokofsky L, Petrokofsky G (2020) Implications of temperate agroforestry on sheep and cattle productivity, environmental impacts and enterprise economics. A systematic evidence map. Forests 11:1–29. https://doi.org/10.3390/f11121321

Jordon MW, Smith P, Long PR, Bürkner PC, Petrokofsky G, Willis KJ (2022) Can Regenerative Agriculture increase national soil carbon stocks? Simulated country-scale adoption of reduced tillage, cover cropping, and ley-arable integration using RothC. Sci Total Environ 825:153955. https://doi.org/10.1016/j.scitotenv.2022.153955

Kataki S, Hazarika S, Baruah DC (2017) Assessment of by-products of bioenergy systems (anaerobic digestion and gasification) as potential crop nutrient. Waste Manage 59:102–117. https://doi.org/10.1016/j.wasman.2016.10.018

Kay S, Rega C, Moreno G, den Herder M, Palma JH, Borek R, Crous-Duran J, Freese D, Giannitsopoulos M, Graves A, Jäger M (2019) Agroforestry creates carbon sinks whilst enhancing the environment in agricultural landscapes in Europe. Land Use Policy 83:581–593. https://doi.org/10.1016/j.landusepol.2019.02.025

Kebreab E, Bannink A, Pressman EM, Walker N, Karagiannis A, van Gastelen S, Dijkstra J (2023) A meta-analysis of effects of 3-nitrooxypropanol on methane production, yield, and intensity in dairy cattle. J Dairy Sci 106(2):927–36. https://doi.org/10.3168/jds.2022-22211

Kelland ME, Wade PW, Lewis AL, Taylor LL, Sarkar B, Andrews MG, Lomas MR, Cotton TA, Kemp SJ, James RH, Pearce CR (2020) Increased yield and CO_2 sequestration potential with the C4 cereal Sorghum bicolor cultivated in basaltic rock dust-amended agricultural soil. Glob Chang Biol 26:3658–3676. https://doi.org/10.1111/gcb.15089

Kemp P, Sear D, Collins A, Naden P, Jones I (2011) The impacts of fine sediment on riverine fish. Hydrol Process 25:1800–1821. https://doi.org/10.1002/hyp.7940

Kliem KE, Humphries DJ, Kirton P, Givens DI, Reynolds CK (2019) Differential effects of oilseed supplements on methane production and milk fatty acid concentrations in dairy cows. Animal 13:309–317. https://doi.org/10.1017/S1751731118001398

Knook J, Eory V, Brander M, Moran D (2020) The evaluation of a participatory extension programme focused on climate friendly farming. J Rural Stud 76:40–48. https://doi.org/10.1016/j.jrurstud.2020.03.010

Kohl C, McIntosh EJ, Unger S, Haddaway NR, Kecke S, Schiemann J, Wilhelm R (2018) Online tools supporting the conduct and reporting of systematic reviews and systematic maps: a case study on CADIMA and review of existing tools. Environ Evid 7:1–17. https://doi.org/10.1186/s13750-018-0115-5

Lee C, Beauchemin KA (2014) A review of feeding supplementary nitrate to ruminant animals: nitrate toxicity, methane emissions, and production performance. Can J Anim Sci 94:557–570. https://doi.org/10.4141/cjas-2014-069

Lefebvre D, Goglio P, Williams A, Manning DA, de Azevedo AC, Bergmann M, Meersmans J, Smith P (2019) Assessing the potential of soil carbonation and enhanced weathering through life cycle assessment: a case study for Sao Paulo State, Brazil. J Clean Prod 233:468–481. https://doi.org/10.1016/j.jclepro.2019.06.099

Leinonen I, Williams AG, Kyriazakis I (2014) The effects of welfare-enhancing system changes on the environmental impacts of broiler and egg production. Poultry Sci 93:256–266. https://doi.org/10.3382/ps.2013-03252

Leinonen I, MacLeod M, Bell J (2018) Effects of alternative uses of distillery by-products on the greenhouse gas emissions of Scottish malt whisky production: a system expansion approach. Sustainability (Switzerland) 10(5):1473. https://doi.org/10.3390/su10051473

Li Z, Zhang Q, Li Z, Qiao Y, Du K, Yue Z, Tian C, Leng P, Cheng H, Chen G, Li F (2023) Responses of soil greenhouse gas emissions to no-tillage: a global meta-analysis. Sustain Prod Consump 36:479–92. https://doi.org/10.1016/j.spc.2023.02.003

Llonch P, Haskell MJ, Dewhurst RJ, Turner SP (2017) Current available strategies to mitigate greenhouse gas emissions in livestock systems: an animal welfare perspective. Animal 11:274–284. https://doi.org/10.1017/S1751731116001440

Lüscher A, Mueller-Harvey I, Soussana JF, Rees RM, Peyraud JL (2014) Potential of legume-based grassland-livestock systems in Europe: a review. Grass Forage Sci 69:206–228. https://doi.org/10.1111/gfs.12124

MacLeod M, Moran D, Eory V, Rees RM, Barnes A, Topp CF, Ball B, Hoad S, Wall E, McVittie A, Pajot G (2010) Developing greenhouse gas marginal abatement cost curves for agricultural emissions from crops and soils in the UK. Agris Syst 103(4):198–209. https://doi.org/10.1016/j.agsy.2010.01.002

MacLeod M, Moran D (2017) Integrating livestock health measures into marginal abatement cost curves. Rev Sci Tech-Off Int Épizooties 36:97–104. https://doi.org/10.20506/rst.36.1.2613

Malhi SS, Soldberg ED, Nyborg M (2005) Influence of formulation of elemental S fertilizer on yield, quality and S uptake of canola seed. Can J Plant Sci 85:793–802. https://doi.org/10.4141/P04-134

Matysek M, Leake J, Banwart S, Johnson I, Page S, Kaduk J, Smalley A, Cumming A, Zona D (2022) Optimizing fen peatland water-table depth for romaine lettuce growth to reduce peat wastage under future climate warming. Soil Use Manag 38:341–354. https://doi.org/10.1111/sum.12729

McAuliffe GA, Chapman DV, Sage CL (2016) A thematic review of life cycle assessment (LCA) applied to pig production. Environ Impact Assess 56:12–22. https://doi.org/10.1016/j.eiar.2015.08.008

McAuliffe GA, Takahashi T, Mogensen L, Hermansen JE, Sage CL, Chapman DV, Lee MR (2017) Environmental trade-offs of pig production systems under varied operational efficiencies. J Clean Prod 165:1163–1173. https://doi.org/10.1016/j.agee.2020.106978

McAuliffe GA, Takahashi T, Orr RJ, Harris P, Lee MR (2018) Distributions of emissions intensity for individual beef cattle reared on pasture-based production systems. J Clean Prod 171:1672–1680. https://doi.org/10.1016/j.jclepro.2017.10.113

Mcauliffe GA, Takahashi T, Lee MRF (2020a) Applications of nutritional functional units in commodity-level life cycle assessment (LCA) of agri-food systems. Int J Life Cycle Ass 25:208–221. https://doi.org/10.1007/s11367-019-01679-7

McAuliffe GA, López-Aizpún M, Blackwell MS, Castellano-Hinojosa A, Darch T, Evans J, Horrocks C, Le Cocq K, Takahashi T, Harris P, Lee MR (2020b) Elucidating three-way interactions between soil, pasture and animals that regulate nitrous oxide emissions from temperate grazing systems. Agric Ecosyst

