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EVALUACION DE HUELLA DE CARBONO Y HUELLA HÍDRICA DE CULTIVOS BIOENERGÉTICOS EN URUGUAY

por

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RESUMEN

La política pública de Uruguay para descarbonizar sus fuentes de energía ha destacado el uso de bioenergía. Sin embargo, la sostenibilidad de esta fuente de energía necesita una evaluación del consumo de agua y su contaminación, así como una estimación de los niveles de emisiones de gases de efecto invernadero (GEI). En este marco, estimamos la huella de carbono (HC) y la huella hídrica (HH) de cuatro sistemas de producción de biomasa: (1) rotación maíz-trigo-sorgo sin retiro de residuos (MWS), (2) rotación maíz-trigo-sorgo con retiro de residuos (MWS-R), (3) sorgo dulce continuo (Ss) y (4) switchgrass (Sw). La estimación de HC siguió una estrategia de análisis del ciclo de vida, considerando los insumos y procesos relevantes para la emisión de GEI de las etapas de preparación del suelo, siembra, post-siembra, cosecha y transporte. Para evaluar la HH, el agua verde y gris se calcularon considerando los diferentes volúmenes de agua involucrados en la producción de cultivos que surgen de la evaporación, la lluvia y la contaminación por fertilizantes. Los sistemas de cultivos anuales tuvieron las emisiones más altas (1,6-2,84 kg $CO_{2\text{-}eq}$ L^{-1}_{etanol}) y los mayores valores de HH (23,1-30,9 m^3 L^{-1}_{etanol}). Las etapas de preparación del suelo, siembra y post-siembra representaron el 80% de las emisiones en promedio en los sistemas de cultivos anuales, mientras que solo la etapa de post-siembra explicó el 60% de las emisiones totales de Sw. Entre los insumos utilizados, el fertilizante nitrogenado representó el 36% de las emisiones de GEI, para todos los sistemas evaluados. El volumen de agua requerido para asimilar los fertilizantes de fósforo y nitrógeno, jugó un papel importante en los sistemas de cultivo de bioetanol. En sistemas anuales, el agua gris fue la fracción principal (87%) de la HH total.

Palabras clave: bioenergía, emisiones de gases de efecto invernadero, huella hídrica, análisis del ciclo de vida, intensificación.

EVALUATION OF CARBON FOOTPRINT AND WATER FOOTPRINT OF BIOENERGY CROPS IN URUGUAY

SUMMARY

Uruguay public policy to decarbonize its energy sources highlighted the use of bioenergy. However, the sustainability of this energy source needs an assessment of water consumption and pollution as well as an estimation of the levels of greenhouse gas emissions (GHG). In this framework, we estimate the carbon footprint (CF) and water footprint (WF) of four bioethanol cropping systems: (1) maize-wheat-sorghum rotation without harvested crop residues (MWS), (2) maize-wheat-sorghum rotation with harvested crop residues (MWS-R), (3) continuos sweet sorghum (Ss) and (4) switchgrass (Sw). The CF estimation followed a life cycle analysis strategy, considering the relevant inputs and processes for the emission of GHG from crop management phases of soil preparation, planting, post-planting, harvesting, and transport. In order to assess the WF of bioethanol production green and grey components were calculated by considering the different volumes of water involved in the crop production arising from evaporation, rainfall, and fertilizer pollution. Annual crop systems had the highest emissions (1.6-2.84 kg CO_{2-eq} $L^{-1}{}_{ethanol}$), and the largest WFs (23.1-30.9 m^3 $L^{-1}{}_{ethanol}$). The stages of soil preparation, planting and post-planting accounted for 80% of the emissions on average in the annual crop systems, while only post-planting explained 60% of the total emissions for Sw. Among the inputs used, nitrogen fertilizer represented 36% of GHG emissions (averaged over bioethanol cropping systems). The volume required to assimilate phosphorous and nitrogen fertilizers, plays a significant role in the bioethanol cropping systems. In annual systems, WF_{grey} was the main fraction (87%) of WF_T.

Keywords: bioenergy, greenhouse gas emissions, water footprint, life cycle assessment, intensification.

1. <u>INTRODUCCIÓN</u>

1.1. MARCO GENERAL

La preocupación por el cambio climático es creciente a nivel internacional y nacional (Ludeña y Ryfisch, 2015). Una de las causas es el aumento de la concentración atmosférica de los gases de efecto invernadero (GEI) de origen antropogénico, principalmente: dióxido de carbono (CO_2), metano (CH_4) y óxido nitroso (N_2O). Según IEA (2017) las emisiones globales de GEI han aumentado un 40% desde 2000, en gran parte por el uso de combustibles fósiles. La generación de energía a partir de biomasa se postula como parte de la solución para reducir las emisiones de GEI (Cherubini et al., 2009; Fazio y Monti, 2011). A nivel internacional, el 70 % de la energía renovable corresponde a la bioenergía y contribuye aproximadamente el 14% al suministro mundial de energía primaria (IEA, 2018). Se han evaluado diferentes materias primas para producir bioenergía, de primera generación (cultivos alimentarios) y segunda generación (cultivos energéticos o residuos agrícolas) (Morales et al., 2015). Sin embargo, es necesario considerar los beneficios e impactos ambientales asociados con cada uno de ellos (Halvorsen et al., 2009). El consumo y la calidad del agua son una dimensión relevante para la sostenibilidad ambiental en la producción de bioenergía (Zhong et al., 2018). La demanda de cultivos energéticos puede conducir a aumentar la presión sobre los recursos hídricos por el consumo inadecuado de agua y su contaminación (Mekonnen y Hoekstra, 2011).

Se ha suscitado un debate importante sobre los beneficios ambientales reales de los diferentes combustibles alternativos a los combustibles fósiles actualmente utilizados en el transporte. Buscando cumplir con los objetivos de sostenibilidad, la Directiva de la Unión Europea sobre Energías Renovables (RED) (EC Directive, 2009) determinó que el 10% del consumo final de energía en el transporte para el 2020, procediera de fuentes renovables, pero al mismo tiempo exige que para el cumplimiento de esta obligación, los biocombustibles deben cumplir criterios de sostenibilidad. La RED mediante el decreto 2009/28/EC, determinó que a partir de

2018 el ahorro de emisiones de GEI será al menos del 60% para biocombustibles y biolíquidos en comparación con los carburantes fósiles, a los que sustituyen (EC Directive, 2009). Además, estas directivas modificaron la metodología para el cálculo de la huella de carbono de los biocombustibles, al incluir las emisiones por los cambios en el uso del suelo y la pérdida de COS (carbono orgánico del suelo) por erosión del suelo (EU Regulation 2018/841, 2018). Por tanto, el incremento del uso de los biocombustibles debería ir acompañado de un análisis detallado de su impacto medioambiental, económico y social para determinar el perfil de cada opción respecto a los combustibles tradicionales. Este impacto depende en gran medida de las condiciones de producción (materias primas) y logística (maquinaria, transporte y almacenamiento) de cada país, en especial las materias primas que se usen para su obtención.

En Uruguay, el transporte y la industria son los sectores que más energía consumen, donde el transporte consume un 28% del total de energía y emite el 55% de las GEI del sector energético (MIEM, 2018). En el año 2002 fue aprobada la Ley de Producción de Combustibles Alternativos, Renovables y Sustitutivos de los Derivados del Petróleo (Ley Nº 17.567, 2002), que declaró de interés nacional la producción de combustibles renovables y sustitutos de los derivados del petróleo elaborados con materia prima nacional de origen tanto animal como vegetal. Lo que se complementó con la Ley de Agrocombustibles, que fomenta y regula la producción, la comercialización y la utilización de biocombustibles (etanol y biodiésel) y establece un uso mínimo de 5% de bioetanol en las gasolinas (Ley Nº 18.195, 2007). La política energética nacional se ha adaptado a los objetivos y criterios del Protocolo de Kyoto y el Acuerdo de París mediante la diversificación y descarbonización de las fuentes de energía, desarrollando un mercado de combustible líquido y generación de energía eléctrica con biomasa (Ludeña y Ryfisch, 2015). De esta manera se ha disminuido el consumo de petróleo en un 25% y se han incrementado las energías renovables un 62% entre 2006 y 2016. Por lo tanto, se pasó de un perfil importador a exportador de energía eléctrica (Uruguay XXI, 2017). Actualmente los residuos de la agroindustria (forestales y agrícolas) representan dos tercios de la bioenergía actual generada. La producción de biomasa en 2017 fue del 43% del total de la energía nacional producida, mientras que en el período 1990-2007 fue solo del 20% (Uruguay XXI, 2017). La compañía petrolera estatal de Uruguay, ANCAP, ha invertido en la producción nacional de etanol a partir de cultivos anuales, pero hasta este momento, hay poca investigación sobre los impactos ambientales de su programa de biocombustibles.

1.2. METODOLOGÍA DE ANÁLISIS DE CICLO DE VIDA (LCA)

La promoción de los biocombustibles como alternativa a los combustibles fósiles para obtener un combustible teóricamente neutro en carbono, genera también contrapartidas: se producen otros impactos, como el consumo de agua o la pérdida de biodiversidad debido a la expansión en el uso de la tierra. La perspectiva del pensamiento de ciclo de vida ilustra los posibles conflictos y necesarios compromisos que deberemos alcanzar entre los distintos Objetivos de Desarrollo Sostenible: energía sostenible para todos, combatir el cambio climático y sus impactos, agua sostenible para todos o proteger los ecosistemas terrestres y la biodiversidad.

El LCA es una de las metodologías más difundidas para evaluar los efectos ambientales derivados de la producción de biocombustibles y ha sido ampliamente utilizada para evaluar la eficiencia energética y el desempeño ambiental de cultivos bioenergéticos (Cherubini et al. 2009; Fazio y Monti 2011). El LCA es una herramienta que permite evaluar el desempeño ambiental durante todo el ciclo de vida de un producto o servicio en un periodo de tiempo y espacio predefinidos (ISO, 2006a). En primer lugar, se cuantifica la extracción y el consumo de recursos (incluyendo la energía), así como las emisiones al aire, al agua y al suelo durante todas las etapas. Después, se analiza su contribución potencial a las categorías de impacto ambiental. Estas categorías incluyen el cambio climático, toxicidad humana y ecotoxicidad, y deterioro de la base de recursos (por ejemplo, agua, suelo, etc.). Una sistematización integral de los requisitos y pasos del LCA está contenida en las normas ISO 14040:2006 (ISO, 2006a) e ISO 14044:2006 (ISO, 2006b). Según la norma técnica internacional ISO 14040 (ISO, 2006a) la metodología LCA consta de 4 fases que pueden ser retroalimentadas y ajustadas entre sí a lo largo del proceso (Figura 1). Las 4 fases son:

1. Definición de objetivos y alcance del estudio: se definen los motivos para la realización del LCA, los límites del sistema (ej: de la cuna a la fábrica), lo que se desea estudiar y la amplitud del estudio (ej: huella de carbono y agua). Es de vital importancia establecer los límites que se incluirán en el análisis, para tener en claro que fases de la cadena de producción/valor del producto o servicio se van a considerar. Otro factor muy importante que se establece en esta fase es la unidad funcional, la cual es la unidad de referencia utilizada para cuantificar el desempeño de un producto (ej.: kg CO_2 eq L^{-1} etanol producido).

2. Realización del inventario de ciclo de vida: incluye la obtención de los datos para cuantificar las entradas y salidas, de materia y energía de los sistemas relacionados con la unidad funcional evaluada.

3. Análisis de inventario y evaluación de impacto: se convierten los consumos y emisiones en unidades estandarizadas del impacto ambiental que se desea evaluar, por ejemplo: $CO_{2\text{-eq}}$ para expresar el calentamiento global potencial y m^3 para expresar el impacto sobre el recurso agua.

4. Interpretación: Esta es la fase final en donde se evalúan los resultados obtenidos en las anteriores fases en función de los objetivos planteados, para establecer conclusiones y facilitar la toma de decisiones. En esta fase se puede determinar cuáles son los procesos dentro del ciclo de vida del producto que generan los mayores impactos ambientales y, por ende, en donde se deberían concentrar los esfuerzos para mejorar el desempeño del producto o servicio.

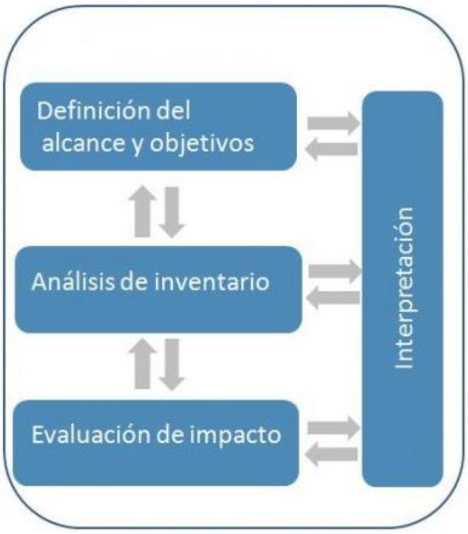

Figura 1: Etapas de un LCA según ISO 14040 (ISO, 2006a)

El propósito de un LCA es obtener conclusiones que puedan respaldar una decisión o que puedan proporcionar un resultado fácilmente comprensible. Los resultados de este tipo de evaluaciones deben permitir identificar problemas ambientales relevantes para obtener conclusiones y recomendaciones que orienten la toma de decisiones. En general, esta metodología se utiliza para distintos objetivos en forma simultánea, a diferentes niveles:

1- Contraste de diferentes opciones (por ejemplo, materias primas) a partir de la estimación de su desempeño ambiental en el tiempo (por ejemplo: huella de carbono, huella hídrica, eutrofización, etc.).
2- Identificar puntos críticos en toda la cadena de valor: identificando las etapas relevantes donde sería pertinente intervenir dado que son las áreas donde es factible que las emisiones/impactos puedan reducirse mediante una mejora tecnológica.
3- Respaldar decisiones estratégicas y aportar datos para los informes de sostenibilidad corporativa o pública. Estas decisiones estratégicas pueden estar a tres niveles: científico, tecnológico y de política pública o privada (ecoetiquetas).
4- Identificar vacíos de información local necesarios para lograr un cálculo de huella con mayor exactitud que el uso de datos no representativos de los sistemas locales.

Los estudios nacionales que han utilizado el LCA en productos agrícolas (Cuadro 1) muestran que la aplicación de esta metodología en el sector agropecuario no es mayor a 9 años. Además, la mayor parte de los estudios han estimado huella de carbono sin considerar en el mismo estudio otros impactos ambientales. La aplicación del LCA ha sido principalmente en los productos de la ganadería (carne y leche) y productos agrícolas (soja, maíz, arroz). En cuanto a la aplicación en el sector bioenergético se han realizado las primeras aproximaciones a la estimación de huella de carbono de algunas materias primas utilizadas en el país para producción de energía (sorgo y caña de azúcar), así como la eficiencia energética (sorgo dulce, sorgo grano, boniato y madera).

Cuadro 1: Estudios nacionales usando LCA del sector agropecuario.

Título	**Documento**	**Cadena agroalimentaria**	**Estimación**	**Unidad Funcional**	**Referencia**
Emisiones de GEI en invernada vacuna en Uruguay	Tesis	Ganadería	Huella de carbono	kg CO_2 eq kg PV	Modernel, 2011.
Emisiones de GEI en sistemas de cría vacuna del Uruguay	Tesis	Ganadería	Huella de carbono	kg CO_2 eq kg PV	Becoña, 2012.
Evaluating the sustainability of potential agro-industrial chains (sweet sorghum, grain sorghum, sweet potato and forestry) for agroenergy production	Reporte	Bioenergía (sorgo dulce, sorgo grano, boniato y madera)	Huella de eficiencia energética	MJ/MJ	Vázquez et al. (2013)
Balance energético de cadenas agro-industriales de interés	Artículo de difusión	Bioenergía (sorgo dulce, sorgo grano, boniato y madera)	Huella de eficiencia energética	MJ/MJ	Carrasco-Letelier et al. (2013)

para la producción de bioenergías					
Huella de carbono del sector arrocero	Artículo de difusión	Arroz	Huella de carbono	kg CO_2 eq kg arroz	Roel et al. (2013)
Global versus local environmental impacts of grazing and confined beef production systems	Artículo científico	Ganadería	Huella de carbono	kg CO_2 eq kg PV	Modernel et al. (2013)
Primer estudio de la huella de carbono de tres cadenas agroexportadoras del Uruguay	Reporte	Cría vacuna, lechería, arroz.	Huella de carbono	kg CO_2 eq kg carne kg CO_2 eq LPE kg CO_2 eq kg arroz	Becoña et al. 2013.
Relación entre la huella de carbono y las prácticas de manejo en predios lecheros en Uruguay	Tesis	Lechería	Huella de carbono	kg CO_2 eq kg LCGP	Lizarralde, 2013.
Sustainability of meat production beyond carbon footprint: a synthesis of case studies from grazing systems in Uruguay	Artículo científico	Ganadería	Huella de carbono	kg CO_2 eq kg PV	Picasso et al. (2014)
Practices to Reduce Milk Carbon Footprint on Grazing Dairy Farms in Southern Uruguay: Case Studies	Artículo científico	Lechería	Huella de carbono	kg CO_2 eq kg LCGP	Lizarralde et al. (2014)
Greenhouse gas emissions of beef cow–calf grazing systems in Uruguay	Artículo científico	Ganadería (23 sistema de cría)	Huella de carbono	kg CO_2 eq kg PV	Becoña et al. (2014)

Análisis de ciclo de vida del proceso de transformación de la caña de azúcar para la producción de bioetanol en la planta de Bella Unión de la empresa ALUR	Reporte	Bioenergía	Huella de carbono	kg CO_2 eq (1 t de caña procesada, 1 MJ de etanol, 1 kg de azúcar, 1 kWh de electricidad)	Herrera et al. (2015)
Análisis de ciclo de vida del proceso de producción de biodiesel a partir de un mix de materias primas grasas en la empresa alcoholes del Uruguay (ALUR)	Reporte	Bioenergía	Huella de carbono	kg CO_2 eq MJ biodiesel	Herrera et al. (2016)
Análisis de ciclo de vida del proceso de producción de etanol a partir de sorgo granífero en la planta de Paysandú de la empresa ALUR	Reporte	Bioenergía	Huella de carbono	kg CO_2 eq MJ etanol	Herrera et al. (2017)
Environmental impacts on water resources from summer crops in rainfed and irrigated systems	Artículo científico	Agricultura (maíz y soja)	Huella hídrica, eco toxicológica y de eutrofización	m^3 ton^{-1}, CTUe ha^{-1} PO_4 eq ton^{-1}	Darré et al. (2019)
Calidad de suelos y eficiencia de uso de energía en rotaciones arroceras contrastantes	Tesis	Arroz, soja, sorgo y carne.	Huella de eficiencia energética	MJ/MJ	Macedo (2018)
Intensification of rice-pasture production systems: changes in energy performance	Artículo científico	Arroz, soja, sorgo y carne.	Huella de eficiencia energética	MJ/MJ	Macedo et al. (2020)

1.3. HUELLA DE CARBONO

La huella de carbono (HC) es un indicador de la cantidad de GEI generados y emitidos durante el ciclo de vida de un servicio o producto a lo largo de la cadena de producción, incluyendo la elaboración de materias primas y el destino final del producto. La HC considera los 6 GEI identificados en el Protocolo de Kioto: dióxido de carbono (CO_2), metano (CH_4), óxido nitroso (N_2O), hidrofluorocarbonos (HFC), perfluorocarbonos (PFC) y hexafluoruro de azufre (SF6). La HC se mide en toneladas equivalentes de dióxido de carbono (CO_{2eq}), a fin de poder expresar las emisiones de los distintos gases de efecto invernadero en una unidad común. Para su cálculo se cuantifican las emisiones (directas e indirectas) de los insumos utilizados para la producción, comercialización, transporte y procesamiento del producto, así como aquellas emisiones generadas para la disposición final de los deshechos.