Environ 300:106978. https://doi.org/10.1016/j.agee.2020.106978
McCalmont JP, Hastings A, McNamara NP, Richter GM, Robson P, Donnison IS, Clifton-Brown J (2017) Environmental costs and benefits of growing Miscanthus for bioenergy in the UK. GCB Bioenerg 9:489–507. https://doi.org/10.1111/gcbb.12294
McIlroy JP, McGeough KL, Laughlin RJ, Carolan R (2019) Abatement of ammonia emissions from dairy cow house concrete floor surfaces through additive application. Biosyst Eng 188:320–330. https://doi.org/10.1016/j.biosystemseng.2019.10.016
Mee JF, Boyle LA (2020) Assessing whether dairy cow welfare is "better" in pasture-based than in confinement-based management systems. New Zeal Vet J 68:168–177. https://doi.org/10.1080/00480169.2020.1721034
Melgar A, Welter KC, Nedelkov K, Martins CM, Harper MT, Oh J, Räisänen SE, Chen X, Cueva SF, Duval S, Hristov AN (2020) Dose-response effect of 3-nitrooxypropanol on enteric methane emissions in dairy cows. J Dairy Sci 103:6145–6156. https://doi.org/10.3168/jds.2019-17840
Montes F, Meinen R, Dell C, Rotz A, Hristov AN, Waghorn JOhG, Gerber PJ, Henderson B, Makkar HPS, Dijkstra J (2013) Mitigation of methane and nitrous oxide emissions from animal operations: II. A review of manure management mitigation options. J Anim Sci 91:5070–5094. https://doi.org/10.2527/jas.2013-6584
Morgan-Davies C, Kyle J, Boman IA, Wishart H, McLaren A, Fair S, Creighton P (2021) A comparison of farm labour, profitability, and carbon footprint of different management strategies in Northern European grassland sheep systems. Agric Syst 191:103155. https://doi.org/10.1016/j.agsy.2021.103155
Müller LJ, Kätelhön A, Bachmann M, Zimmermann A, Sternberg A, Bardow A (2020) A guideline for life cycle assessment of carbon capture and utilization. Front Energy Res 8:15. https://doi.org/10.3389/fenrg.2020.00015
Musarika S, Atherton CE, Gomersall T, Wells MJ, Kaduk J, Cumming AM, Page SE, Oechel WC, Zona D (2017) Effect of water table management and elevated CO_2 on radish productivity and on CH_4 and CO_2 fluxes from peatlands converted to agriculture. Sci Total Environ 584–585:665–672. https://doi.org/10.1016/j.scitotenv.2017.01.094
NFU (2021a) NFU's net zero strategic advisory board. https://www.nfuonline.com/archive?treeid=145375. Accessed 30 June 2022
NFU (2021b) British food leading the way. https://www.nfuonline.com/media/s4xluxgg/british-food-leading-the-way.pdf. Accessed 30 June 2022
Nicholson F, Bhogal A, Cardenas L, Chadwick D, Misselbrook T, Rollett A, Taylor M, Thorman R, Williams J (2017) Nitrogen losses to the environment following food-based digestate and compost applications to agricultural land Environmental Pollution Nitrogen losses to the environment following food-based digestate and 1 compost applications to agricultural land. Environ Pollut 228:504–516. https://doi.org/10.1016/j.envpol.2017.05.023
Nicholson F, Bhogal A, Taylor M, McGrath S, Withers P (2018) Long-term effects of biosolids on soil quality and fertility. Soil Sci 183(3):89–98. https://doi.org/10.1097/SS.0000000000000239
Nworji J (2017) Physical and bioeconomic analysis of ecosystem services from a silvopasture system. Dissertation, Bangor University
O'Neill C, Lim FKS, Edwards DP, Osborne CP (2020) Forest regeneration on European sheep pasture is an economically viable climate change mitigation strategy. Environ Res Lett 15(10):104090. https://doi.org/10.1088/1748-9326/abaf87
OECD (2004) SIDS Initial Assessment Report for Cyanoguanidine CAS No. 461-58-5, p. 75. Microsoft Word - 461585.doc (oecd.org). Accessed 25 June 2022
Ouzzani M, Hammady H, Fedorowicz Z, Elmagarmid A (2016) Rayyan — a web and mobile app for systematic reviews. Syst Rev 5:210. https://doi.org/10.1186/s13643-016-0384-4
Pagella T, Whistance L (2019) Silvopasture. In: Raskin B, Osborn S (eds) The agroforestry handbook: agroforestry for the UK, 1 edn. Soil Association Limited, pp 45-59. https://www.soilassociation.org/media/19141/the-agroforestry-handbook.pdf Accessed 25 June 2022
Pal P, McMillan AM, Saggar S (2016) Pathways of dicyandiamide uptake in pasture plants: a laboratory study. Biol Fert Soils 52:539–546. https://doi.org/10.1007/s00374-016-1096-6
Pérez-Barbería FJ, Mayes RW, Giráldez J, Sánchez-Pérez D (2020) Ericaceous species reduce methane emissions in sheep and red deer: respiration chamber measurements and predictions at the scale of European heathlands. Sci Total Environ 714:136738. https://doi.org/10.1016/j.scitotenv.2020.136738
Poulton P, Johnston J, Macdonald A, White R, Powlson D (2018) Major limitations to achieving "4 per 1000" increases in soil organic carbon stock in temperate regions: evidence from long-term experiments at Rothamsted Research, United Kingdom. Glob Chang Biol 24:2563–2584. https://doi.org/10.1111/gcb.14066
Powlson DS, Dawson CJ (2022) Use of ammonium sulphate as a sulphur fertilizer: implications for ammonia volatilization. Soil Use Manag 38:622–634. https://doi.org/10.1111/sum.12733
Prathap P, Chauhan SS, Leury BJ, Cottrell JJ, Dunshea FR (2021) Towards sustainable livestock production: estimation of methane emissions and dietary interventions for mitigation. Sustainability (Switzerland) 13(11):6081. https://doi.org/10.3390/su13116081
Purakayastha TJ, Bera T, Bhaduri D, Sarkar B, Mandal S, Wade P, Kumari S, Biswas S, Menon M, Pathak H, Tsang DC (2019) A review on biochar modulated soil condition improvements and nutrient dynamics concerning crop yields: pathways to climate change mitigation and global food security. Chemosphere 227:345–365. https://doi.org/10.1016/j.chemosphere.2019.03.170
Ramos-Morales E, Rossi G, Cattin M, Jones E, Braganca R, Newbold CJ (2018) The effect of an isoflavonid-rich liquorice extract on fermentation, methanogenesis and the microbiome in the rumen simulation technique. FEMS Microbiol Ecol 94(3):fiy009. https://doi.org/10.1093/femsec/fiy009
Rial-Lovera K, Davies WP, Cannon ND (2017) Implications of climate change predictions for UK cropping and prospects for possible mitigation: a review of challenges and potential responses. J Sci Food Agric 97:17–32. https://doi.org/10.1002/jsfa.7767
Rice P, O'Brien D, Shalloo L (2017) Evaluation of allocation methods for calculation of carbon footprint of grass-based dairy production. J Environ Manag 202:311–319. https://doi.org/10.1016/j.jenvman.2017.06.071
Rigby H, Clarke BO, Pritchard DL (2016) A critical review of nitrogen mineralization in biosolids-amended soil, the associated fertilizer value for crop production and potential for emissions to the environment. Sci Total Environ 541:1310–1338. https://doi.org/10.1016/j.scitotenv.2015.08.089
Rivero MJ, Lee MRF (2022) A perspective on animal welfare of grazing ruminants and its relationship with sustainability. Anim Prod Sci 62:1739–1748. https://doi.org/10.1071/AN21516
Robertson AD, Davies CA, Smith P, Stott Andy W, Clark Emily L, McNamara Niall P (2017) Carbon Inputs from Miscanthus Displace Older Soil Organic Carbon Without Inducing Priming. Bioenerg Res 10:86–101. https://doi.org/10.1007/s12155-016-9788-1
Rosa L, Rulli MC, Ali S (2021) Energy implications of the 21st century agrarian transition. Nat Commun 12. https://doi.org/10.1038/s41467-021-22581-7
Rudolph G, Hörtenhuber S, Bochicchio D, Butler G, Brandhofer R, Dippel S, Dourmad JY, Edwards S, Früh B, Meier M, Prunier A (2018) Effect of three husbandry systems on environmental

impact of organic pigs. Sustainability 10:3796. https://doi.org/10.3390/su10103796
Sánchez-Rodríguez AR, Carswell AM, Shaw R, Hunt J, Saunders K, Cotton J, Chadwick DR, Jones DL, Misselbrook TH (2018) Advanced processing of food waste based digestate for mitigating nitrogen losses in a winter wheat crop. Front Sustain Food Syst 2:35. https://doi.org/10.3389/fsufs.2018.00035
Sándor R, Ehrhardt F, Brilli L, Carozzi M, Recous S, Smith P, Snow V, Soussana JF, Dorich CD, Fuchs K, Fitton N (2018) The use of biogeochemical models to evaluate mitigation of greenhouse gas emissions from managed grasslands. Sci Total Environ 642:292–306. https://doi.org/10.1016/j.scitotenv.2018.06.020
Scott A, Blanchard R (2021) The role of anaerobic digestion in reducing dairy farm greenhouse gas emissions. Sustainability 13:1–18. https://doi.org/10.3390/su13052612
Shah SHH, Li Y, Wang J, Collins AL (2020) Optimizing farmyard manure and cattle slurry applications for intensively managed grasslands based on UK-DNDC model simulations. Sci Total Environ 714:136672. https://doi.org/10.1016/j.scitotenv.2020.136672
Smith P (2014) Do grasslands act as a perpetual sink for carbon? Glob Change Biol 20:2708–2711. https://doi.org/10.1111/gcb.12561
Smith P, Adams J, Beerling DJ, Beringer T, Calvin KV, Fuss S, Griscom B, Hagemann N, Kammann C, Kraxner F, Minx JC, Minx Popp A, Renforth P, Vicente Vicente JL, Keesstra S (2019) Impacts of land-based greenhouse gas removal options on ecosystem services and the United Nations sustainable development goals. Annu Rev Env Resour 44(1):1–32. https://doi.org/10.1146/annurev-environ-101718-033129
Smith P, Reay D, Smith J (2021) Agricultural methane emissions and the potential for mitigation. Phil Trans R Soc A 379:20200451. https://doi.org/10.1098/rsta.2020.0451
Snelling TJ, Auffret MD, Duthie CA, Stewart RD, Watson M, Dewhurst RJ, Roehe R, Walker AW (2019) Temporal stability of the rumen microbiota in beef cattle, and response to diet and supplements. Anim Microbiome 1:1–14. https://doi.org/10.1186/s42523-019-0018-y
Soteriades AD, Foskolos A, Styles D, Gibbons JM (2019) Diversification not specialization reduces global and local environmental burdens from livestock production. Environ Int: 132:104837. https://doi.org/10.1016/j.envint.2019.05.031
Soteriades AD, Gonzalez-Mejia AM, Styles D, Foskolos A, Moorby JM, Gibbons JM (2018) Effects of high-sugar grasses and improved manure management on the environmental footprint of milk production at the farm level. J Clean Prod 202:1241–1252. https://doi.org/10.1016/j.jclepro.2018.08.206
Stanchev P, Vasilaki V, Egas D, Colon J, Ponsá S, Katsou E (2020) Multilevel environmental assessment of the anaerobic treatment of dairy processing effluents in the context of circular economy. J Clean Prod 261:121139. https://doi.org/10.1016/j.jclepro.2020.121139
Styles D, Gonzalez-Mejia A, Moorby J, Foskolos A, Gibbons J (2018) Climate mitigation by dairy intensification depends on intensive use of spared grassland. Glob Chang Biol 24:681–693. https://doi.org/10.1111/gcb.13868
Sun B, Hallett PD, Caul S (2011) Distribution of soil carbon and microbial biomass in arable soils different tillage regimes. Plant Soil 338:17–25. https://doi.org/10.1007/s11104-010-0459-2
Taft HE, Cross PA, Jones DL (2018) Efficacy of mitigation measures for reducing greenhouse gas emissions from intensively cultivated peatlands. Soil Biol Biochem 127:10–21. https://doi.org/10.1016/j.soilbio.2018.08.020
Tang K, Wang M, Zhou D (2021) Abatement potential and cost of agricultural greenhouse gases in Australian dryland farming system. Environ Sci Pollut R 28:21862–73. https://doi.org/10.1007/s11356-020-11867-w
Tilman D, Clark M (2014) Global diets link environmental sustainability and human health. Nature 515:518–522. https://doi.org/10.1038/nature13959
Tilman D, Balzer C, Hill J (2011) Global food demand and the sustainable intensification of agriculture. Proc Natl Acad Sci 108:20260–20264. https://doi.org/10.1073/pnas.1116437108
UNEP United Nations Environment Programme and Climate and Clean Air Coalition (2021) Global methane assessment: benefits and costs of mitigating methane emissions. Nairobi: United Nations Environment Programme. https://www.unep.org/resources/report/global-methane-assessment-benefits-and-costs-mitigating-methane-emissions. Accessed 27 June 2022
Van Den Berge S, Vangansbeke P, Baeten L, Vanneste T, Vos F, Verheyen K (2021) Soil carbon of hedgerows and 'ghost' hedgerows. Agroforest Syst 95(6):1087–103. https://doi.org/10.1007/s10457-021-00634-6
de Vries M, de Boer IJM (2010) Comparing environmental impacts for livestock products: A review of life cycle assessments. Livest Sci 128:1–11. https://doi.org/10.1016/j.livsci.2009.11.007
de Vries M, van Middelaar CE, de Boer IJM (2015) Comparing environmental impacts of beef production systems: A review of life cycle assessments. Livest Sci 178:279–288. https://doi.org/10.1016/j.livsci.2015.06.020
Wallace RJ, Sasson G, Garnsworthy PC, Tapio I, Gregson E, Bani P, Huhtanen P, Bayat AR, Strozzi F, Biscarini F, Snelling TJ, Saunders N, Potterton SL, Craigon J, Minuti A, Trevisi E, Callegari ML, Cappelli FP, Cabezas-Garcia EH, Vilkki J, Pinares-Patino C, Fliegerová KO, Mrázek J, Sechovcová H, Kopecný J, Bonin A, Boyer F, Taberlet P, Kokou F, Halperin E, Williams JL, Shingfield KJ, Mizrahi I (2019) A heritable subset of the core rumen microbiome dictates dairy cow productivity and emissions. Sci Adv 5:eaav8391. https://doi.org/10.1126/sciadv.aav8391
Walsh JJ, Jones DL, Chadwick DR, Williams AP (2018) Repeated application of anaerobic digestate, undigested cattle slurry and inorganic fertilizer N: impacts on pasture yield and quality. Grass Forage Sci 73:758–763. https://doi.org/10.1111/gfs.12354
Wang J (2014) Decentralized biogas technology of anaerobic digestion and farm ecosystem: opportunities and challenges. Front Energy 2:10. https://doi.org/10.1016/j.eng.2018.05.007
Warner D, Tzilivakis J, Green A, Lewis K (2017) Prioritising agri-environment options for greenhouse gas mitigation. Int J Clim Chang Strateg Manag 9:104–122. https://doi.org/10.1016/bs.aecr.2020.08.004
Water UK (2010) Recycling of biosolids to agricultural land. Issue Number 3, January 2010. Water UK. Queen Anne's Gate, London, pp. 1. https://assuredbiosolids.co.uk/wp-content/uploads/2018/05/Recycling-Biosolids-to-Agricultural-Land.pdf. Accessed 28 June 2022
Webb J (2017) Nitrous oxide and methane emissions from agriculture and approaches to mitigate greenhouse gas emissions from livestock production. University of Wolverhampton. https://wlv.openrepository.com/handle/2436/621013. Accessed 30 June 2022
Welten BG, Ledgard SF, Luo J (2014) Administration of dicyandiamide to dairy cows via drinking water reduces nitrogen losses from grazed pastures. Agr Sci 152:150–158. https://doi.org/10.1017/S0021859613000634
Wiik E, Toberman H, Ford H, Webb B, Healey J, Pagella T, Marley C, Smith A (2019) Science to policy: impacts of trees on farm ecosystem services. Ser Cymru National Research Network for Low Carbon, Energy and Environment, Bangor University. https://research.bangor.ac.uk/portal/files/22766912/NRN_LCEE_SciencetoPolicy_MultilandF1WEB.pdf. Accessed 21 June 2022
Wilkinson JM, Garnsworthy PC (2017) Dietary options to reduce the environmental impact of milk production. J Agr Sci 155:334–347. https://doi.org/10.1017/S0021859616000757

Wilkinson JM, Chamberlain AT, Rivero MJ (2021) The case for grazing dairy cows. Agron 11:1–12. https://doi.org/10.3390/agronomy11122466

Wonfor R (2017) The effect of alternative bedding materials on sheep behaviour and welfare. Farming Connect, 2(3): 235–246. https://businesswales.gov.wales/farmingconnect/news-and-events/technical-articles/effect-alternative-bedding-materials-sheep-behaviour-and-welfare. Accessed 02 June 2022