El objetivo final de la medición de la HC suele ser la reducción de dichas emisiones, contribuyendo así a la mitigación del cambio climático. En lo que refiere al sector bioenergético, se han cuantificado reducciones de 60-70% en las emisiones de GEI durante la producción y el uso de biocombustibles como sustituto de los combustibles fósiles (Kim y Dale, 2005; Fazio y Monti, 2011). Los estudios de huella de carbono sobre materias primas para producción de bioenergía han abarcado cultivos anuales (soja, maíz, colza, sorgo, remolacha, etc.) (Wang et al., 2014; Arodudu et al., 2017) así como también cultivos lignocelulósicos (pasto elefante, miscanthus, switchgrass, etc.) (Gnansounou et al., 2009; Liska et al., 2009). Los cultivos lignocelulósicos producen una mayor cantidad de materia seca en comparación con los cultivos anuales, tienen bajos niveles de erosión del suelo, menores demandas de insumos y, por lo tanto, menores costos de energía asociados (Heaton et al. 2004; Lewandowski y Schmidt, 2006). Además, en contraste con los cultivos anuales, los sistemas de cultivos perennes tienden a acumular COS (Milà I Canals et al., 2007). El COS es un indicador importante de la calidad del suelo que afecta la estructura del suelo, la entrada y retención de agua, el ciclo de nutrientes y la actividad biológica. Las gramíneas perennes tienen la capacidad de aumentar significativamente el COS debido a un extenso sistema de raíces que contribuye a aumentar el carbono (C) en la profundidad del suelo (Follett et al., 2012). Este proceso se denomina secuestro de C, proceso en el que las plantas incorporan CO_2 atmosférico al pool de COS. Es importante evaluar la cantidad de C en la biomasa que puede ser secuestrada en el suelo, ya que la mayor parte del C almacenado en la biomasa aérea se utilizará para la producción de energía (Lal, 2009). La cuantificación de los cambios en el almacenamiento de COS es un factor importante para estimar con exactitud las emisiones de GEI en las evaluaciones del ciclo de vida de la bioenergía (Schmer et al., 2015). Los cambios en el COS afectan el balance de las emisiones de GEI en un LCA. Algunos estudios han intentado cuantificar las tasas de secuestro/emisiones de C asociadas con el uso del suelo y los cambios en el uso del suelo, principalmente utilizando modelos que simulan la rotación de materia orgánica como parte del ciclo del carbono renovable (Brandão et al., 2011). Además del COS, se ha sugerido la erosión como posibles indicadores de los impactos del uso del suelo en LCA (Cherubini et al., 2009). La pérdida de COS por erosión es una causa importante de degradación de la calidad del suelo (Lal, 2004). Tanto las ganancias de C por secuestro como las pérdidas por erosión pueden ser afectadas por el manejo de los residuos (Lemus y Lal, 2005). La utilización de residuos de cultivos en la producción de biocombustibles de segunda generación tiene el potencial de impulsar el sector de la bioenergía sin afectar los precios de los productos alimenticios (Lugato y Jones, 2015). Sin embargo, la eliminación de residuos para biocombustibles puede conducir a un aumento de la erosión del suelo, disminución del COS y, por lo tanto, un aumento de las emisiones de CO_2 (Lal, 2009; Liska et al., 2014).

A nivel local, esta tesis es la primera determinación del efecto de intensificación en la huella de carbono de rotaciones de cultivo de secano para producir biocombustibles. Conocer los valores de HC de diferentes sistemas de rotación permite determinar si la intensificación de los mismos lleva a un incremento de su HC. Además, entender cuáles son las fuentes de emisión y etapas que explican las mayores emisiones, nos permitiría proponer soluciones para reducir las emisiones en la producción de biocombustibles. Esta situación nos plantea las siguientes interrogantes:

1- Sistemas más intensivos, es decir con mayor consumo de insumos, cultivos por año, retiro de residuos, presentarán una mayor HC?
2- Los procesos de secuestro y/o pérdida de C por erosión son relevantes en el balance de GEI de las rotaciones de cultivo?
3- La estimación de la HC permite discriminar cual es el sistema agrícola con menor emisión de CO_2?

1.4. HUELLA HÍDRICA

Como una segunda dimensión de los impactos ambientales potenciales de los biocombustibles, el consumo de agua y su contaminación (incremento de nutrientes y pesticidas) son los más importantes (Ji y Long, 2016). Para evaluar este impacto ambiental, la huella hídrica (HH) permite estimar el volumen de agua consumida y/o contaminada por unidad funcional seleccionada, donde se incluye el uso directo e indirecto del agua (agua virtual) a lo largo de las cadenas de suministro del producto (Hoekstra et al., 2011). Diferentes materias primas han sido evaluadas mediante su HH, encontrándose una amplia gama de valores en función de la materia prima y la ubicación (Gerbens-Leenes et al., 2009; Mekonnen y Hoekstra, 2011; Mathioudakis et al., 2017). Sin embargo, la mayoría de los estudios se han centrado en el consumo de agua sin cuantificar la contaminación. A nivel mundial, la amenaza más importante para la calidad del agua dulce es la eutrofización (Khan y Mohammad, 2014) causada por las emisiones de fósforo (P) y nitrógeno (N). En Uruguay, en varios ríos se han reportado concentraciones de P más altas que el estándar actual establecido (decreto 253/79) (Carrasco-Letelier et al., 2014), estudiándose su vinculación con la erosión de suelos (Carrasco-Letelier y Beretta-Blanco, 2017) y también se han reportado floraciones de cianobacterias (Kruk et al., 2013). La calidad del agua se deteriora en todo el país, con muchos ríos por encima del límite eutrófico (Kruk et al., 2015; Rodríguez-Gallego et al., 2017). Ante un escenario nacional con una alta proporción de biomasa para producción de energía, la cuantificación de la HH de sistemas de producción de biomasa, así como su efecto en la contaminación del agua son de vital importancia.

1.4.1. <u>La Huella Hídrica: Water Footprint Network</u>

La HH de un producto se define, como el volumen total de agua dulce que se utiliza directamente o indirectamente para producir un bien (Hoekstra et al., 2011). Se calcula teniendo en cuenta el consumo de agua directo e indirecto asociado a la contaminación en todos los pasos del proceso productivo. De esta forma se llegan a establecer tres componentes que constituyen la HH de un producto, estos son la huella azul, la huella verde y la huella gris (Figura 2).

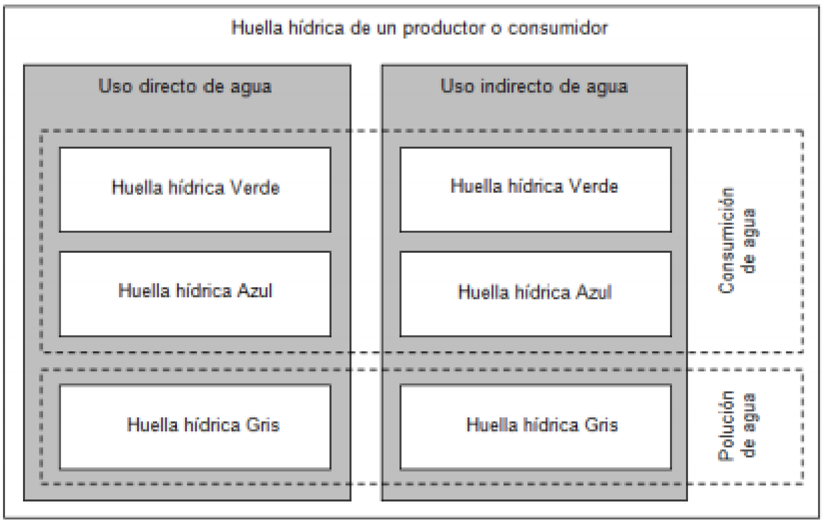

Figura 2: Componentes de la huella hídrica. Adaptado de Hoekstra et al. (2011).

La huella azul corresponde al consumo de los recursos de aguas superficiales y subterráneas a lo largo de la cadena de producción, refiriéndose ese consumo a las salidas del sistema debido a la evaporación, al traslado a otra zona de captación o bien a la incorporación de agua a un producto, esto básicamente corresponde al riego. La huella verde corresponde al consumo de los recursos de agua provenientes de la lluvia almacenada en el suelo, como la humedad del suelo y, por último, la huella gris corresponde al volumen de agua requerida para asimilar la carga de contaminantes hasta alcanzar los valores vigentes de calidad del agua.

1.4.2. La Huella de Agua: norma ISO 14046 (ISO, 2014)

La norma ISO 14046 (ISO, 2014) estandarizó el cálculo de la huella de agua en el marco del ciclo de vida, contemplando las etapas clásicas de un LCA y evaluando el impacto en dos dimensiones: depreciación del recurso y contaminación del mismo (Figura 3); especificando las opciones de cálculo en la ISO/TS 14073 (ISO, 2017).

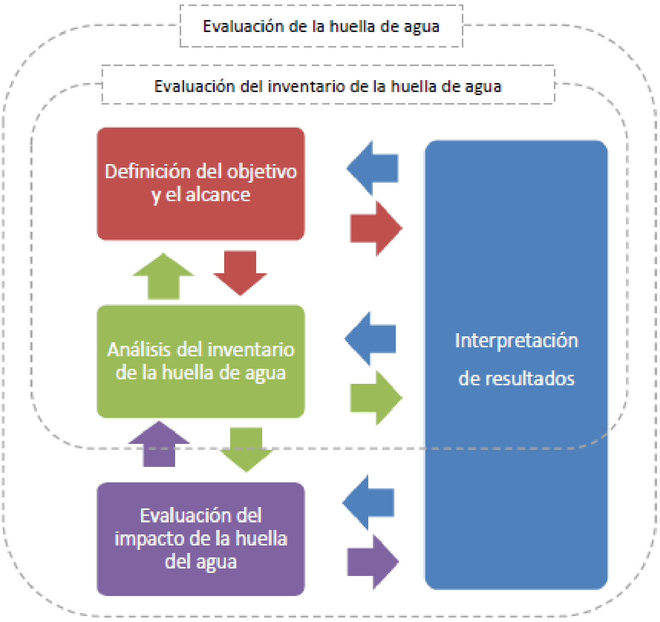

Figura 3: Fases de un estudio de huella de agua. Adaptado de ISO 14046, 2014.

La principal diferencia entre el cálculo de HH propuesta por Hoekstra et al. (2011) y la norma ISO 14046 (ISO, 2014) es que en esta última la contaminación del agua posee diferentes dimensiones de depreciación o impacto (por ejemplo, ecotoxicológica, eutrofización, etc.) detalladas en la ISO/TS 14073 (ISO, 2017). Esta discriminación permite orientar la gestión y dimensionar el impacto ecosistémico, a diferencia de los cálculos de huella gris de agua, estimación que no informa el agente de contaminación que se debe gestionar.

1.5. OBJETIVOS

Estimar las huellas ambientales (carbono y agua) de cuatro sistemas de producción de biomasa con diferente grado de intensificación para la producción de etanol. Esto con el objetivo de conocer si la intensificación de los sistemas (por un mayor consumo de insumos, cultivos por año, retiro de residuos) provoca un incremento de las huellas ambientales.

1.6. HIPÓTESIS DE TRABAJO

Hipótesis 1: La intensificación agrícola (consumo de insumos, cultivos por año, retiro de residuos) aumentará la huella ambiental (carbono y agua) por hectárea de cultivo.

Hipótesis 2: La intensificación agrícola reducirá la huella ambiental por litro de etanol producido.

2. CARBON FOOTPRINT OF FOUR BIOETHANOL CROPPING SYSTEMS IN URUGUAY[1]

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2.1. ABSTRACT

The production of ethanol from biomass, in accordance with the EU Renewable Energy Directive (2009/28/EC), requires an estimation of the levels of greenhouse gas emissions (GHG) from biofuels, as well as a comparison of the values of emission savings as opposed to fossil fuels. In this framework, we estimate the carbon footprint of four bioethanol cropping systems: (1) maize-wheat-sorghum rotation without harvested crop residues (MWS), (2) maize-wheat-sorghum rotation with harvested crop residues (MWS-R), (3) continuous sweet sorghum (Ss) and (4) switchgrass (Sw). The estimation followed a life cycle analysis strategy, considering the relevant inputs and processes for the emission of GHG from crop management phases of: soil preparation, planting, post-planting, harvesting, and transport. The carbon footprint varied between 0.18 and 2.84 $kg\ CO_{2-eq}\ L^{-1}{}_{\text{ethanol}}$. Switchgrass had the smallest footprint and the highest ethanol yield per hectare (4,263 $L\ ha\ yr^{-1}$). Annual crop systems had the highest emissions (1.6-2.84 $kg\ CO_{2-eq}\ L^{-1}{}_{\text{ethanol}}$), 10 times higher than Sw. The stages of soil preparation, planting and post-planting accounted for 80% of the emissions on

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average in the annual crop systems, while only post-planting explained 60% of the total emissions for Sw. Among the inputs used, nitrogen fertilizer represented 36% of GHG emissions (averaged over bioethanol cropping systems). Soil erosion by water accounted for 10% of the soil organic carbon lost in Sw in comparison with MWS-R and Ss systems. In addition, Sw was the only system with carbon sequestration (1.47 $Mg\ CO_{2-eq}\ ha\ yr^{-1}$), a value that corresponded to 70% of global emissions of this bioethanol cropping system.

Keywords: biofuel, greenhouse gas emissions, life cycle assessment, SOC, soil erosion

2.2. RESUMEN

La producción de etanol a partir de biomasa, de acuerdo con la Directiva de Energía Renovable de la UE (2009/28/EC), requiere una estimación de los niveles de emisiones de gases de efecto invernadero (GEI) de los biocombustibles, así como una comparación de los valores de ahorro de emisiones en relación los combustibles fósiles. En este marco, estimamos la huella de carbono de cuatro sistemas de producción de biomasa: (1) rotación de maíz-trigo-sorgo sin retiro de residuos (MWS), (2) rotación de maíz-trigo-sorgo con retiro de residuos (MWS-R) , (3) sorgo dulce (Ss) y (4) switchgrass (Sw). La estimación siguió la metodología de análisis del ciclo de vida, considerando los insumos y procesos relevantes para la emisión de GEI de las fases de: preparación del suelo, siembra, post-siembra, cosecha y transporte. La huella de carbono varió entre 0,18 y 2,84 kg de CO_2-eq L^{-1} etanol. Switchgrass tuvo la menor huella y el mayor rendimiento de etanol por hectárea (4263 L (ha año)$^{-1}$). Los sistemas de cultivo anuales tuvieron las emisiones más altas (1,6-2,84 kg CO_2-eq L^{-1} etanol), 10 veces más que Sw. Las etapas de preparación del suelo, siembra y post-siembra representaron el 80% de las emisiones en promedio en los sistemas de cultivos anuales, mientras que solo la etapa de post-siembra representó el 60% de las emisiones totales de Sw. Entre los insumos utilizados, el fertilizante nitrogenado representó el 36% de las emisiones de GEI (promediado sobre los sistemas de cultivo de bioetanol). En Sw, el proceso de erosión representó el 10% del carbono orgánico del suelo perdido en los sistemas MWS-R y Ss. Además, Sw fue el único sistema con secuestro de carbono

(1,47 Mg CO_2-eq (ha año)$^{-1}$), valor que corresponde al 70% de las emisiones totales de este sistema.

Palabras clave: biocombustibles, emisiones de gases de efecto invernadero, análisis de ciclo de vida, COS, erosión.

2.3. INTRODUCTION

Global greenhouse gas (GHG) emissions have increased 40% since 2000 (IEA, 2017). Biofuels are postulated as part of the solution to reduce GHG emissions and increase energy security (Farrell et al., 2006; Mathews, 2007). The production of biomass to generate biofuels could reduce GHG emissions (Cherubini et al., 2009; Fazio and Monti, 2011). Different raw materials have been evaluated to produce biofuel (Morales et al., 2015); however, it is necessary to consider the environmental benefits and impacts associated with each of them (Halvorsen et al., 2009). In Uruguay, there are several agricultural long-term experiments (LTE) for production and environmental evaluation of different biofuel technology options (Siri-Prieto, 2013).

Significant reductions in GHG emissions have been quantified during the production and use of biofuels as a substitute for fossil fuels (Kim and Dale, 2005; Fazio and Monti, 2011). Several studies on the carbon footprint have been conducted on annual crops (Wang et al., 2014; Arodudu et al., 2017) and lignocellulosic crops (Gnansounou et al., 2009; Liska et al., 2009). Lignocellulosic crops produce a greater amount of dry matter compared to annual crops, have low levels of soil erosion, lower demands for inputs, and therefore, lower associated energy costs (Heaton et al. 2004; Lewandowski and Schmidt, 2006). In addition, in contrast to annual crops, perennial cropping systems tend to accumulate soil organic carbon (SOC) (Milà I Canals et al., 2007). Perennial grasses, like switchgrass, have the ability to significantly increase SOC due to an extensive root system which contributes to increase C in soil depth (Follett et al., 2012). However, most carbon footprint studies on agricultural products focus on single crops, and only a few studies analyzed the cropping systems as a whole

(Goglio et al., 2012; Nemecek et al., 2015). This approach is not adequate for annual crops because each cropping system is affected by the operations along the entire crop rotation system (e.g. crop yield, weed control, nutrient inputs, economic profit, soil erosion control, etc.) (Nemecek et al., 2015).