Yesufu J, McCalmont JP, Clifton-Brown JC, Williams P, Hyland J, Gibbons J, Styles D (2020) Consequential life cycle assessment of miscanthus livestock bedding, diverting straw to bioelectricity generation. GCB Bioenergy 12:39–53. https://doi.org/10.1111/gcbb.12646

Zheng YH, Li ZF, Feng SF (2010) Biomass energy utilization in rural areas may contribute to alleviating energy crisis and global warming: a case study in a typical agro-village of Shandong, China. Renew Sust Energ Rev 14:3132–3139. https://doi.org/10.1016/j.rser.2010.07.052

8.3. Anexo 3. A commentary on key methodological developments related to nutritional life cycle assessment (nLCA) generated throughout a 6-year strategic scientific programme

G. A. McAuliffe[1] T. Takahashi[1,2] M. R. F. Lee[3] A. Jebari[1] L. Cardenas[1]

A. Kumar[1] F. Pereyra Goday[4] H. Scalabrino[5] A. L. Collins[1]

[1]Net Zero and Resilient Farming, Rothamsted Research, Okehampton, UK

[2]University of Bristol, Bristol Veterinary School, Langford, Somerset, UK

[3]Harper Adams University, School of Sustainable Food and Farming, Newport, Shropshire, UK

[4]Instituto Nacional de Investigación Agropecuaria (INIA), Treinta y Tres, Uruguay

[5]University of Normandie, ESIX Normandie Agri-food Department, Caen, France

Received: 1 March 2023 | Accepted: 31 May 2023
DOI: 10.1002/fes3.480

COMMENTARY

Food and Energy Security WILEY

A commentary on key methodological developments related to nutritional life cycle assessment (nLCA) generated throughout a 6-year strategic scientific programme

G. A. McAuliffe[1] | T. Takahashi[1,2] | M. R. F. Lee[3] | A. Jebari[1] | L. Cardenas[1] | A. Kumar[1] | F. Pereyra-Goday[4] | H. Scalabrino[5] | A. L. Collins[1]

[1]Net Zero and Resilient Farming, Rothamsted Research, Okehampton, UK
[2]University of Bristol, Bristol Veterinary School, Langford, Somerset, UK
[3]Harper Adams University, School of Sustainable Food and Farming, Newport, Shropshire, UK
[4]Instituto Nacional de Investigación Agropecuaria (INIA), Treinta y Tres, Uruguay
[5]University of Normandie, ESIX Normandie Agri-food Department, Caen, France

Correspondence
G. A. McAuliffe, Net Zero and Resilient Farming, Rothamsted Research, Okehampton, UK.
Email: gm.scienceprofile@gmail.com

Funding information
Biotechnology and Biological Sciences (BBSRC) via a Scientific Initiative Catalyst Fund (SICA); UK Government's Department for Environment, Food, and Rural Affairs 'LCA Food Basket Project'; Soil to Nutrition (S2N) - BBSRC, Grant/Award Number: BBS/E/C/000I0320 and BBS/E/C/000I0330; Doctoral Fellowship under National Agency of Research, Grant/Award Number: MOV_CA_2021_1_171482

Abstract

Rothamsted Research (RRes) is the world's oldest agricultural research centre, notable for the development of the first synthetic fertilizer (superphosphate) and long-term farming experiments (LTEs) spanning over 170 years. In 2015, RRes recruited several life cycle assessment (LCA) experts and began adopting the method to utilize high resolution agronomical data covering livestock (primarily ruminants), grassland/forage productivity and quality, and arable systems established on its North Wyke Farm Platform (NWFP) and the LTEs. The NWFP is a UK 'National Bioscience Research Infrastructure' (NBRI) developed for informing and testing systems science utilising high-resolution data to determine whether it is possible to produce nutritious food sustainably. Thanks largely to the multidisciplinary knowledge at RRes, and its collaborators, its LCA Team has been at the forefront of methodological advances during a 6-year Institute Strategic Programme (ISP) 'Soil-to-Nutrition' (S2N). While S2N investigated the co-benefits and trade-offs of new mechanistic understanding of efficient nutrient use across scales from pot to landscape, this commentary specifically synthesizes progress in incorporating human nutrition in the context of environmental footprinting, known as 'nutritional LCA' (nLCA). We conclude our commentary with a brief discussion on future pathways of exploration and methodological developments covering various activities along entire agri-food supply-chains.

KEYWORDS
environmental footprints, food systems, net zero, nutritional science, sustainability, synthesis

JEL CLASSIFICATION
Agriculture, Food Security, Food, Biochemistry

1 | INTRODUCTION

Life cycle assessment (LCA) has been used for decades to identify pollution potential 'hotspots' and compare impacts to environmental health arising from various food systems (e.g. de Vries and de Boer, 2010). More recently, however, the LCA method has evolved to consider trade-offs between environmental *and* human health using the 'nutritional-LCA' (nLCA) approach (McAuliffe et al., 2020; McLaren et al., 2021). Rothamsted Research (RRes) is the world's oldest agricultural institute globally famous for its invention of the first commercially synthetic fertilizer (superphosphate) and long-term farming experiments (LTEs), which provide open-access data and information to inform optimal fertilizer rates in relation to various crop yields dating back over 170 years. In 2015, RRes established an LCA team tasked with: (a) utilizing high-resolution (both spatially and temporally) data collected on research platforms at the institute to identify sustainable food systems capable of ensuring food security, and (b) advancing LCA using RRes's interdisciplinary expertise which forms part of the institute's uniqueness; for instance, RRes has in-house modelling capabilities (often informed by high-quality, primary data collected through targeted pot-, plot- and field-scale trials to assess pollutant mitigation measures' feasibility) to estimate farm geospatially heterogeneous farming typologies and interventions to reduce impacts to nature through exploration of interactions between soil health and environmental impacts. This commentary provides a brief synopsis of methodological progression regarding the LCA framework and novel environmental metrics developed as part of a 6-year Institute Strategic Programme—Soil to Nutrition (S2N). S2N's funding comes to an end in March 2023, and therefore, this commentary focusses primarily on developments of novel metrics to explore the nexus between nutritional and environmental sciences, which RRes began its journey in the area through a publication in *Food and Energy Security* (*FES*; McAuliffe et al., 2018), utilising primary (and secondary) data provided directly through S2N experimental research and deep exploration of relevant literature.

2 | NUTRITIONAL DENSITY SCORES

Human nutrient provision is often assessed at the food commodity level in the nutritional sciences using nutritional density scores (NDS), with perhaps the most widely adopted approach, certainly in an LCA context, being the Nutrient Rich Food (NRF9.3; Fulgoni et al., 2009) scoring system which assesses nine encouraged nutrients (i.e. protein, certain minerals and vitamins, and polyunsaturated fatty acids (PUFA)) and three nutrients (saturated fatty acids (SFA), sodium and added sugars) to be limited. The NRF9.3 framework assesses the benefit or risk of each nutrient in a food item against recommended daily intakes (RDI) for the population/geographic region under study. The approach results in a single score for each food, which can be positive or negative. While NRF9.3 is undoubtedly widely used in LCA, it has limitations: for example, unprocessed animal-sourced foods do not contain fibre, a 'nutrient' considered under NRF9.3, making it an imperfect comparison for foods with notably different nutritional profiles (e.g. animal-based produce vs. plant-based produce). McAuliffe et al. (2018) identified this issue and began their nutrition-environment nexus research journey in *FES* by developing a new country-specific framework (UK Nutritional Index; UKNI) to compare animal-based produce (species and production methodology) fairly and transparently. UKNI, inspired by work carried out in Finland (Saarinen et al., 2017), was subsequently used as a scaling factor, known in LCA as a 'functional unit,' to compare the environmental footprints of four meats, thus answering the question: 'how much of a given meat would need to be consumed to meet the RDIs for a range of nutrients and what is the associated environmental footprint?' The results indicated that less beef would need to be consumed to achieve the defined RDIs compared to the three other meats (lamb, chicken meat and pork); this was due primarily to the inclusion of long-chain omega-3 PUFA and zinc, which beef, particularly pasture-produced beef, tends to have higher levels of compared to other meats. While this methodological development was merited at the time, there were limitations to the study such that foods are rarely eaten in isolation and therefore nutritional complementarity at the meal- or diet-level should be explored, as will be discussed in Sections 4 and 5.

3 | COMPLEXITY ASSESSMENT OF NUTRITIONAL LIFE CYCLE ASSESSMENT

McAuliffe et al. (2020) carried out a literature review of nutritional LCA (nLCA) studies and developed a complexity level ranking system under three tiers (Figure 1). Tier 1 was defined as nLCAs, which consider one or multiple nutrients as functional units in isolation. Under Tier 1, protein was found to be the most used nutrient as a functional unit, but issues surrounding digestibility of protein (i.e. the anabolism of amino acids via absorption in the human gut) and quality (i.e. the composition and positioning of amino acids within the proteins quaternary structure) were acknowledged and subsequently addressed as will be described in Section 5. Tier 2, on the other hand,

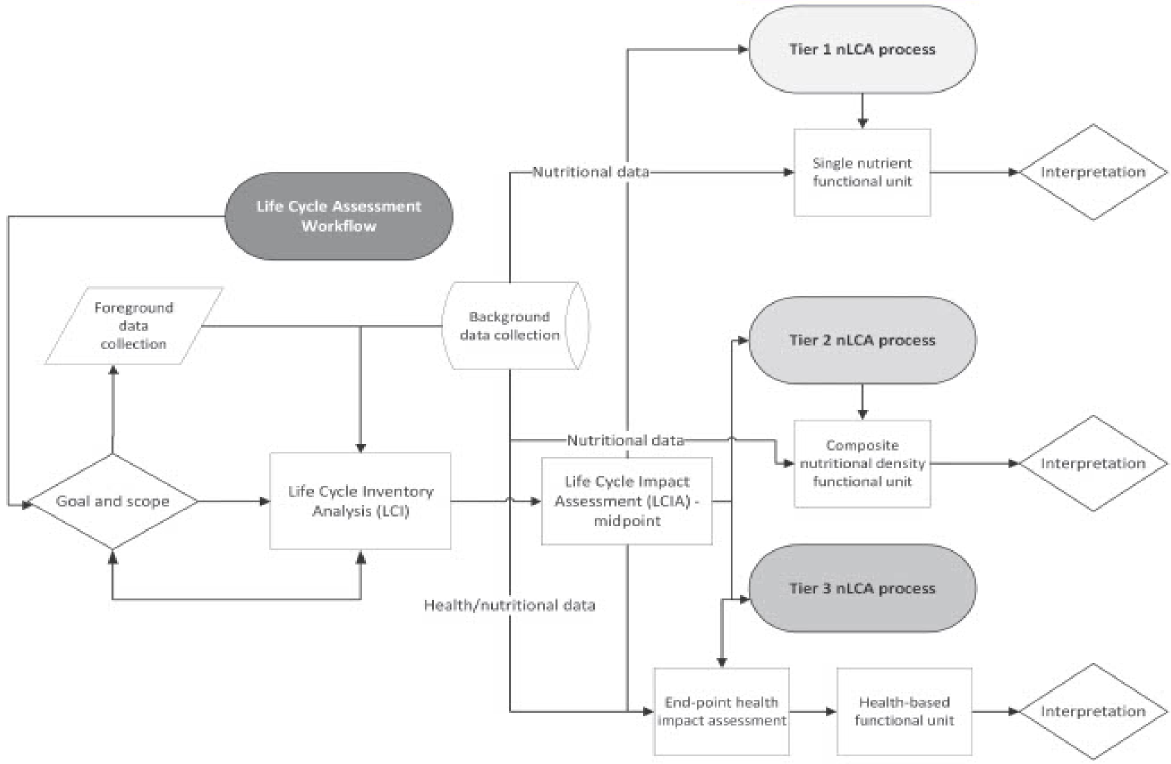


FIGURE 1 Workflow system diagram of how the various tiers of nLCA defined by McAuliffe et al. (2020) are generally conducted.

includes composite scoring systems as functional units (e.g. NDS as in McAuliffe et al., 2018). The second tier is most usually applied to single commodities, and many authors have developed their own NDS scoring systems as per McAuliffe et al. (2018) and explored in more detail in McAuliffe et al. (2020). Finally, Tier 3 typically develops novel 'end-point' impact assessments (i.e. considering in tandem how environmental pollutants *and* nutritional profiles affect a commodity's impact on nature, for example biodiversity losses and gains, and human health, for example using disability-adjusted life years, or DALY). This approach is inevitably the most complex tier under the nLCA framework, but it is worth noting that capturing uncertainties under this approach (and Tier 2, for that matter) is highly complicated and often overlooked, thereby leading to potentially misleading interpretation by consumers, stakeholders and policymakers in the context of food security and human health. RRes's LCA Team and global nutritional scientists are currently working towards highlighting these issues for future nLCA practitioners to be more aware of nutritional complexities while providing solutions to overcome said issues.

4 | GLOBAL EXPERT REPORT ON NLCA BY NUTRITIONAL AND ENVIRONMENTAL SCIENTISTS

In 2021, the United Nations' Food and Agricultural Organization (FAO) commissioned a report to assess state-of-the-art nLCA work holistically and identify strengths, weaknesses and gaps in knowledge which require further research (McLaren et al., 2021). RRes's role in the FAO's global assessment was on data provision, with a specific focus linking environmental footprint databases with nutritional composition databases. In this regard, the data-based element of the report concluded that current combinations of the aforementioned databases lead to disjointed assessments as, for example the temporal and geographic boundaries of such databases may not align. Despite this arguably major limitation, nLCA experts are increasingly working with industry (e.g. farmers, retailers, distributors, etc.) to generate datasets which align environmental footprints with nutritional quantity and quality using primary data (e.g. see Lee et al., 2021 as an example of complexities related to primary data-driven nLCA results from cradle to farmgate), thereby reducing the uncertainty of such analyses. In terms of nutritional *quality*, McLaren et al. (2021) highlighted that comparing nutritional *quantity* of food items is not robust enough to draw clinical conclusions (e.g. at the 'end-point' (n)LCA level); to navigate this restriction, the authors recommended that complexities such as nutrient bioavailability and digestibility, both of which can be affected by 'anti-nutritional factors' such as glucosinolates and tannins, particularly in plant-based produce which restrict the uptake of certain nutrients including protein, should be considered. This is particularly imperative when comparing products which have different nutritional functions (e.g. sources of carbohydrates/energy, fibre, protein and water−/fat- soluble minerals and vitamins).

5 | ASSESSING THE NUTRITIONAL QUALITY OF PROTEIN IN NLCA

As discussed in McAuliffe et al. (2020) and McLaren et al. (2021), protein content of a food tends to be the most commonly used functional unit under Tier 1 nLCA. However, protein anabolism is an incredibly complex process which depends on a balance of 21 proteinaceous amino acids in place and time; nine of which are solely sourced from the diet (indispensable amino acids (IAA) also referred to as essential amino acids) and the others, which although can be assimilated in situ, may become rate limiting. McAuliffe et al. (2023) drew upon findings reported in McLaren et al. (2021), which highlighted that protein *quality* should be incorporated into the nLCA framework when protein is being used as a functional unit. McAuliffe et al. (2023) used a protein quality assessment system known as Digestible Indispensable Amino Acid Score (DIAAS) to generate an 'adjusted' protein functional unit. The protein-quality functional unit was applied to the carbon footprints (kg CO2-eq/100g protein) and land occupation (m^2*year/100g protein) of four animal-based (dairy beef; cheese; eggs; pork) and four plant-based (nuts; peas; tofu; wheat) products. The same analysis was carried using unadjusted protein as a functional unit. The study revealed that animal-based products scored more than 100% (122%–141%) DIAAS due to their higher proportion of IAAs, highly digestible structure and lack of inhibitory compounds; tofu had the highest plant-based DIAAS (105%), while the three other plant-based protein sources scored under 100%, with wheat scoring particularly poorly (43%). This led to dairy beef's (DIAAS = ~140%) environmental footprints reduced substantially (~29%) under the adjusted protein functional unit. On the other hand, due to wheat's low DIAAS, its environmental footprints were increased by a factor of 2.3. McAuliffe et al. (2023), however, urged caution related to their novel approach to protein-based functional units. This was due to the fact that, when consumed as part of a meal (or diet), IAAs can be balanced by combining low DIAAS foods with high DIAAS foods to promote protein anabolism through IAA complementarity, emphasizing the importance of balance between contrasting food groups, that is animal and plant-sourced foods.