Quantifying changes in SOC storage is an important factor in accurately estimating GHG emissions in bioenergy life-cycle assessments (LCA) (Schmer et al., 2015). Changes in SOC affect GHG emissions in an LCA and SOC itself is an important indicator of soil quality affecting soil structure, water entry and retention, nutrient cycling, and biological activity. Some studies have attempted to quantify the rates of C sequestration/emissions associated with land use and land use changes, mainly using models simulating organic matter turnover as part of the renewable carbon cycle (Brandão et al., 2011). In addition to SOC, erosion has been suggested as possible indicators for land-use impacts in LCA (Cherubini et al., 2009). In addition, SOC loss by erosion is a major cause of soil quality degradation (Lal, 2004). Many LCA studies have not accounted for potential SOC changes from biomass cropping systems (Larson, 2006). Furthermore, there is no common methodology for considering SOC changes in LCA. As part of the European Union (EU) sustainability framework for biofuels and bioliquids, the EU Renewable Energy Directive (RED) (2009/28/EC) determinate minimum GHG emission requirements that biofuels must meet on a mandatory basis (EC Directive, 2009). In more detail, the Regulation 2018/841 (EU Regulation 2018/841, 2018) determinate that from 2018 that GHG emission saving shall be at least 60% for biofuels and bioliquids. This regulation also determinate the quantification of land use change and soil erosion loss in GHG balance in LCA studies.

The utilization of crop residues in the production of second-generation biofuels has the potential to boost the bioenergy sector without affecting food commodity prices (Lugato and Jones, 2015). However, the removal of residues for biofuels can lead to increase soil erosion, decreased SOC and therefore increase CO_2 emissions. Several studies have tried to quantify the effects of crop residue harvest on soil properties

(Andrews, 2006; Lal, 2009), and its effect on GHG balance (Gabrielle and Gagnaire, 2008; Liska et al., 2014).

The Kyoto Protocol and the Paris Agreement led to a long-term state policy in Uruguay to diversify and increase the decarbonization of its energy sources, with the development of renewable energies (Ludeña and Ryfisch, 2015). From 2006 to 2016, Uruguay oil consumption decreased 25%, and renewable energy grew 62% (Uruguay XXI, 2017). During this period, Uruguay changed from being an importer to an exporter of electric power energy. Thus, currently agroindustry residues (i.e. forest and agricultural) represent two thirds of the current bioenergy generated. Biomass production in 2017 generated 43% of the total national energy, while in the period 1990-2007 it only generated 20% (Uruguay XXI, 2017).

However, the development of bioethanol cropping systems are still limited in Uruguay, and there are several scientific gaps about agronomic and environmental sustainability that need to be addressed. We studied two hypothesis: first, an agricultural intensification (i.e. consumption of inputs, crops per year and/or residue removal) will increase the carbon footprint of system (i.e. cropping system) and, second, the same intensification will reduce the carbon footprint of product (i.e. bioethanol produced). To evaluate the consequence of these hypothesis, we estimated the carbon footprint of four bioethanol cropping systems and their bioethanol production. The studied bioethanol cropping systems have different degrees of intensification and belong to an eight-year agricultural long term experiment (LTE) and are representative of the main agricultural region in Uruguay.

2.4. MATERIALS AND METHODS

2.4.1. Site description and experimental design

The agronomic information used for GHG estimation was based in the database of the eight-year LTE of bioethanol cropping systems maintained at the Mario Cassinoni Agricultural Experimental Station (EEMAC) at Paysandú Department, Uruguay (32°22' S 58°03' W). These bioethanol cropping systems are representative of the production conditions of the western arable soils of the country. Prior to start of this LTE in the summer of 2008, the research site was under a wheat-soybean-maize rotation from 2003 to 2007.

This region has a temperate-humid climate without a dry season according to the Köppen classification (Kottek et al., 2006), with a daily average temperature of 24°C and 12°C for summer and winter, respectively; and an average annual precipitation of 1598 mm. Monthly climate information on 13-year average (2002–2015) was used. The LTE is on a Brunosol Éutrico Typico (Uruguay soil taxonomy, Durán and García Préchac (2007)) or Argiudol (USDA soil taxonomy) that belong to San Manuel Soil Unit, with a soil depth of 40 - 90 cm. The experiment had plots of 150 m^2 (30 m x 5 m) distributed on a landscape with an average slope of 1%.

Treatments within the LTE were arranged in a randomized complete block design with three replications (Table 1). The bioethanol cropping systems were: (1) maize (*Zea mays*)-wheat (*Triticum aestivum*)-grain sorghum (*Sorghum bicolor*) rotation without crop residue removal (MWS),(2) maize-wheat-grain sorghum rotation with crop residue removal (MWS-R),(3) continuous sweet sorghum with winter cover (Ss), and (4) switchgrass (*Panicum virgatum*) (Sw). Treatments 1 and 2 had three crops in two years. Consequently, these two cropping systems include additional plots so all crops for each crop rotation are present every year. The continuous sweet sorghum treatment includes oat (*Avena sativa*) as a cover crop to reduce soil erosion. The fourth treatment was switchgrass, a perennial lignocellulosic crop planted in 2008 and harvested every year during winter season. Crop yields (grain, residues, total biomass) correspond to means of field measurements over eight years (2009-2016). In MWS and MWS-R systems, total yield (dry matter or ethanol) corresponded to the sum of each crop yield (grain or residue) in the crop rotation. Since intensification is ranked based on number of cash crops in the cropping system and harvested biomass, the system intensities were ranked as follows: MWS-R > MWS > Ss > Sw.

Table 1: Bioethanol cropping systems studied: MWS, maize-wheat-grain sorghum; MWS-R, maize-wheat-grain sorghum with residue harvest; Ss, sweet sorghum; Sw, switchgrass.

Crop Systems	Cover crop	Crop residue harvest (%)	Harvested product
Maize-Wheat-Sorghum (MWS)	No	0	Grain
Maize-Wheat-Sorghum (MWS-R)	No	80	Grain and Residue
Continuos sweet sorghum (Ss)	Yes	0	Chopped biomass
Switchgrass (Sw)	No	0	Baled biomass

2.4.2. **Crop management**

No-tillage was used with annual crops (maize, wheat, sorghum, sweet sorghum and oat) while initial Sw cultivation was carried out using conventional tillage. In Sw, the activities and inputs for soil preparation and planting stages were made only in the year of establishment of the crop, so all the inputs and emissions associated with this period were amortized over a period of eight years. Recommended planting densities were used for all crops. Pre- and post-emergent herbicides were applied in all treatments to control weeds as needed. Phosphorus fertilization was carried out at planting with different doses according to the crop using ammonium phosphate as source (18%N-46%P). Nitrogen fertilizer applications were carried out in the post-planting stage, varying rates according to management practices using urea (46% N).

None fertilization for the oat cover crop in the Ss system was done. In MWS-R system, residue was harvested using additional machinery (mower conditioner, baler; Table 2).

2.4.3. Carbon footprint calculations

The carbon footprint estimates for GHG emission were calculated according to criteria defined by ISO 14064-1 (ISO, 2018a), ISO 14067 (ISO, 2018b), ISO 14069 (ISO, 2013) and GHG Protocol (Ranganathan et al., 2018). The GHG estimation considered emissions of carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) of all inputs of cropping systems. A temporal scope of 100 years was considered for the global warming potential (GWP) emissions according to the IPCC Fifth Assessment Report (Allen et al., 2014), with a GWP of 1, 25, and 265 for CO_2, CH_4, and N_2O, respectively.

This study considered two operational levels of the GHG Protocol: (1) direct GHG emissions (fuel consumption and NO_2 emissions), (2) indirect emissions associated with the bioethanol cropping systems (seeds, fertilizers, pesticides and machinery). The potential GHG emission account also considered the fuel consume ratio, the time of use and the energy involved for the development of each kind of utilized machinery. Consumed electric power or thermal energy were not considered. The socioeconomic dimension of the studied systems was not considered in our research because there was not public available information.

The functional unit utilized for the systems was emissions per unit of area expressed in CO_{2-eq} ha^{-1} and emissions per liter of ethanol, expressed in CO_{2-eq} $L^{-1}{}_{\text{ethanol}}$. The life cycle inventory (LCI) considered the phases of field preparation, planting, post-planting, harvest and transport to a bioethanol production plant located 50 km away. Fresh matter yields were transported by truck to ethanol production plant; however, dry matter yields were used for calculations. The carbon footprint was expressed as annual kilograms of CO_2 equivalents and, for two-year crop systems, half of the total impact of the entire rotation was considered for the annual calculations.

Total GHG net emissions correspond to the sum of agricultural and transport inputs, SOC lost by water erosion and mineralization, and N_2O emissions of applied N. In Sw, SOC sequestration was considered in net emissions as a gain of C, therefore, net emissions were lowest than global emissions.

2.4.4. <u>Soil organic carbon and GHG emissions</u>

The GHG emissions related to carbon loss by soil erosion and land-use changes (LUC) were estimated through two models: Erosion 6.0 for soil erosion by water, a USLE/RUSLE model calibrated for Uruguay (García-Préchac et al., 2013), and the AMG model (Andriulo et al., 1999) to estimate LUC effects in the carbon balance. These emissions do not belong to the ISO standards criteria but it is at the framework of EU Directive 2018/841.

Erosion carbon losses, expressed in $Mg\ (ha\ yr)^{-1}$, were estimated with Erosion 6.0 software based on the combination of validated and calibrated USLE-RUSLE model for our conditions (García-Préchac et al., 2013). The Erosion 6.0 software was downloaded from its official website (García-Préchac et al., 2013). This model estimates the soil annual erosion for each agricultural land use based on the USLE factors (Foster et al., 1981) conversion of eroded soil times the SOC concentration of the soil eroded.

For annual crops, SOC losses by LUC effects were estimated with the AMG model (Andriulo et al., 1999). The harvest index was used to estimate the whole above-ground biomass based on the grain yield of each crop. The root biomass in annual crops was estimated as 18% of the above-ground biomass according to Bolinder et al. (2007). The carbon content in the crop residues was assumed as 40% according to Saffih-Hdadi and Mary (2008), and the initial soil organic carbon content was measured by the Walkley & Black method (Nelson and Sommers, 1982). In the case of Sw, a SOC sequestration rate of 0.4 $Mg\ (ha\ yr)^{-1}$ was assumed according to Anderson-Teixeira et al. (2009).

Direct emissions of N_2O from each unit of applied fertilizer were estimated using the emission factors of Bouwman et al. (2002). Direct emissions from stubbles and indirect emissions of N-volatilization, and losses by runoff and leaching were estimated using the emission factors of IPCC (Eggleston et al., 2006). OpenLCA v1.7 software (Green Delta, 2014) was used to model each system, using the impact evaluation method CML Baseline v4,4 (Guinee, 2002) and the USDA database of the Nexus OpenLCA repository.

2.4.5. Statistical analysis

Mean CO_2 emissions of the different treatments were compared by analysis of variance if there were normal distribution and homoscedasticity of data, assumptions that were assessed with the Shapiro-Wilk's test and Levene's test. For data without normal distribution and/or homogeneity of variance, a Kruskal-Wallis test was used with a Dunn's test like and a *post hoc*. All these analyses were performed with a level of significance of 5%. Data analysis were performed using the InfoStat/P program (Di Rienzo et al., 2014) and R software (R Core Team, 2013).

2.5. RESULTS

2.5.1. Life cycle inventory of systems and products

Life cycle inventory is presented in tables 2 and 3. Information of inputs (i.e. seed, fertilizers, pesticides, fuels, machinery) and products (i.e. grain and harvest residues, biomass) was included. In MWS and MWS-R systems, each crop had different grain yields as a result of residue harvest effect in the crop rotation.

Table 2: Annual mean values, from 8 years-LTE, of inputs and activities of the crops: maize (M), wheat (W), grain sorghum (S), sweet sorghum (Ss) and switchgrass (Sw).

	Crop Systems				
	M	W	S	Ss[¥]	Sw
Machinery (n ha^{-1})					
Eccentric plow[§]					2
Disc harrow[§]					1
Chisel plow[§]					1
Sprayer	3	4	2	4	3
Seed drill	1	1	1	2	1
Fertilizer spreader	2	2	2	2	2
Combine harvester	1	1	1		
Mower conditioner[α]	1	1	1		1
Baler[α]	1	1	1		1
Forage harvester				1	
Diesel (L ha^{-1})	68[β] 106[ε]	60[β] 99[ε]	69[β] 104[ε]	166	107
Seed (kg ha^{-1})	20	100	15	108	6
Fertilization (kg ha^{-1})					
Ammonium phosphate (18-46)	100	120	100	120	120
Urea (46% N)	120	160	120	120	200
Pesticides (L ha^{-1})					
Glyphosate	6	6	3	10	8

Atrazine	5		5	5	5
Alpha-metholachlor	1		1	1	
Methsulfuron		0.007			
Dicamba		0.15			
Fungicide		1			

M: maize; W: wheat; S: grain sorghum; Ss: sweet sorghum; Sw: switchgrass.

[¥]: Ss system includes cover crop inputs.

[§]: Conventional tillage was used only for Sw, in the rest of crops no-tillage was used.

[α]: Used for residue harvest in M, W and S. Total biomass harvest in Sw.

[β]: Diesel used in MWS system. Field activities and transport.

[ε]: Diesel used in MWS-R system. Field activities and transport. Also includes harvest and transport of residues.

Table 3: Mean yields of crop systems, from 8 years-LTE, expressed in mega grams of dry matter per hectare for maize (M), wheat (W), grain sorghum (S), sweet sorghum (Ss) and switchgrass (Sw).

		M	W	S	Ss	Sw
	Crop residue harvested	Dry Matter Yield (Mg ha^{-1} yr^{-1})				
Grain	No	4.68	2.97	4.51		
	Yes	4.22	3.22	4.28		
Biomass[α]		2.66	1.94	2.91	9.30	15.22

M: Maize; W: wheat; S: Grain Sorghum; Ss: sweet sorghum; Sw: switchgrass.

[α] : Straw for M, W and S; stem yield for Ss (85% of total biomass); total biomass for Sw.

2.5.2. <u>Biomass production yields</u>

Switchgrass achieved the highest dry matter and ethanol yields, with values of 15 $Mg\ (ha\ yr)^{-1}$ and 4,263 $L\ (ha\ yr)^{-1}$, respectively (Table 4). The ethanol yield produced by Sw was followed by MWS-R with 3,104 $L\ (ha\ yr)^{-1}$. The MWS rotation presented lower yields both in dry matter (6.08 $Mg\ (ha\ yr)^{-1}$) and ethanol (2,128 $L\ (ha\ yr)^{-1}$). On the other hand, the harvest of crop residues showed an increase of 58% and 46% for dry matter and ethanol, respectively.

Table 4: Mean dry matter and ethanol yield of bioethanol cropping systems [8 year long term experiment (LTE)] located at the Mario Cassinoni Agricultural Experimental Station (EEMAC), Paysandú Department, Uruguay.

Crop Systems	Dry Matter Yield ($Mg\ ha^{-1}\ yr^{-1}$)				Ethanol Yield$^{\beta}$ ($L\ ha^{-1}\ yr^{-1}$)	
	Grain	Biomass$^{\alpha}$	Total			
MWS	6.08		6.08	c	2128	c
MWS-R	5.86	3.75	9.62	b	3104	b
Ss		9.30	9.30	b	1861	c
Sw		15.22	15.22	a	4263	a

MWS = Maize-Wheat-Sorghum, MWS-R = Maize-Wheat-Sorghum, Ss= Sweet Sorghum, and Sw = Switchgrass. Values followed by different letters are significantly different at $P < 0.05$.

$^{\alpha}$: Straw yield for M, W, and S; stem yield for Ss (85% of total biomass); total biomass for Sw.

$^{\beta}$: Assuming an ethanol yield of 350, 280 and 200 $L\ Mg^{-1}$ of dry matter of grain, lignocellulosic and sweet sorghum stem, respectively. Theoretical ethanol yields were calculated using the theoretical ethanol conversion factors by the US Energy Department (DOE, 2009), Gnansounou et al. (2005) and McLauhlin et al. (1999).

2.5.3. <u>Bioethanol potential GHG emissions</u>

Net CO_2 emissions from the different systems (Table 5) showed three emission profiles: low emission with the Sw; medium emission for MWS crop rotation; and high emissions for the Ss and the MWS-R crop rotation; with 619, 3162, 4703 and 5078 $kg\ CO_{2-eq}\ (ha\ yr)^{-1}$, respectively (Fig. 1).

The CO_2 equivalent emissions expressed on ethanol yield identified three main groups: low emissions, for Sw crop (0.18 $kg\ CO_{2-eq}\ L^{-1}{}_{\text{ethanol}}$), medium emissions for MWS and MWS-R with values of 1.60 and 1.70 $kg\ CO_{2-eq}\ L^{-1}{}_{\text{ethanol}}$, respectively and high emissions for Ss (2.84 $kg\ CO_{2-eq}\ L^{-1}{}_{\text{ethanol}}$). However, when emissions were expressed per hectare, MWS-R emissions were 60% higher than MWS crop system.

Table 5: Total net emissions of CO_2 per hectare and per liter of ethanol for the different agricultural systems, based in mean crop values from 8 years-LTE. MWS, maize-wheat-grain sorghum; MWS-R, maize-wheat-grain sorghum with residue harvest; Ss, sweet sorghum; Sw: switchgrass. For CO_2 equivalent emissions per produced ethanol, this was expressed in MJ, considering a LHV of 26.8 $MJ\ Kg^{-1}$ and a density of 790 $g\ L^{-1}$.

Crop Systems	kg CO_2-eq ha^{-1} yr^{-1}	kg CO_2-eq L^{-1} ethanol	g CO_2-eq MJ^{-1} ethanol	GHG reduction (%)$^{\alpha}$
MWS	3,162	1.60 b	76	-9.3
MWS-R	5,078	1.70 b	81	-3.4
Ss	4,703	2.84 a	135	61.1
Sw	619	0.18 c	8.6	-89.7

Values followed by the different letter are significant different for a P < 0.05.

α : For comparison was used actual average emissions from petrol consumed (83.8 $g\ CO_{2eq}\ MJ^{-1}$) reported under Directive 2009/28/EC.

GHG emission saving according the RED (2009/28/EC) was 89.7% for Sw. Annual cropping systems present 9.3 and 3.4% of reduction for MWS and MWS-R, respectively. However, Ss present an increment of GHG emissions 60% compared to fossil fuels.

The GHG emissions of the different bioethanol cropping systems had a negative exponential relationship with yield, like $GHG_{emissions} = \lambda_{cropping\ system}/Yield$ (Fig. 1). Where a $\lambda_{cropping\ system}$ has values of 619, 3162, 4703 and 5078 to Sw, MWS, Ss and MWS-R, respectively. The curvature of this negative exponential relationship was more pronounced with highest λ values.

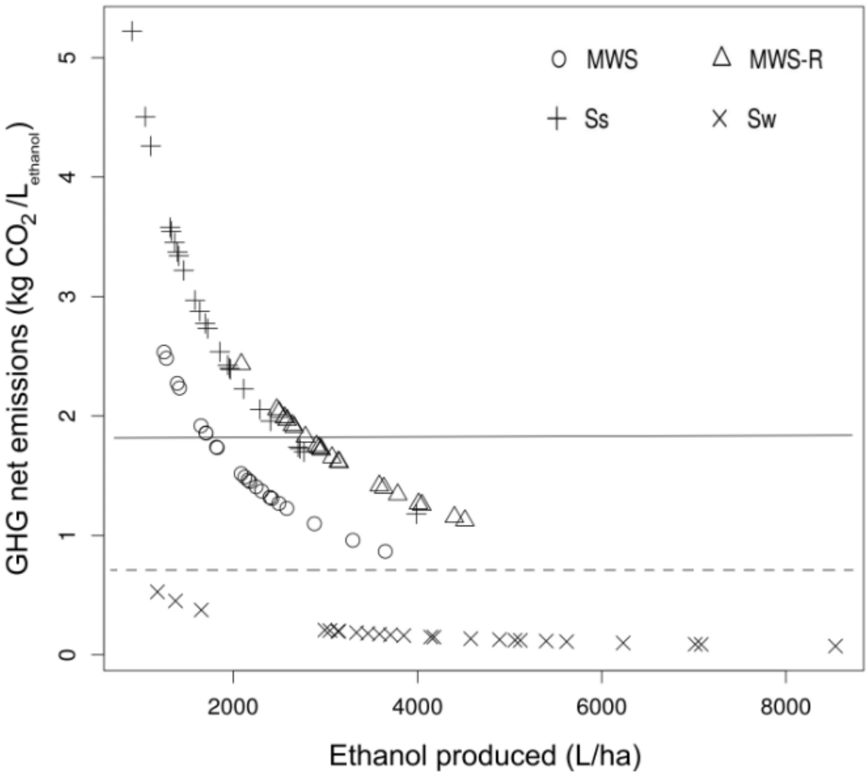

Figure 1: Potential annual GHG emissions for each bioethanol cropping system. The GHG emissions were expressed as CO_{2-eq} per liter ethanol and crop rotation yields in liters of ethanol produced per hectare per year. The GHG references values of petrol (Directive 2009/28/EC) and its 40% was indicated with a continuous and dashed lines, respectively.

2.5.4. NO_2 emissions

Nitrous oxide emissions, expressed in kg CO_2-equivalents, ranged between 272 to 470 kg CO_{2-eq} $(ha\ yr)^{-1}$ (Table 6). The studied systems ranked in the following way according to their nitrous oxide emissions: *MWS > Sw > MWS-R > Ss*. When the nitrous oxide emissions were expressed by ethanol yield for each system, MWS rotation had the highest emissions (0.24 kg CO_{2-eq} L^{-1}_{ethanol}).

Table 6: Emissions of nitrous oxide expressed in N_2O kg CO_{2-eq} $(ha\ yr)^{-1}$ for the agricultural systems: MWS, maize-wheat-grain sorghum; MWS-R, maize-wheat-grain sorghum with harvested crop residues; Ss, continuos sweet sorghum crop; Sw: switchgrass crop.

	Nitrous Oxide Emissions *(kg N_2O-N ha^{-1})*			Annual Total Emissions	
	Direct emissions				
Crop Systems	By applied N	By crop residues	Indirect emissions	*kg CO_2-eq ha^{-1} yr^{-1}*	*kg CO_2-eq L^{-1} ethanol*
MWS	2.26	1.04	0.24	470	0.24 a
MWS-R	2.26	-	0.24	331	0.11 c
Ss	0.95	-	0.08	272	0.16 b
Sw	1.51	-	0.11	431	0.13 c

Values followed by the different letter are significant different for a $P < 0.05$.

2.5.5. Relevance of the components in GHG emissions

The analysis of the participation of GHG emissions for the different stages of production (Fig. 2) showed that 80% of total carbon emissions on average of the annual crop systems were due to operations and inputs used in the phases of soil preparation, planting and post-planting. In contrast, for Sw the stages of soil preparation and planting accounted for only 10% of emissions, while post-planting accounted for 59% of total carbon emissions.

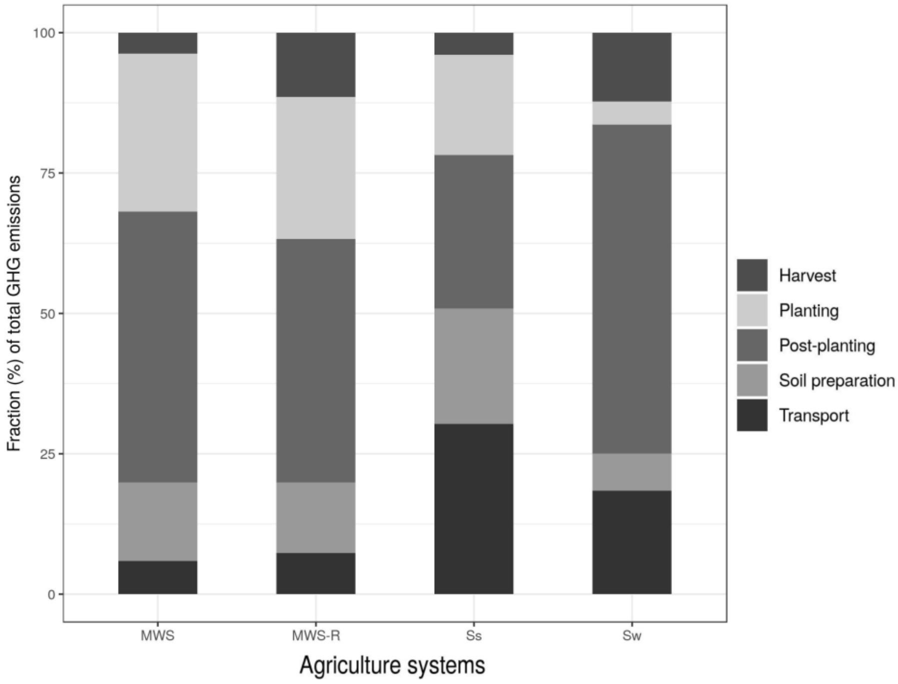

Figure 2: Carbon emissions of each system according to production stage.

The use of nitrogen fertilizers represents 36% of total GHG emissions. Likewise, the use of pesticides and harvesting activities (grain and /or biomass) represent a 16 and 19% of the total emissions, respectively (Fig. 3).

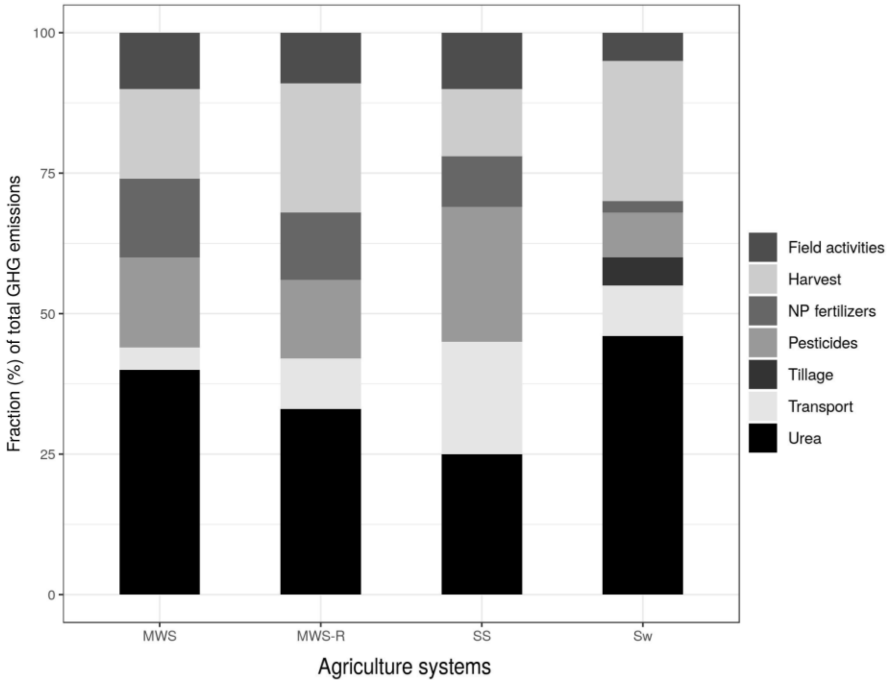

Figure 3: Carbon emissions of each system according to the inputs used.

2.5.6. <u>Emissions and sequestration of GHGs related to soil</u>

The LUC effects on the carbon balance significantly affected the overall GHG emissions of the crop systems evaluated (Fig. 4). In the MWS-R and Ss with a total harvest of biomass, the SOC lost due to soil erosion by water and LUC corresponded to 37 and 20%, respectively, of the total carbon emissions of these systems. Moreover, in the MWS-R system the SOC lost was 70% higher than the MWS system.

The Sw system presented a very different emission profile, where 70% of the emissions can be attributed to production (inputs and activities). The loss of SOC in Sw was 10% of the SOC lost in MWS-R and Ss. In addition, Sw was the only system that sequestered carbon (1.47 $Mg\ CO_{2-eq}\ ha^{-1}$), a value that corresponds to 70% of the total net emissions of this system.

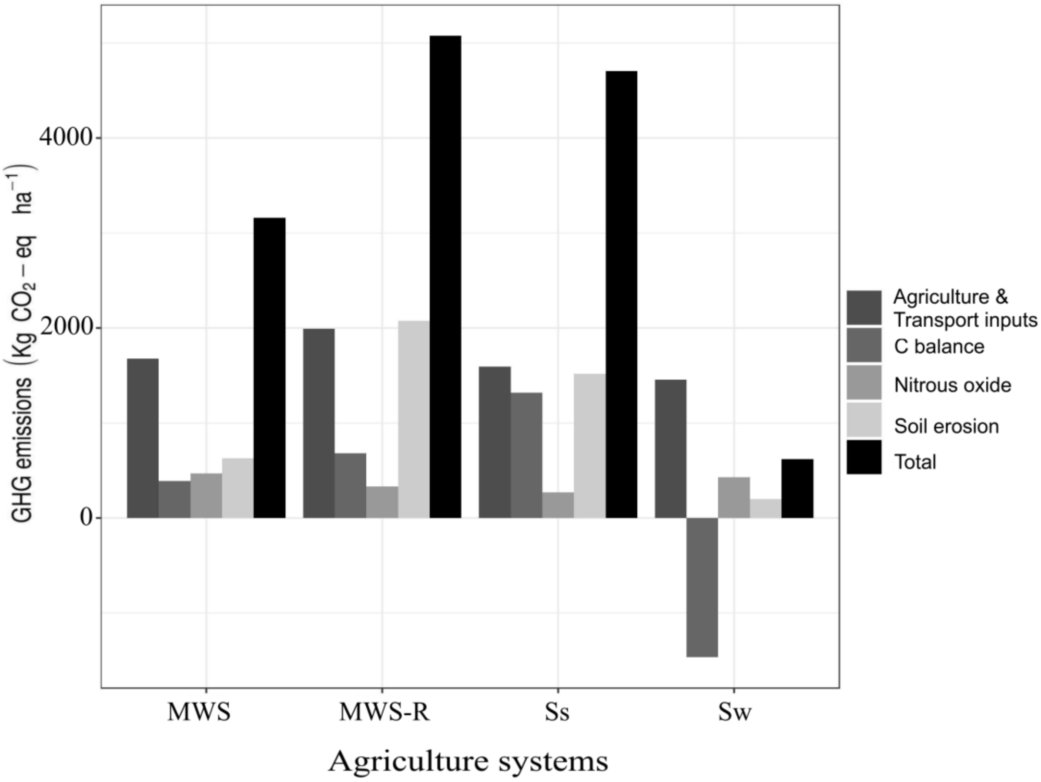

Figure 4: Total GHG emissions (kg CO_{2-eq} $(ha\ yr)^{-1}$) and specific contribution of different components of agriculture systems

2.6. DISCUSSION

2.6.1. Potential GHG emissions

The GHG emissions of the crops evaluated had a similar profile to those previously reported (Fazio and Monti, 2011; Dunn et al., 2013; Muñoz et al., 2014; Qin et al., 2015). Reviewed values could range from 0.18 to 1.5 $kg\ CO_{2-eq}\ L^{-1}$ ethanol with a significant difference in GHG emissions between annual and perennial crops. Muñoz et al. (2014) found that annual crops have values of 0.7 – 1.5 $kg\ CO_{2-eq}\ L^{-1}$ ethanol meanwhile Qin et al. (2015) for switchgrass found values of 0.18 - 0.39 $kg\ CO_{2-eq}$ L^{-1} ethanol. A similar pattern was found when comparing Sw net emissions (0.18 kg $CO_{2-eq}\ L^{-1}$ ethanol), which represented 10% of the potential emissions of the annual crops studied (1.6 - 2.84 $kg\ CO_{2-eq}\ L^{-1}$ ethanol). The reduced carbon footprint of Sw found in this work is likely to the included annual carbon sequestration rate (1.47 $Mg\ CO_{2-eq}$

($ha\ yr$)$^{-1}$, which was equivalent to 70% of potential GHG emissions (2,089 $kg\ CO_{2-eq}$ ($ha\ yr$)$^{-1}$) of the Sw system. Our results agreed with Cherubini and Jungmeier (2010), where soil carbon sequestration was 52% of GHG emissions. Moreover, the found λ values showed the potential GHG emissions have a specific range of possibilities for each bioethanol cropping system. The annual meteorological variation produced different yields in each system, and therefore specific GHG footprints. If the frequency distribution of these GHG footprints is compared with the Directive 2009/28/EC acceptability criteria (GHG equal or at least lower than 40% of petrol emissions), only Sw can satisfy this condition in all the meteorological conditions occurred (Fig. 1). Annual cropping systems, did not reached a value lower than diesel's GHG footprint (Directive 2009/28/EC) in any meteorological conditions.

Fazio and Monti (2011) found that annual crops have values of 0.30 - 0.84 $kg\ CO_{2-eq}\ L^{-1}{}_{\text{ethanol}}$. The higher values found in this work are due to potential GHG emissions from the loss of SOC by LUC and soil erosion by water, which represents 49% of the potential emissions of each agricultural system. Annual crop rotations were not statistically different when their emissions were expressed per unit of ethanol produced ($kg\ CO_{2-eq}\ L^{-1}{}_{\text{ethanol}}$). However, emissions per hectare for the Ss and MWS-R systems were 48 and 61% higher than MWS system, respectively. This difference is due to losses of carbon by LUC and erosion that were three times greater. The MWS-R and Ss treatments were characterized by low soil cover at post-harvest, which generates, according to (Lal, 2009), a low carbon income and therefore a negative carbon balance. Annual crops showed greater GHG emissions because yields overall were lower compared to Sw (Fig. 1). Yield reduction led to significant increases in emissions. This is more important in years where climatic conditions are adverse for the growth of crops. Switchgrass had greater variability in ethanol production from 1170 to 8540 $L\ (ha^{-1}\ yr^{-1})$ (Fig. 1). However, Sw emissions had a more stable pattern. Given the great climatic variability to which crops are exposed, Sw would be the most recommended crop to maintain low GHG emissions.

2.6.2. GHG emissions changes by crop residue management

Results indicate that the harvest of crop residues (MWS-R) increases GHG emissions, similar to the work of Gabrielle and Gagnaire (2008), where the removal of all crop residues caused a SOC reduction of 350 kg of C $(ha\ yr)^{-1}$. In our study, the harvest of crop residues caused a SOC reduction of 620 kg of C $(ha\ yr)^{-1}$, compared to no crop residue removal (Fig. 4). This value is greater than that reported by Gabrielle and Gagnaire (2008) since in our scenario we also consider soil carbon lost via water erosion. In contrast, Cherubini and Jungmeier (2010) evaluated the use of maize and wheat residues for biofuel and concluded that the use of crop residues can generate a 50% decrease in the GHG emissions, but they did not consider losses of SOC by soil erosion. On the other hand, our results agreed with Fazio and Monti (2011) that showed a 20% increase in GHG emissions with crop residue harvest due to the need for higher fertilization rates to compensate for the removal of nutrients with crop residues. In our LTE, fertilization amounts were equal for each crop in all treatments, for this reason is not a source of variation in GHG emissions. However, some nutrient compensation should be considered in the future since crop yield losses with the current fertilization schemes used in Uruguay have been recently reported (Beretta-Blanco et al., 2019).

2.6.3. Effects of soil erosion on GHG emissions

Most LCI studies on crop carbon footprint have not consider carbon losses due to soil erosion, perhaps due to the lack of erosion models calibrated for their conditions or because detailed soil information was not available. However, detailed soils information (scale 1: 20,000) is available for Uruguay, and a calibrated model for estimating soil water erosion under different managements is also available (Durán and García Préchac, 2007). Núñez et al. (2013) highlighted the importance of soil erosion in global GHG accounting, but there is no international agreement for the best way to account erosion's impact on carbon footprint. However, in soil science the

USLE model is widely considered as a valid method to estimate soil water erosion and its use in LCIs (Beck et al., 2010; Núñez et al., 2013).

In our work, systems with removal of crop residues (MWS-R) and in crops with total biomass harvest (Ss), soil carbon lost by water erosion represented 37% of total GHG emissions. According to Uruguay's Law of Soil and Water Conservation (Law N° 15,239), for our study area, a sustainable system must have an erosion rate of less than 7 $Mg\ (ha\ yr)^{-1}$. Under this criteria, the MWS-R and Ss systems with losses of 12.3 and 9.0 $Mg\ (ha\ yr)^{-1}$, respectively, are not sustainable; while the MWS and Sw systems with soil losses of 3.5 and 1.1 $Mg\ (ha\ yr)^{-1}$ are more than acceptable according the law.

2.6.4. Biomass for biofuels

In 2018, the European Renewable Energy (EC Directive, 2009) defined as a criterion for biofuels sustainability to have at least a 60% reduction in GHG relative to fossil fuels. Therefore, taking as reference an emission value of 83.8 $g\ CO_{2eq}/MJ$ for fossil fuels (petrol and diesel), we could conclude that Sw would meet these requirements given that it presented 89.7% lower emissions (Table 5), which is similar to values reported by Adler et al. (2007) and Bai et al. (2010). MWS and MWS-R system only generated a reduction of 9.3 and 3.4%, respectively compared to fossil fuels, making them not suitable for sustainable biofuel production. In addition, Ss had the worst performance as generate an increment in GHG emissions 60% compared to fossil fuels.

These results agreed with the hypotheses proposed by Pimentel and Patzek (2008) related to the limitations of food crops for the production of biofuels. Pimentel and Patzek (2008) raise concerns on the diverse conflicts that exist in using land, water, energy and other environmental resources for food and biofuel production. In addition to the high GHG emissions, grain crop use for bioethanol have direct negative impacts on soil erosion and nutrient exports (Beretta-Blanco et al., 2019), and indirect impacts through LUC and their emissions (Mueller et al., 2011). Development of perennial bioethanol cropping systems are an option for arable land and land of low agricultural suitability (Dauber et al., 2010; Manning et al., 2015).