6 | FUTURE DIRECTIONS FOR (N)LCA TO IMPROVE FOOD SECURITY AND METHODOLOGICAL RIGOUR

6.1 | Inclusion of carbon stock changes in LCAs

Globally, soil organic carbon (SOC) accumulation as well as carbon uptake in plants (including trees and hedgerows) on agricultural land is expected to hold major potential to mitigate land-based greenhouse gas (GHG) emissions (Petersen et al., 2013). However, there is a lack of reported impacts concerning SOC changes in LCAs of agricultural products (Jebari et al., 2022). This suggests that LCA practitioners may not have a well-defined procedure to account for soil C in their assessments, despite it being a highly debated topic among sustainability experts. The evidence and impacts of C stock changes on LCA may differ among various agricultural products and management practices. For instance, in the case of dairy products, a major contributor to GHG emissions in the agricultural sector, including C stock changes has been shown to reduce the global warming potential of European dairy products by 9% of the overall GHG emissions in moist temperate Spanish grasslands associated with dairy production (Jebari et al., 2022). Regardless of whether being applied to environmental LCA or nLCA, more robust assessments of food supply-chains using dynamic carbon models, such as RothC (Nemo-Klumpp et al., 2017), will have implications for interpretation of (n)LCA results. RRes is currently addressing this gap in knowledge using primary data from the NWFP.

6.2 | Applying nutritional science to LCAs of rotational systems producing multiple co-products

On-going work at the National Agricultural Research Institute of Uruguay (INIA) in collaboration with RRes has been assembling high-resolution data, including carbon stocks, crop yields, soil quality and animal performance in no-till rotational systems which produce multiple co-products. For example, INIA's Palo a Pique Long-Term Experiment was installed in 1995, where the main objective was evaluating no-till technology in four rotational systems under direct grazing in soils with severe limitations (e.g. erosion and degradation risk and poor soil drainage). These systems produce multiple products (e.g. beef, wheat, oat and soybean) both on an annual basis and cross-year basis, depending on the system under investigation. This provides ample opportunities to advance (n)LCA by considering nutritional provision from rotational agricultural systems, an understudied aspect in terms of agri-food sustainability and food security, which produce multiple food products (e.g. Shrestha et al., 2020). For example, nutritional metrics can be applied to the four systems trialed at Palo a Pique (each of which provides different agricultural products) to determine which system produces the most nutritionally- and environmentally friendly outputs at a land-use level rather than at an individual product level, e.g. beef or soybean.

6.3 | Food waste and implications of reducing losses throughout entire supply-chains

Food waste occurs at various stages in the food supply chain (e.g. production, processing, transportation and consumption), with maximum losses (70%) at consumption (e.g. households, restaurants and supermarkets. Therefore, it is essential to consider agri-food systems beyond the farmgate including food losses to avoid environmental impacts from food production which does not get consumed. Food waste can be managed through various means; for instance (in no particular order): composting, anaerobic digestion, incineration, donation to food banks, animal feed production and landfilling ideally with landfill gas utilization, are all promising options for further exploration. Due to the heterogeneity in the characteristics and composition of food waste generated at retail and consumer stage, a region-specific (e.g. national scale) LCA study is essential to evaluate the environmental footprints of food waste and its implications on food security and nutritional provision. Indeed, it is evident that a lot of gaps are available which can be filled with LCA studies beyond the farmgate to reduce the overall environmental impacts of the food supply chain. The availability of reliable foreground and background data is the most critical part of LCA studies. RRes has been extensively working at the farm-scale (i.e. cradle to farmgate) and providing scientific communities, government and farmers with scaled-up and fit-for-purpose sustainability solutions for UK food production. However, in progressing, RRes has acknowledged that nLCA research needs to cover the entire food supply chain and, as a result, has built an LCA team with expertise beyond the farmgate. Future research will consider nutritional implications of food waste for major food commodities consumed within the UK.

6.4 | Future directions for nLCA

In addition to the novel areas of research identified above, RRes is also working alongside global nutritional experts to improve the scientific rigor of nLCA, covering all tiers defined by McAuliffe et al. (2020). For example, as mentioned in Section 5, McAuliffe et al. (2023) provided a simple yet informative case study to build upon in terms of incorporating digestibility and bioavailability of various food items and their nutritional composition into the nLCA framework. Further, on-going work is assessing the complementarity of quality-adjusted metrics for broader nutrients than protein at the meal- and diet-level. Lastly, nLCA has, to date, focused on the intersection between food security and environmental impacts. Future research streams are exploring the nexus between nutritional provision and societal and economic impacts beyond human health including, for example, rural economies, human and animal welfare and food production displacement. Despite its current limitations, nLCA is a promising tool for informing policymaking in terms of delivering equitable, environmentally friendly and healthy food systems across the globe. However, more scientifically robust primary data (i.e. sourced from industry) and interpretation of results (e.g. uncertainty and sensitivity analyses) require urgent attention and methodological development.

ACKNOWLEDGEMENTS

G.A.M., A.L.C., M.R.F.L., L.C. and T.T. were funded by Soil to Nutrition (S2N), a Rothamsted Research Institute Strategic Programme supported by UK Research and Innovation (UKRI)—Biotechnology and Biological Sciences Research Council (BBSRC) under grant awards (BBS/E/C/000I0320) and (BBS/E/C/000I0330). A.J. is funded by BBSRC via a Scientific Initiative Catalyst Fund (SICA). A.K. is funded by the UK Government's Department for Environment, Food and Rural Affairs (Defra). F.P was funded by National Institute of Agricultural Research (INIA) Doctoral Fellowship under National Agency of Research and Innovation (ANII) (MOV_CA_2021_1_171482; Mobility Fellowship).

CONFLICT OF INTEREST STATEMENT

The authors have stated explicitly that there are no conflicts of interest in connection with this article.

DATA AVAILABILITY STATEMENT

All underlying data discussed and summarised in this commentary are available from the individual papers covered herein. Rothamsted Research has an Open Access policy following UKRI mandates; as a result, any data (with the exception of commercially or personally sensitive information) utilised or generated through the RRes LCA Team and collaborators is always made publically available through data repositories and/or data publications. The authors will be happy to provide further information upon request, if necessary.

ORCID

G. A. McAuliffe https://orcid.org/0000-0001-6031-1394
H. Scalabrino https://orcid.org/0009-0002-9376-7720

REFERENCES

de Vries, M., & de Boer, I. J. M. (2010). Comparing environmental impacts for livestock products: A review of life cycle assessments. *Livestock Science*, *128*, 1–11.

Fulgoni, V. L., Keast, D. R., & Drewnowski, A. (2009). Development and validation of the nutrient-rich foods index: A tool to measure nutritional quality of foods. *The Journal of Nutrition*, *139*, 1549–1554.

Jebari, A., Álvaro-Fuentes, J., Pardo, G., Batalla, I., Martín, J. A. R., & Del Prado, A. (2022). Effect of dairy cattle production systems on sustaining soil organic carbon storage in grasslands of northern Spain. *Regional Environmental Change*, *22*, 67.

Lee, M. R. F., McAuliffe, G. A., Tweed, J. K. S., Griffith, B. A., Morgan, S. A., Rivero, M. J., Harris, P., Takahashi, T., & Cardenas, L. (2021). Nutritional value of suckler beef from temperate pasture systems. *Animal*, *15*, 100257.

McAuliffe, G. A., Takahashi, T., Beal, T., Huppertz, T., Leroy, F., Buttriss, J., Collins, A. L., Drewnowski, A., McLaren, S. J., Ortenzi, F., van der Pols, J. C., van Vliet, S., & Lee, M. R. F. (2023). Protein quality as a complementary functional unit in life cycle assessment (LCA). *The International Journal of Life Cycle Assessment*, *28*, 146–155.

McAuliffe, G. A., Takahashi, T., & Lee, M. R. F. (2018). Framework for life cycle assessment of livestock production systems to account for the nutritional quality of final products. *Food and Energy Security*, *7*, e00143.

McAuliffe, G. A., Takahashi, T., & Lee, M. R. F. (2020). Applications of nutritional functional units in commodity-level life cycle assessment (LCA) of Agri-food systems. *The International Journal of Life Cycle Assessment*, *25*, 208–221.