2.6.5. Agricultural management and associated emissions

Differences in agricultural management practices between the systems studied (number of operations and load of inputs) had a great influence on GHG emissions, representing 53% for annual crops and 90% for perennial crops. These results agreed with those reported by Fazio and Monti (2011), who estimated that agricultural practices represent 35 to 80% of GHG emissions for annual crops, and 61 to 95% for perennial crops. The importance of the different agricultural practices in the GHG balance is greater in Sw systems due to the low carbon losses by LUC and erosion processes.

The post-planting stage in Sw presented a greater contribution to total GHG emissions (60%) due to a greater use of inputs, such as nitrogen fertilizer (e.g. 100 kg of N $(ha\ yr)^{-1}$) which generated N_2O emissions that accounted for 20% of the GHG emissions. This result agreed with other studies where nitrogen fertilizer was one of the most expensive inputs in terms of GHG emissions (Monti et al., 2012; Morales et al., 2015) and defined nitrogen management as a key factor to reduce emissions (Hillier et al., 2009). Then, any strategy that allows reducing nitrogen fertilizer use and/or increasing its efficiency will result in a reduction of GHG emissions. However, in current agricultural systems the capacity of soil to contribute nitrogen is decreasing (Morón et al., 2012) and reductions in nitrogen fertilization lead to lower yields. Including a legume pasture crop phase in agricultural cropping systems to increase soil carbon sequestration and promote biological nitrogen fixation could help reduce N-fertilizer requirements (Köpke and Nemecek, 2010).

In addition, N_2O emissions were strongly affected by crop residue harvest. Maintaining crop residues on the field increased N_2O emissions by 30% compared to residue harvest in MWS-R. A greater amount of biomass on the soil increases the denitrification potential and the capacity to produce N_2O (Cherubini et al., 2009). GHG emissions were affected by the high moisture content (60-70%) of sweet sorghum at harvest since the transport stage demanded a greater consumption of fossil fuel, and thus increasing GHG emissions by 30%. These results agreed with Zegada-Lizarazu and Monti (2012), who found a high moisture content limited the amount of biomass that could be transported, influencing transport costs and associated emissions.

2.6.6. Strengths, limitations and recommendations

Most carbon footprint studies on agricultural products focus on single crops, and only a few studies analyzed the cropping systems as a whole (Goglio et al., 2012; Nemecek et al., 2015). Although this approach could be acceptable for perennial crops, using a similar approach for studying carbon footprint of an industrial process, this approach is not adequate for annual crops because each cropping system is affected by the operations along the entire crop rotation system (e.g. weed control, nutrient inputs, economic profit, soil erosion control, etc.). In addition, in our case two additional variables were added to improve the carbon footprint estimation: the variability of each agriculture system and two additional GHG emission sources (soil erosion and nitrification of residues). The available records allowed us to get a better estimation of potential GHG emission than other studies that are based on study cases that use farmer's data; because the LTE records have standardized procedures and monitoring systems. The climatic variability and crop yield were the unique source of variations.

Compared to annual crop systems, Sw showed the lowest GHG emissions. Therefore, Sw as a biofuel crop should be recommended in Uruguay. Moreover, Sw increased soil carbon sequestration and had a relative low soil erosion. In this context, two main suggestions can be made: first, Sw is a good option for soils with low or very low agriculture diesel suitability in order to avoid competition for land with food crops. Second, Sw could be a good tool for improving physical and chemical properties (Zan et al., 2001; Liebig et al., 2008; Anderson-Teixeira et al., 2009; Monti et al., 2012) of soils degraded by the current intensification and/or expansion of agriculture like described by Beretta-Blanco et al. (2019) and Carrasco-Letelier and Beretta-Blanco (2017). As mentioned earlier, Sw is a feasible option for soils with low agriculture suitability (Fritsche et al., 2010; Cai et al., 2011) where the annual crops have low yield and profitability (Gelfand et al., 2013). Nevertheless, since soils in our study are of medium agriculture suitability, this study's results likely have higher yield than those possible in soils with low agriculture suitability. Therefore, it can be assumed that Sw in low agriculture suitability conditions will have higher CO_2 emission per produced energy equivalent, because the CO_2 emissions by crop inputs will be the same for a lowest yield. For this reasons, Sw must be further studied under low agriculture suitability conditions.

Given the agro-environmental conditions of rainfed crops in Uruguay, our results might be of interest for neighboring countries (Argentina and Brazil) that have similar conditions. These two countries are key players in the biofuel market, and have evaluated different biomass sources for energy generation (Mathews and Goldsztein, 2009; Fontoura et al., 2015; Portugal-Pereira et al., 2015). For example, the adaptation of Sw to different climatic zones has been studied in Argentina (Falasca et al., 2017), including its environmental and socio-economic performances (Diogo et al., 2014; Van Dam et al., 2009). In the case of Brazil, as one of the largest sugarcane ethanol producers, many environmental studies of this feedstock have been made (Bordonal et al., 2018; Jaiswal et al., 2017). In addition, other C4 species had been evaluated as possible feedstocks for bioethanol production, like Ss (Barcelos et al., 2016; Rezende and Richardson, 2017) and perennials as elephant grass (*Pennisetum purpureum* Schum.) (Flores et al., 2012; Fontoura et al., 2015; Rocha et al., 2017).

2.7. CONCLUSIONS

Our hypothesis that an intensification of agricultural systems would result in an increase in GHG emissions should be accepted given the results. The most intensified agricultural system (annual crops) have a carbon footprint higher than the other studied. Further, if intensification called for the removal of crop residues (MWS-R), the GHG emissions were increased. Second, the lowest carbon footprint was that of a perennial crop (Sw), the less intensive crop, per surface area and equivalent energy; where, the post-planting represented 60% of the total GHG emissions due to the use of N-fertilizers. Based on these carbon footprint results, the best option for the production of rainfed bioethanol cropping systems should be Sw.

2.8. REFERENCES

Adler, P.R., Grosso, S.J. Del, Parton, W.J., 2007. Life-cycle assessment of net greenhouse-gas flux for bioenergy cropping systems. Ecol. Appl. 17, 675–691. https://doi.org/10.1890/05-2018

Allen, M., Barros, V., Broome, J., Cramer, W., Christ, R., Church, J., Clarke, L., Dahe, Q., Dasgupta, P., Dubash, N., Edenhofer, O., Elgizouli, I., Field, C., Forster, P., Friedlingstein, P., Fuglestvedt, J., Gomez-Echeverri, L., Hallegatte, S., Hegerl, G., Howden, M., Jiang, K., Jimenez Cisneros, B., Kattsov, V., Lee, H., Mach, K., Marotzke, J., Mastrandrea, M., Meyer, L., Minx, J., Mulugetta, Y., O'Brien, K., Oppenheimer, M., Pachauri, R., Pereira, J., Pichs-Madruga, R., Plattner, G.-K., Pörtner, H.-O., Power, S., Preston, B., Ravindranath, N., Reisinger, A., Riahi, K., Rusticucci, M., Scholes, R., Seyboth, K., Sokona, Y., Stavins, R., Stocker, T., Tschakert, P., van Vuuren, D., van Ypersele, J.-P., Blanco, G., Eby, M., Edmonds, J., Fleurbaey, M., Gerlagh, R., Kartha, S., Kunreuther, H., Rogelj, J., Schaeffer, M., Sedláček, J., Sims, R., Ürge-Vorsatz, D., Victor, D., Yohe, G., 2014. IPCC fifth assessment synthesis report - Climate Change 2014 synthesis report.

Anderson-Teixeira, K.J., Davis, S.C., Masters, M.D., Delucia, E.H., 2009. Changes in soil organic carbon under biofuel crops. GCB Bioenergy. 1, 75–96. https://doi.org/10.1111/j.1757-1707.2008.01001.x

Andrews, S.S., 2006. Crop residue removal for biomass energy production: Effects on soils and recommendations [WWW Document]. USDA-Natural Resour. Conserv. Serv.

Andriulo, A., Mary, B., Guerif, J., 1999. Modelling soil carbon dynamics with various cropping sequences on the rolling pampas, EDP Sciences.

Arodudu, O.T., Helming, K., Voinov, A., Wiggering, H., 2017. Integrating agronomic factors into energy efficiency assessment of agro-bioenergy production – A case study of ethanol and biogas production from maize feedstock. Appl. Energy 198, 426–439. https://doi.org/10.1016/J.APENERGY.2017.02.017

Bai, Y., Luo, L., van der Voet, E., 2010. Life cycle assessment of switchgrass-derived ethanol as transport fuel. Int. J. Life Cycle Assess. 15, 468–477. https://doi.org/10.1007/s11367-010-0177-2

Barcelos, C.A., Maeda, R.N., Santa Anna, L.M.M., Pereira, N., 2016. Sweet sorghum as a whole-crop feedstock for ethanol production. Biomass and Bioenergy. 94, 46–56. https://doi.org/10.1016/j.biombioe.2016.08.012

Beck, T., Bos, U., Wittstock, B., Baitz, M., Fischer, M., Seldbauer, K., 2010. Land Use Indicator Value Calculation in Life Cycle Assessment – Method Report, Cycle.

Beretta-Blanco, A., Pérez, O., Carrasco-Letelier, L., 2019. Soil quality decrease over 13 years of agricultural production. Nutr. Cycl. 114, 45-55. https://doi.org/10.1007/s10705-019-09990-3

Bolinder, M.A., Janzen, H.H., Gregorich, E.G., Angers, D.A., VandenBygaart, A.J., 2007. An approach for estimating net primary productivity and annual carbon inputs to soil for common agricultural crops in Canada. Agric. Ecosyst. Environ. 118, 29–42. https://doi.org/10.1016/J.AGEE.2006.05.013

Bordonal, R. de O., Carvalho, J.L.N., Lal, R., de Figueiredo, E.B., de Oliveira, B.G., La Scala, N., 2018. Sustainability of sugarcane production in Brazil. A review. Agron. Sustain. Dev. 38, 13. https://doi.org/10.1007/s13593-018-0490-x

Bouwman, A.F., Boumans, L.J.M., Batjes, N.H., 2002. Modeling global annual N_2O and NO emissions from fertilized fields. Global Biogeochem. Cycles. 16, 6-1. https://doi.org/10.1029/2001GB001812

Brandão, M., Milà i Canals, L., Clift, R., 2011. Soil organic carbon changes in the cultivation of energy crops: Implications for GHG balances and soil quality for use in LCA. Biomass and Bioenergy. 35, 2323-2336. https://doi.org/10.1016/j.biombioe.2009.10.019

Cai, X., Zhang, X., Wang, D., 2011. Land availability for biofuel production. Environ. Sci. Technol. 45, 334-339. https://doi.org/10.1021/es103338e

Carrasco-Letelier, L., Beretta-Blanco, A., 2017. Soil erosion by water estimated for 99 Uruguayan basins. Cienc. e Investig. Agrar. 44, 184–194. https://doi.org/10.7764/rcia.v44i2.1717

Cherubini, F., Bird, N.D., Cowie, A., Jungmeier, G., Schlamadinger, B., Woess-Gallasch, S., 2009. Energy- and greenhouse gas-based LCA of biofuel and bioenergy systems: Key issues, ranges and recommendations. Resour. Conserv.Recycl. 53, 434–447. https://doi.org/10.1016/j.resconrec.2009.03.013

Cherubini, F., Jungmeier, G., 2010. LCA of a biorefinery concept producing bioethanol, bioenergy, and chemicals from switchgrass. Int. J. Life Cycle Assess. 15, 53–66. https://doi.org/10.1007/s11367-009-0124-2

Dauber, J., Jones, M.B., Stout, J.C., 2010. The impact of biomass crop cultivation on temperate biodiversity. GCB Bioenergy. 2, 289–309. https://doi.org/10.1111/j.1757-1707.2010.01058.x

Diogo, V., Van der Hilst, F., Van Eijck, J., Verstegen, J.A., Hilbert, J., Carballo, S., Volante, J., Faaij, A., 2014. Combining empirical and theory-based land-use modelling approaches to assess economic potential of biofuel production avoiding iLUC: Argentina as a case study. Renew. Sustain. energy Rev. 34, 208–224.

Di Rienzo, J.A., Casanoves, F., Balzarini, M.G., Gonzalez, L., Tablada, M., Robledo, C.W., 2014. InfoStat. Grup. Infoestat, FCA, Univ. Nac. Córdoba, Argentina.

DOE, 2009. Theoretical Ethanol Yield Calculator. US Department of Energy. Consultado 15 marzo 2018. Disponible en: www.eere.energy.gov/biomass/ ethanol yield calculator.html

Dunn, J.B., Mueller, S., Kwon, H., Wang, M.Q., 2013. Land-use change and greenhouse gas emissions from corn and cellulosic ethanol. Biotechnol. Biofuels. 6, 51. https://doi.org/10.1186/1754-6834-6-51

Durán, A., García Préchac, F., 2007. Suelos del Uruguay: Origen, clasificación, manejo y conservación. Editorial Hemisferio Sur, Montevideo, Uruguay.

EC Directive. 2009. Directive 2009/28/EC of The European Parliament and of The Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing directives 2001/77/EC and 2003/30/EC. Official Journal of the European Union.

Eggleston, S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K., 2006. 2006 IPCC guidelines for national greenhouse gas inventories. Institute for Global Environmental Strategies Hayama, Japan.

EU Regulation. 2018. Regulation 2018/841 of the European parliament and of the council of 30 May 2018 on the inclusion of greenhouse gas emissions and removals from land use, land use change and forestry in the 2030 climate and energy framework, and amending Regulation (EU) No 525/2013 and Decision No 529/2013/EU. Official Journal of the European Union 19 June 2018, L 156/1–25. 2018.

Falasca, S., Pitta-Alvarez, S., Miranda Del Fresno, C., 2017. Possibilities for growing switch-grass (Panicum virgatum) as second generation energy crop in dry-subhumid, semiarid and arid regions of the Argentina. J. Cent. Eur. Agric. 18, 95–116.

Farrell, A.E., Plevin, R.J., Turner, B.T., Jones, A.D., O'hare, M., Kammen, D.M., 2006. Ethanol can contribute to energy and environmental goals. Science. 311, 506–508.

Fazio, S., Monti, A., 2011. Life cycle assessment of different bioenergy production systems including perennial and annual crops. Biomass and Bioenergy. 35, 4868–4878. https://doi.org/10.1016/J.BIOMBIOE.2011.10.014

Flores, R.A., Urquiaga, S., Alves, B.J.R., Collier, L.S., Boddey, R.M., 2012. Yield and quality of elephant grass biomass produced in the cerrados region for bioenergy. Eng. Agrícola. 32, 831–839. https://doi.org/10.1590/S01006916 2012000500003

Follett, R.F., Vogel, K.P., Varvel, G.E., Mitchell, R.B., Kimble, J., 2012. Soil Carbon Sequestration by Switchgrass and No-Till Maize Grown for Bioenergy. Bioenergy Res. 5, 866-875. https://doi.org/10.1007/s12155-012-9198-y

Fontoura, C.F., Brandão, L.E., Gomes, L.L., 2015. Elephant grass biorefineries: towards a cleaner Brazilian energy matrix? J. Clean. Prod. 96, 85–93. https://doi.org/10.1016/J.JCLEPRO.2014.02.062

Foster, G.R., McCool, D.K., Renard, K.G., Moldenhauer, W.C., 1981. Conversion of the universal soil loss equation to SI metric units. J. Soil Water Conserv. 36, 355–359.

Fritsche, U.R., Sims, R.E.H., Monti, A., 2010. Direct and indirect land-use competition issues for energy crops and their sustainable production–an overview. Biofuels, Bioprod. Biorefining. 4, 692–704.

Gabrielle, B., Gagnaire, N., 2008. Life-cycle assessment of straw use in bio-ethanol production: A case study based on biophysical modelling. Biomass and Bioenergy. 32, 431–441. https://doi.org/10.1016/j.biombioe.2007.10.017

García-Préchac, F., Hill, M., Clericí, C., 2013. Erosión: modelo de estimación de erosión de suelos en Uruguay y región sur de la Cuenca del Plata (programa informático). Dep. Suelos y Aguas, Fac. Agron. Univ. la República-Ministerio Ganad. Agric. y Pesca-Banco Mundial, Montevideo, Uruguay.

Gelfand, I., Sahajpal, R., Zhang, X., Izaurralde, R.C., Gross, K.L., Robertson, G.P., 2013. Sustainable bioenergy production from marginal lands in the US Midwest. Nature. 493, 514–517. https://doi.org/10.1038/nature11811

Gnansounou, E., Dauriat, A., Wyman, C. E, 2005. Refining sweet sorghum to ethanol and sugar: economic trade-offs in the context of North China. Bioresource technology. 96, 985-1002. https://doi.org/10.1016/j.biortech.2004.09.015

Gnansounou, E., Dauriat, A., Villegas, J., Panichelli, L., 2009. Life cycle assessment of biofuels: Energy and greenhouse gas balances. Bioresour. Technol. 100, 4919–4930. https://doi.org/10.1016/J.BIORTECH.2009.05.067

Goglio, P., Bonari, E., Mazzoncini, M., 2012. LCA of cropping systems with different external input levels for energetic purposes. Biomass and bioenergy. 42, 33–42.

Green Delta, 2014. OpenLCA. [WWW Document]. URL. http://www.openlca.org (accessed 4.6.19).

Guinee, J.B., 2002. Handbook on life cycle assessment operational guide to the ISO standards. Int. J. Life Cycle Assess. 6, 255. https://doi.org/10.1007/bf02978897

Halvorsen, K.E., Barnes, J.R., Solomon, B.D., 2009. Upper Midwestern USA Ethanol Potential from Cellulosic Materials. Soc. Nat. Resour. 22, 931–938. https://doi.org/10.1080/08941920902755382

Heaton, E., Voigt, T., Long, S.P., 2004. 2004. A quantitative review comparing the yields of two candidate C4 perennial biomass crops in relation to nitrogen, temperature and water. Biomass and Bioenergy. 27, 21–30. https://doi.org/10.1016/j.biombioe.2003.10.005

Hillier, J., Hawes, C., Squire, G., Hilton, A., Wale, S., Smith, P., 2009. The carbon footprints of food crop production. Int. J. Agric. Sustain. 7, 107–118.

IEA, 2017. World Energy Outlook 2017, IEA, Paris. Available in: https://www.iea.org/reports/world-energy-outlook-2017

ISO, 2013. ISO/TR 14069:2013(en), Greenhouse gases quantification and reporting of greenhouse gas emissions for organizations-Guidance for the application of ISO 14064-1. Switzerland, Geneva

ISO, 2018a. ISO 14064-1:2018 (en), greenhouse gases-part 1: Specification with guidance at the organization level for quantification and reporting of greenhouse gas emissions and removals.