McLaren, S. J., Berardy, A., Henderson, A., Holden, N., Huppertz, T., Jolliet, O., De Camillis, C., Renouf, M., Rugani, B., Saarinen, M., van der Pols, J., Vázquez-Rowe, I., Antón Vallejo, A., Bianchi, M., Chaudhary, A., Chen, C., CooremanAlgoed, M., Dong, H., Grant, T., … van Zanten, H. (2021). *Integration of environment and nutrition in life cycle assessment of food items: Opportunities and challenges*. Food and Agriculture Organization of the United Nations.

Nemo-Klumpp, K., Coleman, K., Dondini, M., Goulding, K., Hastings, A., Jones, M. B., Leifeld, J., Osborne, B., Saunders, M., Scott, T., Teh, Y. A., & Smith, P. (2017). Soil organic carbon (SOC) equilibrium and model initialisation methods: An application to the Rothamsted carbon (RothC) model. *Environmental Modeling and Assessment*, *22*, 215–229.

Petersen, B. M., Knudsen, M. T., Hermansen, J. E., & Halberg, N. (2013). An approach to include soil carbon changes in life cycle assessments. *Journal of Cleaner Production*, *52*, 217–224.

Saarinen, M., Fogelholm, M., Tahvonen, R., & Kurppa, S. (2017). Taking nutrition into account within the life cycle assessment of food products. *Journal of Cleaner Production*, *149*, 828–844.

Shrestha, P., Karim, R. A., Sieverding, H. L., Archer, D. W., Kumar, S., Nleya, T., Graham, C. J., & Stone, J. J. (2020). Life cycle assessment of wheat production and wheat-based crop rotations. *Journal of Environmental Quality*, *49*, 1515–1529.

How to cite this article: McAuliffe, G. A., Takahashi, T., Lee, M. R. F., Jebari, A., Cardenas, L., Kumar, A., Pereyra-Goday, F., Scalabrino, H., & Collins, A. L. (2023). A commentary on key methodological developments related to nutritional life cycle assessment (nLCA) generated throughout a 6-year strategic scientific programme. *Food and Energy Security*, *12*, e480. https://doi.org/10.1002/fes3.480

8.4. Anexo 4. Supplementary Material - Management and Productivity of Key Integrated Crop–Livestock Systems in Uruguay: The Palo a Pique Long-Term Experiment's Third Phase.

	May	June	July	Aug	Sept	Oct	Nov	Dec	Jan	Feb	March	April
Oat	14.4±2.73	13.0±2.05	12.4±2.22	13.7±2.10	10.3±1.43	10.4±0.25	-	-	-	-	-	-
Ryegrass	-	15.6±5.88	15.9±5.70	13.8±2.60	9.4±1.23	9.9±1.84	-	-	-	-	-	-
Sorghum	-	-	-	-	-	-	-	10.9±1.83	10.1±0.97	9.8±0.46	6.5±1.27	-
Permanent Pasture (LR)	11.3±2.79	11.3±3.07	12.9±2.10	11.5±2.10	12.3±2.83	10.2±1.73	10.3±1.48	9.5±1.97	10.3±1.61	11.5±6.44	11±1.23	13±3.15
Permanent Pasture (SR)	17.3±7.66	14.5±4.25	18.4±6.64	18.6±7.35	16±8.22	16.7±5.52	13.2±5.98	13.8±2.76	13.3±3.91	18.9±2.51	14.9±1.12	15.4±3.25
Permanent improvement	9.4±0.42	9.8±1.04	9.8±1.41	9.4±2.14	11.8±5.48	11.5±3.83	8.5±1.74	8.4±2.04	7.1±1.32	10.6±4.15	8.8±2.35	9.2±1.34
Tall fescue	8.7±0.45	16.6±4.57	18±4.21	14.4±1.72	14.1±1.47	10.3±2.81	9.9±2.65	7.4±2.32	8.5±1.05	12.5±1.86	12.5±1.96	12.5±2.21
Natural Grassland	6.8±1.57	6±1.83	7.4±1.41	8.4±0.74	7.8±1.73	9.4±0.38	8±0.15	7.3±0.53	6.4±0.55	7.7	6.4±1.84	7.5±0.71

Table S1. Crude Protein content in each pasture at the 'Palo a Pique' pasture-crop rotations long term experiment in Treinta y Tres, Uruguay.

[1] Crude protein content (%) (average ± sd)

Table S2. Metabolisable Energy content in each pasture at the 'Palo a Pique' pasture-crop rotations long term experiment in Treinta y Tres, Uruguay.

	Metabolisable Energy [1]											
	May	June	July	Aug	Sept	Oct	Nov	Dec	Jan	Feb	March	April
Oat	2.5± 0.34	2.4± 0.16	2.3± 0.12	2.4± 0.18	2.1± 0.33	2.5± 0.13	-	-	-	-	-	-
Ryegrass	-	2.75± 0.10	2.6± 0.63	2.4± 0.17	2.5± 0.12	2.5± 0.17	-	-	-	-	-	-
Sorghum	-	-	-	-	-	-	-	2.5± 0.14	2.4± 0.16	1.6± 1.37	2±0.14	-
Permanent Pasture (LR)	2.3± 0.17	2.25± 0.13	2.3± 0.12	2.2± 0.24	2.3± 0.24	2.3± 0.15	2.3± 0.15	2.3± 0.18	2.3± 0.14	2.3± 0.17	2.3± 0.13	2.3± 0.24
Permanent Pasture (SR)	2.3± 0.45	2.14± 0.31	2.5± 0.24	2.5± 0.35	2.5± 0.14	2.6± 0.26	2.1± 0.87	2.3± 0.28	2.3± 0.32	2.5	2.2± 0.16	1.5± 1.35
Permanent improvement	2.17± 0.06	2.26± 0.11	2.3± 0.21	2.2	2.2	2.4± 0.23	2.1± 0.29	2.1± 0.16	2.2± 0.18	2.2± 0.25	2.3± 0.16	2.2± 0.12
Tall fescue	2.3± 0.11	2.34± 0.15	2.4± 0.32	2.3± 0.10	2.4± 0.14	2.3± 0.33	2.3± 0.28	2.2± 0.14	2.1± 0.18	2.2± 0.15	2.3± 0.15	2.3
Natural Grassland	2.1± 0.06	2.05± 0.08	2.0± 0.13	2.1	2.1± 0.03	2.2± 0.12	2.2	2.1	2.1± 0.19	2.3	2.2	2.2

[1] Metabolisable Energy content (MJ/kg DM) (average ± sd)

Table S3. Neutral detergent fiber content in each pasture at the 'Palo a Pique' pasture-crop rotations long term experiment in Treinta y Tres, Uruguay.

	Neutral detergent fiber [1]											
	May	June	July	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr
Oat	47.2 ±5.57	51.2 ±3.33	54.9 ±2.91	53.9 ±3.61	55.5 ±3.04	39.0 ±26.2	-	-	-	-	-	-
Ryegrass	-	38.7 ±2.01	44.6 ±6.31	47.2 ±3.31	47.4 ±3.52	46.1 ±2.35	-	-	-	-	-	-
Sorghum	-	-	-	-	-	-	-	38 ±32.9	57.5 ±0.72	60.4 ±1.42	67.2 ±2.95	-
Permanent Pasture (LR)	58.3 ±5.25	59.5 ±5.87	56.1 ±7.72	57.6 ±5.69	54.0 ±5.32	55.2 2.31	56.8 ±2.41	58.3 ±3.51	57.7 ±3.72	52.6 ±2.03	57.6 ±3.54	47.8 ±21.5
Permanent Pasture (SR)	52.1 ±17.5	44.2 ±22.3	40.3 ±5.32	44.1 ±12.1	41.8 ±8.19	39.9 ±11.5	47.6 ±8.21	52.7 ±7.23	41.5 ±16.5	45 ±3.41	50.5 ±3.65	38 ±33.1
Permanent improvement	69.6 ±5.43	63.5 ±3.10	62.6 ±4.20	61.6 ±2.62	57.2 ±6.54	56.9 ±7.51	63.4 ±5.82	66.2 ±4.06	67.8 ±2.75	61.9 ±9.72	67.0 ±3.14	66.6 ±2.65
Tall fescue	61.4 ±2.81	59.9 ±4.78	53.2 ±5.19	57.0 ±2.13	53.7 ±2.15	58.5 ±2.51	60.0 ±2.34	63.5 ±3.12	64.5 ±3.45	59.5 ±4.64	60.6 2.42	60.7 ±2.52
Natural Grassland	69.2 ±4.01	70.8 ±6.79	65.9 ±4.72	65.7 ±2.22	59.0 ±2.76	59 ±2.53	62.6 ±0.46	66.9 ±1.42	63.1 ±5.87	66.9	67 ±0.51	67.6 ±0.52

[1] Neutral detergent fiber (%) (average ± sd)

8.5. Anexo 5. Supplementary Material - Carbon footprint of mixed farming crop-livestock rotational-based grazing beef systems using long term experimental data.

Supplementary Material 1. Pasture quality in grazed pastures at Palo a Pique long-term experiment in Treinta y Tres, Uruguay.

Table 1. Crude Protein content (%) (average ± standard deviation) in grazed pastures at Palo a Pique long-term experiment, Treinta y Tres, Uruguay.

Rotation	May-June	June-July	July-Aug.	Aug.-Sept.	Sept.-Oct.	Oct.-Nov.	Nov.-Dec.	Dec.-Jan.	Jan.-Feb.	Feb.-Mar.	March-Apr.
Continuous Cropping											
Calves	9.10± 0.700	9.6± 1.367	10.6± 1.901	9.37± 2.316	10.4± 1.041	8.4± 0.557	8.47± 1.343	9.27± 2.194	8.9± 0.819	8.1± 0.321	8.30± 0.872
Short Rotation											
Heifers	132± 2.905	13.1± 3.329	12.4± 3.287	11.6± 2.055	11.3± 2.554	10.0± 1.201	10.6± 1.704	8.47± 1.595	10.6± 3.386	8.43± 2.053	8.70± 2.828
Cows	12.5± 3.325	9.37± 0.839	11.0± 1.650								
Large Rotation											
Calves	8.67± 3.811	9.40± 3.279	9.87± 0.850	10.8± 1.858	10.9± 2.166	9.90± 0.781	9.13± 1.595	8.07± 1.210	8.83± 3.439	9.43± 2.654	10.4± 3.247
Steers	9.90± 2.265	11.6± 1.193	11.8± 1.311	11.2± 1.249	10.2± 1.124	9.97± 0.850	9.03± 1.767	7.57± 1.250	6.85± 0.212	6.85± 0.212	
Forage Rotation											
Steers	12.65± 0.734	14.91 ±3.27 0	13.3± 0.443	13.1± 0.776	11.8± 2.774	8.15± 1.595	7.60± 0.704	7.36± 0.598	12.7± 5.252	10.9± 3.112	10.8± 3.114

Table 2. Digestibility (%) (average ± standard deviation) in grazed pastures at Palo a Pique long-term experiment, Treinta y Tres, Uruguay

Rotation	May-June	June-July	July-Aug.	Aug.-Sept.	Sept.-Oct.	Oct.-Nov.	Nov.-Dec.	Dec.-Jan.	Jan.-Feb.	Feb.-Mar.	March-Apr.
Continuous Cropping											
Calves	61.6± 2.51	62.7± 2.03	61.3± 2.01	59.9± 3.27	63.1± 1.02	58.8± 2.54	59.2± 2.75	60.4± 3.16	60.7± 2.10	60.7± 3.77	58.9± 5.44
Short Rotation											
Heifers	61.0± 3.03	62.3± 1.45	61.9± 1.5	62.9± 3.69	63.6± 0.74	61.1± 1.63	60.6± 2.23	59.6± 3.81	60.8± 3.89	59.1± 3.38	60.2± 4.03
Cows	63.0± 2.05	60.3± 1.47	60.1± 2.61								
Large Rotation											
Calves	61.1± 4.50	60.0± 4.38	59.2± 1.85	58.8± 1.51	62.3± 1.27	60.8± 1.43	59.8± 1.46	60.2± 3.45	60.6± 3.35	61.1± 5.20	60.5± 5.60
Steers	59.6± 4.2	61.5± 1.39	60.7± 3.25	60.6± 3.37	62.1± 1.7	60.6± 1.4	60.7± 2.4	58.4± 1.48	60.2± 4.03	60.2± 4.03	
Forage Rotation											
Steers	61.6± 1.75	63.8± 2.94	60.5± 1.95	63.0± 2.36	64±0. 93	59.0± 2.68	58.9± 2.06	57.7± 2.06	61.0± 5.28	59.6± 2.26	60.6± 3.50

Supplementary Material 2. Equations and constants according by IPCC (2019).

Metabolic functions and other estimates	Equation	Units
Maintenance (NE_m)	10.3[a]	MJ/day
Activity (NE_a)	10.4[a]	MJ/day
Growth (NE_g)	10.6[a]	MJ/day
Ratio of net energy available in diet for maintenance to digestible energy consumed (REM)	10.14[a]	-
Ratio of net energy available in diet for growth to digestible energy consumed (REG)	10.15[a]	-
Gross Energy	10.16[a]	MJ/day
Emission factor for enteric fermentation from a livestock category	10.21[a]	Kg CH_4/head/year
Volatile solid	10.24[a]	Kg VS/day
Emission factor from manure management (MM)	10.23[a]	Kg CH_4/head/year
Dry matter intake	10.18[a]	Kg DM/day
N intake rates	10.32[a]	Kg N/head/day
N retention rates for cattle	10.33[a]	Kg N/head/day
N excretion rates	10.31[a]	Kg N/head/day
Direct N_2O emissions from MM	10.25[a]	Kg N_2O/year
N losses due to volatilisation from MM	10.26[a]	Kg N/year
N losses due to leaching from MM	10.27[a]	Kg N/year
Indirect emissions of N_2O due to volatilisation from MM	10.28[a]	Kg N_2O/year
Indirect emissions of N_2O due to leaching from MM	10.29[a]	Kg N_2O/year
N_2O from N leaching/runoff from managed soils	11.10[b]	Kg N_2O/year
N_2O from atmospheric deposition of N volatilised	11.11[b]	Kg N_2O/year
Direct N_2O emissions from managed soils	11.2[b]	Kg N_2O/year
N in urine and dung deposited by grazing animals in pastures	11.5[b]	Kg N/year
N from crop residues and forage/pastures renewal	11.6[b]	Kg N/year
Emissions from urea and lime application	11.13[b]	Kg CO_2/year

[a] Chapter 10, emissions from Livestock and Manure Management.

[b] Chapter 11, N_2O Emissions from Managed Soils, and CO_2 Emissions from Lime and Urea Application.

Constant	Equation	Value	Unit	Reference
C_f	10.3	0.322	MJ/day/kg	IPCC (2019)
Activity coefficient (C_a)	10.4	0.17	-	IPCC (2019)
C (females)	10.6	0.8	-	NRC (1996)
C (steers)	10.6	1	-	NRC (1996)
MW (mature weight)	10.6	450	kg	Local data
NE_{mf}	10.18	7	MJ/kg DM	IPCC (2019)
Methane conversion factor (Y_m)	10.21	7	-	IPCC (2019)
B_0	10.23	0.1	m^3 CH_4/kg of volatile solid	Becoña et al (2022)
Ash content of feed	10.24	12	%	Local data
AWMS	10.25; 10.26; 10.27	0.9	-	IPCC (2019)
EF_1	11.2	0.01	-	IPCC (2019)
EF_3	11.5	0.004	-	IPCC (2019)
EF_4	11.11	0.01	-	IPCC (2019)
N_{AG}; N_{BG}; R_{AG}; R_S	11.6	Values for each crop	-	IPCC (2019)
$FRAC_{GASF}$	11.11	0.11	-	IPCC (2019)
$FRAC_{GASM}$	11.11	0.21	-	IPCC (2019)
$FRAC_{LEACH}$	11.10	0.24	-	IPCC (2019)
$FRAC_{REMOVE}$	11.6	0.3-0.5	-	Local data

Supplementary Material 3. Economic income (US$) by product and by rotation (2019-2022), at Palo a Pique long-term experiment in Treinta y Tres, Uruguay.

Output	Continuous Cropping	Short Rotation	Large Rotation	Forage Rotation
Livestock	71,472	111,946	249,064	192,778
Soybean	9,367	9,627	11,006	0
Wheat	2,551	3,054	2,949	0
Oat	0	2,216	2,647	0