ISO, 2018b. ISO 14067:2018 (en), Greenhouse gases - carbon footprint of products-requirements and guidelines for quantification.

Jaiswal, D., De Souza, A.P., Larsen, S., Lebauer, D.S., Miguez, F.E., Sparovek, G., Bollero, G., Buckeridge, M.S., Long, S.P., 2017. Brazilian sugarcane ethanol as an expandable green alternative to crude oil use. Nat. Clim. Chang. 7, 788–792. https://doi.org/10.1038/nclimate3410

Kim, S., Dale, B.E., 2005. Life cycle assessment of various cropping systems utilised for producing biofuels: Bioethanol and biodiesel. Biomass and Bioenergy. 29, 426–439. https://doi.org/http://dx.doi.org/10.1016/j.biombioe.2005.06.004

Köpke, U., Nemecek, T., 2010. Ecological services of faba bean. F. Crop. Res. 115, 217–233.

Kottek, M., Grieser, J., Beck, C., Rudolf, B., Rubel, F., 2006. World map of the Köppen-Geiger climate classification updated. Meteorol. Zeitschrift. 15, 259–263.

Lal, R., 2004. Soil carbon sequestration to mitigate climate change. Geoderma. 123, 1-22. https://doi.org/10.1016/j.geoderma.2004.01.032

Lal, R., 2009. Soil quality impacts of residue removal for bioethanol production. Soil Tillage Res. 102, 233-241. https://doi.org/10.1016/j.still.2008.07.003

Larson, E.D., 2006. A review of life-cycle analysis studies on liquid biofuel systems for the transport sector. Energy Sustain. Dev. 10, 109-126. https://doi.org/10.1016/S0973-0826(08)60536-0

Lewandowski, I., Schmidt, U., 2006. Nitrogen, energy and land use efficiencies of miscanthus, reed canary grass and triticale as determined by the boundary line approach. Agric. Ecosyst. Environ. 112, 335–346. https://doi.org/10.1016/j.agee.2005.08.003

Liebig, M.A., Schmer, M.R., Vogel, K.P., Mitchell, R.B., 2008. Soil Carbon Storage by Switchgrass Grown for Bioenergy. BioEnergy Res. 1, 215–222. https://doi.org/10.1007/s12155-008-9019-5

Liska, A.J., Yang, H.S., Bremer, V.R., Klopfenstein, T.J., Walters, D.T., Erickson, G.E., Cassman, K.G., 2009. Improvements in Life Cycle Energy Efficiency and Greenhouse Gas Emissions of Corn-Ethanol. J. Ind. Ecol. 13, 58–74. https://doi.org/10.1111/j.1530-9290.2008.00105.x

Liska, A.J., Yang, H., Milner, M., Goddard, S., Blanco-Canqui, H., Pelton, M.P., Fang, X.X., Zhu, H., Suyker, A.E., 2014. Biofuels from crop residue can reduce soil carbon and increase CO_2 emissions. Nat. Clim. Chang. 4, 398–401. https://doi.org/10.1038/nclimate2187

Ludeña, C.E., Ryfisch, D., 2015. Uruguay: Mitigación y adaptación al cambio climático. Banco Interamericano de Desarrollo. División de Cambio Climático y Sostenibilidad, Washington D.C. Nota Técnica. 860, 1-28.

Lugato, E., Jones, A., 2015. Modelling Soil Organic Carbon Changes Under Different Maize Cropping Scenarios for Cellulosic Ethanol in Europe. Bioenergy Res. 8, 537–545. https://doi.org/10.1007/s12155-014-9529-2

Manning, P., Taylor, G., Hanley, M.E., 2015. Bioenergy, Food Production and Biodiversity - An Unlikely Alliance? GCB Bioenergy. 7, 570–576. https://doi.org/10.1111/gcbb.12173

Mathews, J.A., 2007. Biofuels: What a Biopact between North and South could achieve. Energy Policy. 35, 3550–3570. https://doi.org/10.1016/j.enpol.2007.02.011

Mathews, J.A., Goldsztein, H., 2009. Capturing latecomer advantages in the adoption of biofuels: the case of Argentina. Energy Policy. 37, 326–337.

McLauhlin, J., Bouton, D., Conger, B., Ocumpaugh, W., Parrish, D., Taliaferro, C., Vogel, K., Wullschleger, S. 1999. Developing Switchgrass as a Bioenergy Crop. J. Janick (ed.), ASHS Press, Alexandria, VA.

Milà I Canals, L., Bauer, C., Depestele, J., Dubreuil, A., Knuchel, R.F., Gaillard, G., Michelsen, O., Müller-Wenk, R., Rydgren, B., 2007. Key elements in a framework for land use impact assessment within LCA. Int. J. Life Cycle Assess. 12, 5-15. https://doi.org/10.1065/lca2006.05.250

Monti, A., Barbanti, L., Zatta, A., Zegada-Lizarazu, W., 2012. The contribution of switchgrass in reducing GHG emissions. GCB Bioenergy. 4, 420-434. https://doi.org/10.1111/j.1757-1707.2011.01142.x

Morales, M., Quintero, J., Conejeros, R., Aroca, G., 2015. Life cycle assessment of lignocellulosic bioethanol: Environmental impacts and energy balance. Renew. Sustain. Energy Rev. 42, 1349–1361. https://doi.org/10.1016/J.RSER.2014.10.097

Morón, A., Quincke, A., Molfino, J., Ibáñez, W., García, A., 2012. Soil quality assessment of Uruguayan agricultural soils. Agrociencia. 16, 135–143.

Mueller, S.A., Anderson, J.E., Wallington, T.J., 2011. Impact of biofuel production and other supply and demand factors on food price increases in 2008. Biomass and Bioenergy. 35, 1623–1632.

Muñoz, I., Flury, K., Jungbluth, N., Rigarlsford, G., Canals, L.M., King, H., 2014. Life cycle assessment of bio-based ethanol produced from different agricultural feedstocks. Int. J. Life Cycle Assess. 19, 109–119. https://doi.org/10.1007/s11367-013-0613-1

Nelson, D.W., Sommers, L., 1982. Total carbon, organic carbon, and organic matter 1. Methods soil Anal. Part 2. Chem. Microbiol. Prop. 539–579.

Nemecek, T., Hayer, F., Bonnin, E., Carrouée, B., Schneider, A., Vivier, C., 2015. Designing eco-efficient crop rotations using life cycle assessment of crop combinations. Eur. J. Agron. 65, 40–51.

Núñez, M., Antón, A., Muñoz, P., Rieradevall, J., 2013. Inclusion of soil erosion impacts in life cycle assessment on a global scale: application to energy crops in Spain. Int. J. Life Cycle Assess. 18, 755–767. https://doi.org/10.1007/s11367-012-0525-5

Pimentel, D., Patzek, T., 2008. Ethanol production using corn, switchgrass and wood; Biodiesel production using soybean, in: Biofuels, Solar and Wind as Renewable Energy Systems: Benefits and Risks. Springer, Dordrecht. 373-394. https://doi.org/10.1007/978-1-4020-8654-0_15

Portugal-Pereira, J., Soria, R., Rathmann, R., Schaeffer, R., Szklo, A., 2015. Agricultural and agro-industrial residues-to-energy: Techno-economic and environmental assessment in Brazil. Biomass and Bioenergy. 81, 521–533. https://doi.org/10.1016/J.BIOMBIOE.2015.08.010

Qin, Z., Zhuang, Q., Cai, X., 2015. Bioenergy crop productivity and potential climate change mitigation from marginal lands in the United States: An ecosystem modeling perspective. GCB Bioenergy. 7, 1211–1221. https://doi.org/10.1111/gcbb.12212

R Core Team, 2013. R: A Language and Environment for Statistical Computing.

Ranganathan, J., Corbier, L., Schmitz, S., Oren, K., Dawson, B., Spannagle, M., Bp, M.M., Boileau, P., Canada, E., Frederick, R., Vanderborght, B., Thomson, H.F., Kitamura, K., Woo, C.M., Naseem, &, Kpmg, P., Miner, R., Pricewaterhousecoopers, L.S., Koch, J., Bhattacharjee, S., Cummis, C., Eaton, R., Gillenwater, M., Pricewaterhousecoopers, M.M., Acosta, R., Camobreco, V., 2018. GHG Protocol Initiative Team World Business Council for Sustainable Development Pankaj Bhatia World Resources Institute World Business Council for Sustainable Development Peter Gage World Resources Institute Revision Working Group Core Advisors.

Rezende, M.L., Richardson, J.W., 2017. Risk analysis of using sweet sorghum for ethanol production in southeastern Brazil. Biomass and Bioenergy. 97, 100–107. https://doi.org/10.1016/j.biombioe.2016.12.016

Rocha, J.R. do A.S. de C., Machado, J.C., Carneiro, P.C.S., Carneiro, J. da C., Resende, M.D.V., Pereira, A. Vander, Carneiro, J.E. de S., 2017. Elephant grass ecotypes for bioenergy production via direct combustion of biomass. Ind. Crops Prod. 95, 27–32. https://doi.org/10.1016/j.indcrop.2016.10.014

Saffih-Hdadi, K., Mary, B., 2008. Modeling consequences of straw residues export on soil organic carbon. Soil Biol. Biochem. 40, 594–607. https://doi.org/10.1016/j.soilbio.2007.08.022

Schmer, MR., Jin, VL., Wienhold, BJ., 2015. Sub-surface soil carbon changes affects biofuel greenhouse gas emissions. Biomass and Bioenergy. 81, 31-34. https://doi.org/10.1016/j.biombioe.2015.05.011

Siri-Prieto, G., 2013. Bioenergía en el Uruguay ¿cómo hacerla productiva y sustentable? In: Simposio Nacional de Agricultura (3o, 2013, Paysandú, Uruguay). No Se Llega, Si No Se Sabe a Dónde Ir. Pensar En Las Causas y No Sólo Medir Las Consecuencias. Montevideo, Uruguay, Hemisferio Sur. 21-38.

Uruguay XXI. 2017. Energías renovables: Oportunidades de inversión. Montevideo. Recuperado de: https://www.uruguayxxi.gub.uy/uploads/informacion/Informe%20de%20Energ%C3%A Das%20Renovables%20-%20Setiembre%20de%202017%20-%20Uruguay%20XXI9.pdf. [Consultado: 8/12/2019]

Van Dam, J., Faaij, A.P.C., Hilbert, J., Petruzzi, H., Turkenburg, W.C., 2009. Large-scale bioenergy production from soybeans and switchgrass in Argentina: Part B. Environmental and socio-economic impacts on a regional level. Renew. Sustain. Energy Rev. 13, 1679–1709.

Wang, M., Chen, Y., Xia, X., Li, J., Liu, J., 2014. Energy efficiency and environmental performance of bioethanol production from sweet sorghum stem based on life cycle analysis. Bioresour. Technol. 163, 74–81. https://doi.org/10.1016/j.biortech.2014.04.014

Zan, C.S., Fyles, J.W., Girouard, P., Samson, R.A., 2001. Carbon sequestration in perennial bioenergy, annual corn and uncultivated systems in southern Quebec. Agric. Ecosyst. Environ. 86, 135–144. https://doi.org/10.1016/S0167-8809(00)00273-5

Zegada-Lizarazu, W., Monti, A., 2012. Are we ready to cultivate sweet sorghum as a bioenergy feedstock? A review on field management practices. Biomass and Bioenergy. 40, 1-12. https://doi.org/10.1016/j.biombioe.2012.01.048

3. WATER FOOTPRINT OF BIOETHANOL CROPPING SYSTEMS IN URUGUAY[1]

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3.1. ABSTRACT

Bioenergy is the most widely used type of renewable energy. However, an assessment of water consumption and pollution is necessary to determine the sustainability of this energy source. The Uruguayan public policy to decarbonize energy sources highlighted the use of bioenergy. In this regard, we analyzed the water footprint (WF) of four bioethanol cropping systems: (1) maize-wheat-sorghum rotation without harvested crop residues (MWS), (2) maize-wheat-sorghum rotation with harvested crop residues (MWS-R), (3) continuous sweet sorghum (Ss), and (4) switchgrass (Sw). In order to assess the WF of bioethanol production, green (WF_{green}) and grey (WF_{grey}) components of crop production were calculated by considering the different volumes of water involved in evaporation, rainfall, and fertilizer pollution. Annual cropping systems (i.e., MWS, MWS-R, Ss) had the largest WFs (23.1-30.9 m^3 L^{-1} ethanol). Switchgrass had the lowest values per hectare and per liter of ethanol (12,735 m^3 ha^{-1} yr^{-1} and 3.8 m^3 L^{-1} ethanol, respectively). The volume required to assimilate phosphorous (P) and nitrogen (N) fertilizers played a significant role in bioethanol cropping systems. In annual systems, WF_{grey} was the main fraction (87%)

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of total WF (WF_T). Averaged across all cropping systems, WF_{grey} related to P was 13 times larger than WF_{grey} related to N.

Keywords: Bioenergy, water footprints, grey water, soil erosion, intensification.

3.2. RESUMEN

La bioenergía es el tipo más ampliamente utilizado de energía renovable. Sin embargo, es necesaria una evaluación del consumo y contaminación de agua para determinar la sostenibilidad de esta fuente de energía. La política pública uruguaya para descarbonizar las fuentes de energía ha destacado el uso de bioenergía. En este sentido, analizamos la huella hídrica (HH) de cuatro sistemas de cultivo: (1) rotación maíz-trigo-sorgo sin retiro de residuos (MWS), (2) rotación maíz-trigo-sorgo con retiro de residuos (MWS-R), (3) sorgo dulce continuo (Ss) y (4) switchgrass (Sw). Para evaluar la HH, se calcularon los componentes verdes (HH_{verde}) y gris (HH_{gris}) considerando los diferentes volúmenes de agua involucrados en la evapotranspiración, lluvia y contaminación por fertilizantes. Los sistemas de cultivo anuales (es decir, MWS, MWS-R, Ss) presentaron los mayores valores de HH (23,1-30,9 m^3 L^{-1} etanol). Switchgrass tuvo los valores más bajos por hectárea y por litro de etanol (12.735 m^3 ha^{-1} $año^{-1}$ y 3,8 m^3 L^{-1} etanol, respectivamente). El volumen de agua requerido para asimilar la carga de fósforo (P) y nitrógeno (N) jugó un papel importante. En los sistemas anuales, HH_{gris} fue la fracción principal (87%) de la HH. En el promedio de todos los sistemas de cultivo, HH_{gris} relacionado con P fue 13 veces más grande que HH_{gris} relacionado con N.

Palabras clave: Bioenergía, huellas hídricas, agua gris, erosión del suelo, intensificación.

3.3. INTRODUCTION

Bioenergy is the largest renewable energy source globally, constitutes 70% of all renewable energy sources, and contributes ~14% to global primary energy supply (IEA, 2018). However, water consumption and water quality continue to be key factors affecting environmental sustainability in bioenergy production (Zhong et al., 2018). The increasing demand of energy crops will lead to increased pressure on water resources due to inefficient water consumption and pollution issues (Mekonnen and Hoekstra, 2011). Water requirements for bioenergy production vary according to feedstock used, geographic location, and climatic variables. Therefore, these factors must be considered to determine water consumptive use and to identify critical scenarios and mitigation strategies (Dominguez-Faus et al., 2009). Different raw materials have been evaluated to produce bioenergy (Morales et al., 2015); however, it is necessary to consider long-term environmental benefits and impacts associated with each (Halvorsen et al., 2009). In Uruguay, there are several agricultural long-term experiments (LTE) investigating technology development and environmental performance of different agronomic cropping systems (Siri-Prieto, 2013).

Recently, energy policy in Uruguay has focused on the decarbonization of energy sources. In response to this goal, a liquid fuel market and electric power generation from biofuel and biomass has been developed (Ludeña and Ryfisch, 2015). Within this framework, Uruguay decreased oil consumption by 25% from 2006 to 2016, increased renewable energy by 62%, and became an exporter rather than importer of electrical energy (Uruguay XXI, 2017). Thus, agroindustry residues (i.e. forest and agricultural) currently represent two thirds of current bioenergy production. In 2017, energy from biomass accounted for 43% of total produced energy, compared to only 20% for the 1990-2007 period (Uruguay XXI, 2017). Uruguay's government-owned oil company (i.e., ANCAP) has invested in domestic ethanol production from annual crops, but little prior research has investigated environmental impacts of this biofuel program.

Among potential environmental issues related to biofuel production, water consumption and water pollution are critically important (Ji and Long, 2016). To address these issues, evaluating the water footprint (WF) allows assessment of water volume consumed or polluted when producing a product; this includes direct and indirect water use (virtual water) along product supply chains (Hoekstra et al., 2011). Total water footprint (WF_T) is the sum of three types of water: blue (WF_{blue}), green (WF_{green}), and grey (WF_{grey}), where WF_{blue} is the consumptive use of freshwater from water bodies, WF_{green} refers exclusively to consumed or evapotranspired rainwater in crop production, and WF_{grey} is the volume of freshwater required to assimilate pollutant loads based on existing ambient water quality standards (Hoekstra et al., 2011). Global estimates of WF_T associated with different bioenergy feedstocks varied widely depending on feedstock and location (Gerbens-Leenes et al., 2009; Mekonnen and Hoekstra, 2011; Mathioudakis et al., 2017). For example, Gerbens-Leenes et al. (2009) quantified the WF_T of biomass based energy and compared to fossil energy. Study results indicated that biomass production increased WF_T due to large crop water consumption.

Most studies have focused on water consumption without any quantification of water pollution. In fact, the reduction of WF_{grey} is important in reducing the environmental impact during the production process (Chukalla et al., 2018). However, estimates of WF_{grey} associated with biofuel feedstock production have been relatively limited in WF analyses (Wu et al., 2012; Mekonnen and Hoekstra, 2015) and generally have not consider regional variability of contaminant leaching and loading rates (Zhong et al., 2018). Globally, the prevailing threat to freshwater quality is eutrophication (Khan and Mohammad, 2014) caused by excess phosphorous (P) and nitrogen (N) in freshwater bodies. Most studies only considered N pollution in WF_{grey} estimates (Chukalla et al., 2018); however, P from fertilizers can be a contributor due to losses by leaching and transport by soil erosion to nearby water bodies (Scherer and Pfister, 2015).

The development of bioethanol cropping systems is still limited in Uruguay, and there are several scientific gaps concerning environmental sustainability related to water quality and usage. To address these issues, we evaluated two hypotheses: 1) agricultural intensification (i.e., consumption of inputs, crops per year and/or residue removal) will increase the WF_T of the system (i.e., cropping system), and 2) this intensification will reduce the WF_T of the product (i.e., bioethanol produced). To evaluate the consequence of these hypotheses, we estimated the WF_T of four bioethanol cropping systems and their bioethanol production. The studied bioethanol cropping systems (which belong to an eight-year agricultural LTE) had different degrees of intensification and were representative of the main agricultural region in Uruguay.

3.4. MATERIALS AND METHODS

3.4.1. <u>Site description and experimental design</u>

Agronomic information used for WF_T estimates utilized the database of the eight-year LTE of bioethanol cropping systems maintained at the Mario Cassinoni Agricultural Experimental Station (EEMAC), Paysandú Department, Uruguay (32º22' S 58º03'W). These bioethanol cropping systems are representative of production conditions for western arable soils of the country.

According to the Köeppen classification (Kottek et al., 2006), this region has a temperate-humid climate without a dry season, daily average temperatures of 24°C (summer) and 12°C (winter), and average annual precipitation of 1598 mm. Monthly climate information was used to determine the 13-year average (2002–2015). The LTE is on a Brunosol Éutrico Typico (Uruguay soil taxonomy; Durán and García Préchac, 2007) or Argiudol (USDA soil taxonomy; USDA, 1999) that belongs to the San Manuel soil unit, with a soil depth of 40 – 90cm, and average slope of 1%. The experimental plots were 150 m^2 (30 m x 5 m). Prior to start of this LTE in the summer of 2008, the research site had been under a wheat-soybean-maize rotation from 2003 to 2007.

Treatments within the LTE were arranged in a randomized complete block design with three replications (Table 1). The bioethanol cropping systems were: (1) maize (*Zea mays*)-wheat (*Triticum aestivum*)-grain sorghum (*Sorghum bicolor*) rotation without crop residue removal (MWS), (2) maize-wheat-grain sorghum rotation with crop residue removal (MWS-R), (3) continuous sweet sorghum with winter cover (Ss), and (4) switchgrass (*Panicum virgatum*) (Sw). Treatments 1 and 2 had three crops in two years. Consequently, these two cropping systems include additional plots so all crops for each crop rotation are present every year. The continuous sweet sorghum treatment includes oat (*Avena sativa*) as a cover crop to reduce soil erosion. The fourth treatment was switchgrass, a perennial lignocellulosic crop planted in 2008 and harvested every year during winter season. Crop yields (grain, residues, total biomass) correspond to means of field measurements over eight years (2009-2016). In MWS and MWS-R systems, total yield (dry matter or ethanol) corresponded to the sum of each crop yield (grain or residue) in the crop rotation. Since intensification is ranked based on number of cash crops in the cropping system and harvested biomass, the system intensities were ranked as follows: MWS-R > MWS > Ss > Sw.

3.4.2. <u>Water footprint calculations</u>

We estimated the WF_T of four bioenergy cropping systems following the methodology proposed by Hoekstra et al.(2011). The WF_T for a product is the sum of three types of water: blue, green, and grey (Hoekstra et al., 2011) (Equation 1).

$$WF_T\,(m^3) = WF_{blue} + WF_{green} + WF_{grey} \qquad \text{Equation 1}$$

where blue water is the volume of surface and ground water consumed (irrigation water), green water is soil water loss due to evapotranspiration in the cropping system, and grey water is the water volume needed to dilute pollutant emissions to concentrations allowed by water protection laws (Hoekstra et al., 2011). Since WF_{blue} is zero under rainfed conditions, the WF_T of bioethanol cropping systems correspond to the sum of WF_{green} and WF_{grey}. In the Ss system, WF_T correspond to the sum of WF_T values for Ss and the cover crop (oats). Since cover crops received no fertilizer, WF_T for oats corresponded to WF_{green}.

Table 1: Description of bioethanol cropping systems [8 year long term experiment (LTE)] located at the Mario Cassinoni Agricultural Experimental Station (EEMAC), Paysandú Department, Uruguay.

Crop Systems	Cash crops per year	Crop coverage	Crop residue harvest (%)	Harvested product
Maize-Wheat-Sorghum (MWS)	1.5	No	0	Grain
Maize-Wheat-Sorghum (MWS-R)	1.5	No	80	Grain and Residue
Sweet Sorghum (Ss)	1	Yes	0	Chopped biomass
Switchgrass (Sw)	1	No	0	Baled biomass

Functional units used for comparing WF_T of systems were m^3 ha^{-1} yr^{-1}, and units used for product (ethanol) comparisons were m^3 L^{-1} ethanol. Water consumption per crop was allocated to crop ethanol yield from dry matter (harvested grains and residues) of maize, wheat, grain sorghum, and to total aboveground biomass for sweet sorghum and switchgrass. Crop dry matter and ethanol yields were previously described in Bustamante et al. (Unpublished results) (Table 2).

3.4.2.1. Green water footprint assessment

The crop evaporation requirement (ETc, mm day^{-1}) is the product of reference evapotranspiration (ETo, mm) multiplied by a single crop coefficient (Kc) (Equation 2). The WF_{green} (m^3 ha^{-1}) is equal to the integration of ETc daily values over the growing season. Green water depends on crop-specific evapotranspiration and soil moisture availability in the field (Allen et al., 1998).

$$ETc = ETo \times Kc \qquad \text{Equation 2}$$

Table 2: Mean dry matter and ethanol yield of bioethanol cropping systems [8 year long term experiment (LTE)] located at the Mario Cassinoni Agricultural Experimental Station (EEMAC), Paysandú Department, Uruguay.

Crop Systems	Dry Matter Yield ($Mg\ ha^{-1}\ yr^{-1}$)				Ethanol Yield$^{\beta}$ ($L\ ha^{-1}\ yr^{-1}$)	
	Grain	Biomass$^{\alpha}$	Total			
MWS	6.08		6.08	c	2128	c
MWS-R	5.86	3.75	9.62	b	3104	b
Ss		9.30	9.30	b	1861	c
Sw		15.22	15.22	a	4263	a

MWS = Maize-Wheat-Sorghum, MWS-R = Maize-Wheat-Sorghum, Ss= Sweet Sorghum, and Sw = Switchgrass. Values followed by different letters are significantly different at $P < 0.05$.

$^{\alpha}$: Straw yield for M, W, and S; stem yield for Ss (85% of total biomass); total biomass for Sw.

$^{\beta}$: Assuming an ethanol yield of 350, 280 and 200 L Mg^{-1} of dry matter of grain, lignocellulosic and sweet sorghum stem, respectively. Theoretical ethanol yields were calculated using the theoretical ethanol conversion factors by the US Energy Department (DOE, 2009), Gnansounou et al. (2005) and McLauhlin et al. (1999).

CROPWAT 8.0 (FAO, 2010) software was used to calculate ETc using the FAO Penman-Monteith method (Allen et al., 1998). The 'irrigation schedule CROPWAT option' was used in order to calculate actual evapotranspiration, which included a soil water balance that tracks soil moisture content over time using a daily time step rather than accounting for effective precipitation. The calculated evapotranspiration was referred to the adjusted crop evapotranspiration (ETa), which can be smaller than ETc due to non-optimal conditions. The stress coefficient (Ks) describes the effect of water stress on crop transpiration.

$$ETa = ETc \times Ks \qquad \text{Equation 3}$$

CROPWAT model inputs for estimating evapotranspiration were climate, crop, and soil parameters. Climatic information (temperature, precipitation, humidity, radiation, and wind speed) was measured using a Vantage Pro2 automatic weather station (Model 6510; Davis Instruments, Hayward, CA) located approximately 100 m from the LTE. Monthly climate information based on 13-years averages (2002–2015) was used. Crop parameters such as crop coefficients (Kc) and rooting depths were based on studies by Allen et al. (1998) and Chapagain and Hoekstra (2004). Crop planting and harvest dates and lengths of each crop developmental stage were from the LTE data base (Table 3).

Soil parameters included available soil water content, maximum infiltration rate, maximum rooting depth, and initial soil moisture depletion. Available soil water content (AD_H) in the entire soil profile was estimated as 110 mm using the following empirical function adjusted for Uruguayan soils (Silva et al., 1988):

$$AD_H = FC - PWP \qquad \text{Equation 4}$$

where FC represents field capacity and PWP is the permanent wilting point. These parameters were estimated based on percent contents of sand, silt, clay, and organic matter on soil horizons. A maximum infiltration rate of 90 mm day^{-1} was measured in the LTE by the Müntz method using double ring-infiltrometers (Custodio and Llamas, 1976). An average soil depth of 0.7 m was taken as the maximum rooting depth for the LTE. Different initial soil moisture contents at the start of the growing season for each crop were used.

CROPWAT outputs were real and potential crop water use and irrigation requirements. For our work, we used the real crop water use as our output. This value corresponded to real ETc under rainfed conditions. Rainfed conditions were simulated by the model by choosing to apply no irrigation. In our case of rainfed production, the WF_{green} was equal to total evapotranspiration as simulated by the model.

Table 3: Planting and harvest dates, and crop coefficients (Kc) for crops studied at the Mario Cassinoni Agricultural Experimental Station (EEMAC), Paysandú Department, Uruguay.

	Date$^{\alpha}$		Crops Kc $^{\beta}$		
Crops	Planting	Harvest	Initial	Medium	Final
Maize	Nov-1	Mar-30	0.3	1.20	0.35
Wheat	Jun-15	Nov-11	0.3	1.15	0.25
Grain Sorghum	Dec-1	Apr-5	0.3	1.10	0.55
Sweet Sorghum	Nov-15	Apr-15	0.3	1.20	1.05
Oat	May-1	Aug-31	0.3	1.15	0.25
Switchgrass	Sep-1	Apr-28	0.5	0.90	0.85

$^{\alpha}$: Information from the long term experiment (LTE) database.

$^{\beta}$: Kc values from Allen et al. (1998) and Chapagain and Hoekstra (2004).

3.4.2.2. Grey water footprint assessment

Emissions related to N and P were considered for WF_{grey} estimation. The WF_{grey} corresponded to the nutrient having the largest volume of water. The grey component of the WF ($m^3\ ton^{-1}$) was calculated according the Equation (5):

$$WF_{grey}\ (m^3\ ton^{-1}) = ((AL\ x\ F)\ /\ (Cmax - Cnat))/\ Y \qquad \text{Equation 5}$$

where AL is the annual pollutant load per unit area of each crop expressed in kg $ha^{-1}\ yr^{-1}$, F is the fraction of AL emitted to the ecosystem, Cmax is the maximum permissible pollutant concentration in freshwater (kg m^{-3}), Cnat is the natural pollutant concentration in water (kg m^{-3}), and Y is crop yield (Mg $ha^{-1}\ yr^{-1}$).

Further, we assumed that an average of 10% N and 3% P of applied fertilizers (Table 4) was lost through rainfall and leaching according to Chapagain et al. (2006). Nutrient losses by soil erosion (Mg $ha^{-1}\ yr^{-1}$) were estimated with Erosion 6.0 software based on the combination of validated and calibrated USLE-RUSLE model for Uruguay (García-Préchac et al., 2013). The Erosion 6.0 software was downloaded from its official website (García-Préchac et al., 2013). This model estimates annual soil erosion for each combination of agricultural land use and soil properties based on USLE factors (Foster et al., 1981). Losses of N and P were estimated as the product of the amount of eroded soil and N and P enrichment of the first 5 cm of soil. In this study, we used water concentrations of N and P allowed for tap water production in Uruguay (Decree 253/79, MVOTMA, 1979): 10 mg of nitrate-nitrogen (NO_3-N) and 0.025 mg of phosphorous (PO_4-P) per liter.

Table 4: Fertilizations rates (kg ha^{-1} yr^{-1}) applied to each crop studied at the Mario Cassinoni Agricultural Experimental Station (EEMAC), Paysandú Department, Uruguay.

Crops	Diammonium Phosphate (18%N-46%P)	Urea (46%N)
	kg ha^{-1} yr^{-1}	
Maize	100	120
Wheat	120	160
Grain Sorghum	100	120
Sweet Sorghum	120	120
Oat	0	0
Switchgrass	15	200

3.4.3. **Statistical analysis**

Water footprint means for the different bioethanol cropping systems were compared by analysis of variance (ANOVA). Normal distribution and homoscedasticity were tested with the Shapiro-Wilk's and Levene tests, respectively. When these assumptions were not met, the ANOVA was done with the Kruskal-Wallis test and *post hoc* analysis by the Dunn test. All analyses were performed at a 5% significance level using the InfoStat/P program (Di Rienzo et al., 2014) and R software (R Core Team, 2013).

3.5. RESULTS

3.5.1. WF estimation

The WF_T of different cropping systems can be seen in Table 5. The MWS-R system had the largest WF_T per hectare (68,862 m^3 ha^{-1} yr^{-1}), which was 23% larger than the MWS system (53,018m^3 ha^{-1} yr^{-1}). By comparison, Sw displayed the lowest WF_T value per hectare (12,735 m^3 ha^{-1} yr^{-1}). When WF_T was expressed on a per liter of ethanol basis, MWS (26.8 m^3 L^{-1} ethanol) was not different from MWS-R (23.1 m^3 L^{-1} ethanol) and Ss (30.9 m^3 L^{-1} ethanol) (Table 5). Sw had the lowest WF_T value per liter of ethanol (3.8 m^3 L^{-1} ethanol). For both functional units (hectare and liter of ethanol), annual systems (MWS, MWS-R, Ss) had larger values than the perennial system (Sw) (Table 5).

In the Sw system, WF_{green} was the main fraction (68%) of WF_T and had the largest value (8,680 m^3 ha^{-1} yr^{-1}). MWS-R and MWS systems were not different from each other, and had a WF_{green} value (7,180 m^3 ha^{-1} yr^{-1}) that was 20% lower than Sw. When WF_{green} was expressed on a per liter of ethanol basis, MWS (3.6 m^3 L^{-1} ethanol) was different from MWS-R (2.4 m^3 L^{-1} ethanol) and Sw (2.6 m^3 L^{-1} ethanol) (Table 5). Overall cropping systems, total loss of N was 30 times higher than P (Table 6). However, WF_{grey} related to P was 13 times higher than WF_{grey} related to pollution by N compounds (Table 5). In annual systems (MWS, MWS-R, Ss), WF_{grey} was the main fraction (87%) of WF_T. MWS-R had the largest WF_{grey} per hectare (61,682 m^3 ha^{-1} yr^{1}), which was 30% larger than MWS (45,838 m^3 ha^{-1} yr^{-1}). The Sw system had the lowest WF_{grey} value per hectare (4,055 m^3 ha^{-1} yr^{-1}).

Table 5: Total water footprint (WF_T),green water footprint (WF_{green}), and grey water footprint (WF_{grey}) per hectare and per ethanol yield of the bioethanol cropping systems [8 year long term experiment (LTE)] located at the Mario Cassinoni Agricultural Experimental Station (EEMAC), Paysandú Department, Uruguay.

	WF_{green}	WF_{grey} (P)	WF_{grey} (N)	WF_T $^{\alpha}$	WF_{green}	WF_{grey} (P)	WF_{grey} (N)	WF_T
Crop Systems	m^3 ha^{-1} yr^{-1}				m^3 L^{-1} ethanol			
MWS	7,180	45,838	2,262	53,018	3.6 a	23.2 a	1.1 b	26.8 ab
MWS-R	7,180	61,682	4,902	68,862	2.4 b	20.7 a	1.6 a	23.1 b
Ss	8,160	42,958	3,468	51,118	4.9 a	25.9 a	2.1 a	30.9 a
Sw	8,680	4,055	1,277	12,735	2.6 b	1.2 b	0.4 c	3.8 c

P = phosphorus, N = nitrogen, MWS = Maize-Wheat-Sorghum, MWS-R = Maize-Wheat-Sorghum, Ss = Sweet Sorghum, and Sw = Switchgrass. Values followed by different letters are significantly different at $P < 0.05$.

$^{\alpha}$: WF_T (m^3 ha^{-1} yr^{-1}) correspond to the sum of WF_{green} and WF_{grey} (P)

Source of nutrient loss affected the WF_{grey} in the bioethanol cropping systems (Table 6). Nutrient loss by leaching was largest in systems with high fertilizations rates (MWS, MWS-R), and nutrient loss by soil erosion was largest in systems with total biomass harvests (MWS-R, Ss). In regard to P, loss by leaching was the most important fraction of WF_{grey} for all systems. On average, this represented 73% of the WF_{grey} within systems. However, in MWS-R and Ss systems having total biomass harvests, P losses by soil erosion represented 38% of the WF_{grey}.

Compared to the MWS system, the higher value of WF_T per hectare in MWS-R corresponded to the largest P loss by soil erosion (0.58 vs 0.18 kg P-PO_4 ha^{-1} yr^{-1}) (Table 6). In addition, the lowest WF_T seen in the Sw system corresponded to P loss (kg P-PO_4 ha^{-1} yr^{-1}) that was 12 times lower than the annual cropping systems.

Table 6: Nutrient loss (P and N) of the different bioethanol cropping systems [8 year long term experiment (LTE)] located at the Mario Cassinoni Agricultural Experimental Station (EEMAC), Paysandú Department, Uruguay.

Crop Systems	P (kg P-PO_4 ha^{-1} yr^{-1})			N (kg N-NO_3 ha^{-1} yr^{-1})		
	Soil Erosion	Leaching	Total	Soil Erosion	Leaching	Total
MWS	0.18	0.96	1.14	10.5	12.1	22.6
MWS-R	0.58	0.96	1.54	36.9	12.1	49.0
Ss	0.35	0.72	1.07	27	7.7	34.7
Sw	0.01	0.09	0.10	3.3	9.5	12.8

P = phosphorus, N = nitrogen, MWS = Maize-Wheat-Sorghum, MWS-R = Maize-Wheat-Sorghum, Ss = Sweet Sorghum, and Sw = Switchgrass.

3.6. DISCUSSION

3.6.1. WF_T of bioethanol systems

Comparing our results with other studies, WF_T of bioethanol cropping systems was observed to be higher than calculated values (Mekonnen and Hoekstra, 2011; Gerbens-Leenes and Hoekstra, 2012; Gerbens-Leenes, 2017). Gerbens-Leenes, (2017) evaluated the total WF_T of different feedstocks (food crops, energy crops, and residues) for bioethanol production. The WF_T (green, blue, and grey) ranged between 1.05 and 3.8 and between 0.15 to 0.7 m^3 L^{-1} ethanol for first- and second-generation from residues, respectively. The WF_T of second-generation bioethanol from energy crops correspond only to WF_{green}, and ranged between 1.5 and 7.4 m^3 L^{-1} ethanol with a mean value of 3.2 m^3 L^{-1} ethanol (Gerbens-Leenes, 2017). In our study, annual crops showed a mean value of 26.9 m^3 L^{-1} ethanol, while the perennial Sw had the smallest value (3.8 m^3 L^{-1} ethanol).

In addition, the WF_T of annual cropping systems (MWS, MWS-R, and Ss) had a different profile to those previously reported (Gerbens-Leenes, 2017; Holmatov et al., 2019; Mathioudakis et al., 2017), where WF_{green} was the main fraction of WF_T (ranging 62%–95% regardless of feedstock). The contribution of WF_{blue} and WF_{grey} varied between crops. On the other hand, the WF of corn and sweet sorghum ethanol feedstocks evaluated by Su et al. (2015) ranged from 25 to 50, 40 to 45, and 10 to 30% for green, blue, and grey, respectively. In our work, WF_{green} and WF_{grey} in annual systems represented 13 and 87% of WF_T, respectively. Unlike other reports, our results highlight the large WF_{grey} component of WF_T. Switchgrass exhibited a profile similar to that reported by others, where green and grey water represent 60 and 40% of WF_T, respectively. The difference in contributions between our results and other reported values could be partially due to lack of a WF_{blue} component in rainfed systems.

When comparing annual and perennial crops, Sw achieved the smallest WF_T per unit of ethanol (m^3 L^{-1} ethanol) and per area (m^3 ha^{-1} yr^{-1}). Our results did not agree with reported values of Gerbens-Leenes (2017), where WF_T of second-generation bioethanol from energy crops (miscanthus, pine, eucalyptus) was similar to WF_T mean values of first generation feedstock. Our results show that this difference is due to the important role of WF_{grey} in WF_T (Table 5). The largest values of WF_{grey} in annual systems correspond to intensive systems that are characterized by a greater number of crops and therefore higher fertilizer application rates. In addition, these systems have the highest rates of soil loss and nutrient loss by erosion. In contrast, Sw is the least intensive system and had a WF_{grey} volume that was 7 times lower than the mean of annual systems.

In reviewed studies (Gerbens-Leenes et al., 2009; Mekonnen and Hoekstra, 2011; Gerbens-Leenes, 2017; Mathioudakis et al., 2017), WF_{green} of bioethanol feedstocks were calculated globally at high spatial resolutions and did not consider climatic and soil variations in regions under study. In our work, the use of specific soil parameters representative of the region (i.e., available water content, maximum infiltration rate, maximum rooting depth, and initial soil moisture depletion) allowed for more accurate WF_{green} values. Given that there are no reported values of WF_{green} for Uruguayan conditions, our results can be considered the first accurate assessment of values for this region.

3.6.2. <u>WF_T changes by crop residue management</u>

Harvest of crop residues (MWS-R) affected the WF_T. Although MWS and MWS-R systems were not statistically different when WF was expressed per unit ethanol produced (m^3 L^{-1} ethanol), WF_T per hectare in the MWS-R system was 30% higher than MWS due to an increased WF_{grey}. The larger WF_{grey} was due to P losses by soil erosion that were 3 times greater (Table 6). Low soil coverage at post-harvest in the MWS-R system led to higher soil erosion levels and therefore larger P losses.

3.6.3. WF_{grey} analyses of evaluated bioethanol systems

Larger WF_{grey} values in bioethanol cropping systems imply that the volume required to assimilate nutrient loads (without regard to other agrochemicals) to meet quality standards plays a significant role and demonstrates the importance of accounting for water pollution and water quality to avoid underestimation. The literature shows that most previous studies did not assess the grey component due to difficulties in evaluating pollutants and integrating real water component volumes (Bocchiola et al., 2013; Mekonnen and Hoekstra, 2015). In cases where WF_{grey} was quantified, only N compounds were considered for water quality restoration (Lovarelli et al., 2016). In agricultural production, WF_{grey} is predominantly due to nutrient load from fertilizers (Liu et al., 2012; Mekonnen and Hoekstra, 2015), and N often has greater impact on WF_{grey} since N fertilizers are more abundantly applied to fields. Although our results showed that N loss was 30 times larger than P (kg ha^{-1} yr^{-1}) overall systems, WF_{grey} related to P was 13 times larger than N (Table 5). The lower permissible limit values for P in potable water production requires larger volumes of freshwater to assimilate nutrient concentrations to established limits.

In our work, the different WF_{grey} values compared to others (Gerbens-Leenes, 2017; Mathioudakis et al., 2017; Holmatov et al., 2019) are largely due to the manner in which WF_{grey} was calculated. In Figure 1, WF_T and WF_{grey} (m^3 ha^{-1} yr^{-1}) of bioethanol cropping systems were presented using our methodology compared to the methodology recommended by Hoekstra et al. (2011). In previously reported values of WF_{grey} in bioethanol cropping systems (Gerbens-Leenes, 2017; Mathioudakis et al., 2017; Holmatov et al., 2019), losses of N by leaching were considered for water quality restoration but P losses by leaching and losses of P and N by soil erosion were not considered.

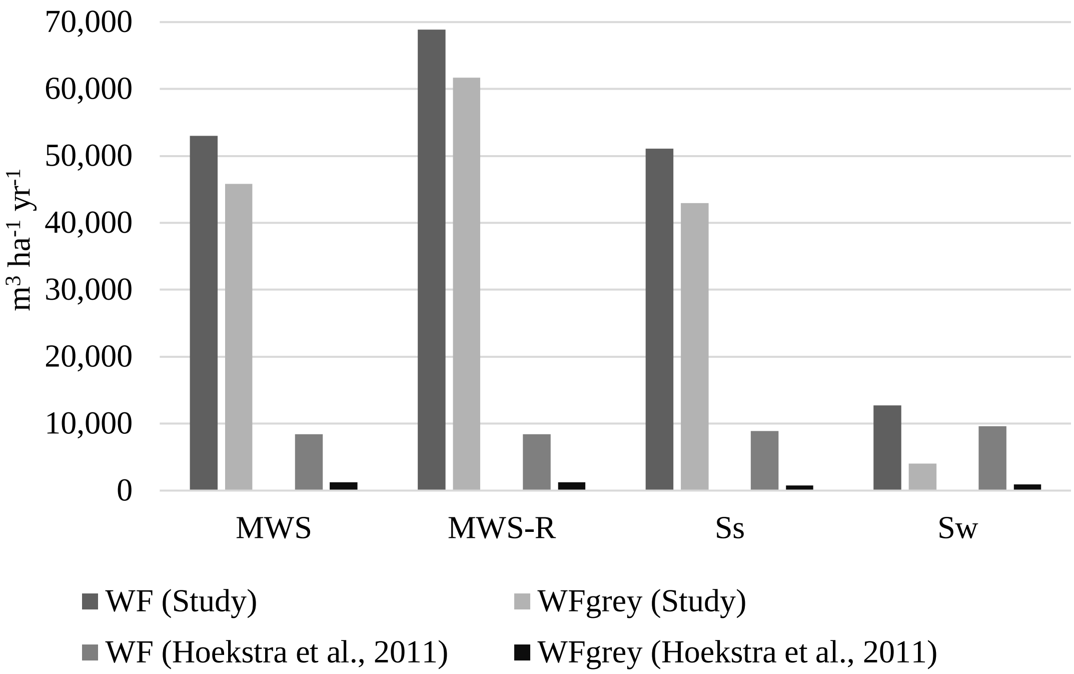

Figure 1. Comparison of total water footprint (WF) and grey water footprint (WFgrey), utilizing our methodology and that of Hoekstra et al. (2011), of the bioethanol cropping systems [8 year long term experiment (LTE)] located at the Mario Cassinoni Agricultural Experimental Station (EEMAC), Paysandú Department, Uruguay. P = phosphorus, N = nitrogen, MWS = Maize-Wheat-Sorghum, MWS-R = Maize-Wheat-Sorghum, Ss = Sweet Sorghum, and Sw = Switchgrass.

Compared to our results, the methodology of Hoekstra et al. (2011) produced an average WF_T that was 6.7 and 1.3 times lower in annual systems and Sw, respectively. This suggests that the methodology in proposed our study detected more differences in annual systems. In addition, the methodology of Hoekstra et al. (2011) not detect important differences in WF_T between bioethanol cropping systems (8,400-9,600 m^3 ha^{-1} yr^{-1}) as were observed using our methodology (12,700-68,800 m^3 ha^{-1} yr^{-1}). The proposed methodology allowed better quantification of N and P losses and showed which nutrient had greater impact on WF_{grey}. Therefore, our methodology can reveal greater impact on water quality in more intensive systems. Another important difference between methodologies was the WF_T profile. On average, WF_{grey} represented 87% of WF_T in annual systems while representing only 13% when methodology of Hoekstra et al. (2011) was applied. In general, WF_T of bioethanol cropping systems and contributions of WF_{grey} were much larger using our proposed methodology.

According to Lovarelli et al. (2016), the methodology proposed by Hoekstra et al. (2011) is over simplified and has limitations for assessing WF_{grey}. When WF is applied to agricultural production, these authors further suggest that WF_{grey} should be calculated following a more detailed methodology and using specific field estimates. In this regard, estimates of pollutants released to water bodies should consider site-specific parameters such as daily precipitation, field slope, soil characteristics (e.g., texture, carbon content), and run-off. In agreement with these authors, our use of site-specific data to estimate WF_{grey} allowed for more accurate values. Furthermore, the use of Erosion 6.0 software with LTE data made it possible to estimate local soil erosion levels and subsequent nutrient losses.

3.6.4. Strengths, limitations, and recommendations

Most WF studies on agricultural products have focused on single crops and only a few studies analyzed cropping systems as a whole (Núñez et al., 2013). Although this approach could be acceptable for perennial crops, annual cropping system cannot be adequately assessed due to multiple operations inherent to crop rotation (e.g., nutrient inputs, soil erosion control, etc.). In this regard, analyzing cropping systems as a whole allowed us to account for different fertilization rates and the initial depletion of soil moisture due to the previous crop. In addition, it was possible to quantify rates of soil loss in the different cropping systems and their impacts on $WF_{T.}$

In regards to WF_{green} estimates, the CROPWAT model is limited in that crop biomass is not considered when determining evapotranspired water values. Therefore, two-year cropping systems (MWS and MWS-R) composed of the same crops had equal values of WF_{green} per hectare despite having different biomass yields. This is a consequence of the model being insensitive to crop yield. Due to low values of WF_{green}

in annual crops, this limitation did not introduce important changes in our study. Nevertheless, this could be detrimental to research estimating WF_{green} for different crops. CROPWAT is not a crop growth model and does not provide real-time scheduling advice, rather it was developed as a tool for estimating crop irrigation requirements (Van Heerden et al., 2008). Intensification of bioethanol cropping systems by increasing harvested biomass (grain and residues) demonstrates the necessity for calibrating models (e.g., specific to Uruguay) that estimate water consumption by unit of product (total biomass). Regarding this, AquaCrop is a water driven model that can simulate annual growing cycle of crops and incorporates the concept of crop water productivity to transform estimated crop evapotranspiration to final crop yield (Steduto et al., 2009). Climate input requirements are very similar to those of CROPWAT, but AquaCrop is a more sophisticated model that requires more detailed soil and crop information (Tsakmakis et al., 2018).

Our results showed that P management is a factor key to reducing WF_{grey}. Thus, strategies that reduce P loss or increase system efficiency will result in a reduction of WF_{grey}. Since Uruguayan soils are characterized by low P availability (Berretta et al., 2000), reductions in P fertilization can lead to lower yields. According to Herath et al. (2014), fertilizer application can be minimized by adopting alternative fertilizer management practices. This requires that farmers are familiar with soil characteristics, especially nutrient content and nutrient requirements at various growth stages, in order to determine optimal timing and fertilizer amount.

Results showed that high rates of soil erosion had a large effect on WF_{grey} values. Therefore, erosion control is a key to reducing WF_{grey} in cropping systems. In fact, results showed that crop residue removal promoted increased WF_T due to increased nutrient loss by soil erosion. To reduce water pollution in bioethanol production systems, removal of annual crop residues is not recommended. On the other hand, cropping systems characterized by high amounts of post-harvest residues will reduce WF_{grey}. In addition, nutrient recycling from decomposing residue can lead to lower fertilization requirements in these cropping systems.

Perennial grasses such as switchgrass (Sw) have been proposed as potential candidates for bioethanol production due to promising productivity and water quality benefits. Switchgrass has high adaptability, high yield, and high nutrient turnover with little carbon debt (Fargione et al., 2008; David and Ragauskas, 2010). To avoid land competition with food crops, Sw is a feasible option for low agriculture suitability soils (Cai et al., 2011) where annual crops have low yields and profitability (Gelfand et al., 2013). Our results showed that Sw had lower WF_T than annual cropping systems under Uruguayan conditions. Moreover, Sw had the largest yields (biomass and ethanol) and the lowest carbon footprint of all studied systems (Bustamante et al. submitted). Therefore, Sw can be a recommended biofuel crop for Uruguay.

3.7. CONCLUSIONS

Our results support the hypothesis that the intensification of agricultural systems resulted in increased WF_T. The most intensified agricultural system (MWS-R) had the highest WF_T per hectare. The least intensive crop (Sw) had the lowest WF_T per hectare and per ethanol yield. The volume required to assimilate P played a significant role in the bioethanol cropping systems. In annual systems, WF_{grey} was the primary component of WF_T. System intensification due to greater number of crops per year and removal of crop residues increased WF_{grey} as a result of large nutrient losses by leaching and soil erosion. In addition, the WF_T of Sw was lowest and corresponded to P losses that were 12 times lower than annual cropping systems. Based on these WF_T results, Sw is the best option for bioethanol crop production.

3.8. REFERENCES

Allen, R.G., Pereira, L.S., Raes, D., Smith, M., others, 1998. Crop evapotranspiration-Guidelines for computing crop water requirements-FAO Irrigation and drainage paper 56. Fao, Rome 300, D05109.

Berretta, E., Risso, D., Montossi, F., Pigurina, G., 2000. Campos in Uruguay. Grassl. Ecophysiol. grazing Ecol. Ed. G Lemaire, J Hogdson, A Moraes, C Nabinger, PC Carvalho. New York, USA. CABI Publishing. 377–394.

Bocchiola, D., Nana, E., Soncini, A., 2013. Impact of climate change scenarios on crop yield and water footprint of maize in the Po valley of Italy. Agric. Water Manag. 116, 50–61.

Bustamante, M., Siri G., Carrasco-Letelier, L. Unpublished results. Carbon footprint of four bioethanol cropping systems in Uruguay.

Cai, X., Zhang, X., Wang, D., 2011. Land availability for biofuel production. Environ. Sci. Technol. https://doi.org/10.1021/es103338e

Chapagain, A.K., Hoekstra, A.Y., 2004. Water footprints of nations.

Chapagain, A.K., Hoekstra, A.Y., Savenije, H.H.G., Gautam, R., 2006. The water footprint of cotton consumption: An assessment of the impact of worldwide consumption of cotton products on the water resources in the cotton producing countries. Ecol. Econ. 60, 186–203.

Chukalla, A.D., Krol, M.S., Hoekstra, A.Y., 2018. Grey water footprint reduction in irrigated crop production: Effect of nitrogen application rate, nitrogen form, tillage practice and irrigation strategy. Hydrol. Earth Syst. Sci. https://doi.org/10.5194/hess-22-3245-2018

Custodio, E., Llamas, M.R., 1976. Hidrologia Subterránea. Tomo I, Ed. Omega, Barcelona.

David, K., Ragauskas, A.J., 2010. Switchgrass as an energy crop for biofuel production: A review of its ligno-cellulosic chemical properties. Energy Environ. Sci. 3, 1182-1190. https://doi.org/10.1039/b926617h

Di Rienzo, J.A., Casanoves, F., Balzarini, M.G., Gonzalez, L., Tablada, M., Robledo, C.W., 2014. InfoStat. Grup. Infoestat, FCA, Univ. Nac. Córdoba, Argentina.

DOE, 2009. Theoretical Ethanol Yield Calculator. US Department of Energy. Consultado 15 marzo 2018. Disponible en: www.eere.energy.gov/biomass/ethanol yield calculator.html

Dominguez-Faus, R., Powers, S.E., Burken, J.G., Alvarez, P.J., 2009. The Water Footprint of Biofuels: A Drink or Drive Issue? Environ. Sci. Technol. 43, 3005–3010. https://doi.org/10.1021/es802162x

Durán, A., García Préchac, F., 2007. Suelos del Uruguay: Origen, clasificación, manejo y conservación. Editorial Hemisferio Sur, Montevideo, Uruguay.

FAO, 2010. CROPWAT 8.0 for windows user guide. Rome, Italy.

Fargione, J., Hill, J., Tilman, D., Polasky, S., Hawthorne, P., 2008. Land clearing and the biofuel carbon debt. Science. https://doi.org/10.1126/science.1152747

Foster, G.R., McCool, D.K., Renard, K.G., Moldenhauer, W.C., 1981. Conversion of the universal soil loss equation to SI metric units. J. Soil Water Conserv. 36, 355–359.

García-Préchac, F., Hill, M., Clericí, C., 2013. Erosión: modelo de estimación de erosión de suelos en Uruguay y región sur de la Cuenca del Plata (programa informático). Dep. Suelos y Aguas, Fac. Agron. Univ. la República-Ministerio Ganad. Agric. y Pesca-Banco Mundial, Montevideo, Uruguay.

Gelfand, I., Sahajpal, R., Zhang, X., Izaurralde, R.C., Gross, K.L., Robertson, G.P., 2013. Sustainable bioenergy production from marginal lands in the US Midwest. Nature 493, 514–517. https://doi.org/10.1038/nature11811

Gerbens-Leenes, W., Hoekstra, A.Y., van der Meer, T.H., 2009. The water footprint of bioenergy. Proc. Natl. Acad. Sci. 106, 10219–10223.

Gerbens-Leenes, W., Hoekstra, A.Y., 2012. The water footprint of sweeteners and bio-ethanol. Environ. Int. 40, 202-211. https://doi.org/10.1016/j.envint.2011.06.006

Gerbens-Leenes, P.W., 2017. Bioenergy water footprints, comparing first, second and third generation feedstocks for bioenergy supply in 2040. Eur. Water. 59, 373-380. https://doi.org/10.1073/pnas.0812619106

Gnansounou, E., Dauriat, A., Wyman, C. E, 2005. Refining sweet sorghum to ethanol and sugar: economic trade-offs in the context of North China. Bioresource technology. 96, 985-1002. https://doi.org/10.1016/j.biortech.2004.09.015

Halvorsen, K.E., Barnes, J.R., Solomon, B.D., 2009. Upper Midwestern USA Ethanol Potential from Cellulosic Materials. Soc. Nat. Resour. 22, 931–938. https://doi.org/10.1080/08941920902755382

Herath, I., Green, S., Horne, D., Singh, R., Clothier, B., 2014. Quantifying and reducing the water footprint of rain-fed potato production, part I: Measuring the net use of blue and green water. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2014.06.026

Hoekstra, A.Y., Chapagain, A.K., Mekonnen, M.M., Aldaya, M.M., 2011. The water footprint assessment manual: Setting the global standard. London: Routledge.

Holmatov, B., Hoekstra, A.Y., Krol, M.S., 2019. Land, water and carbon footprints of circular bioenergy production systems. Renew. Sustain. Energy Rev. https://doi.org/10.1016/j.rser.2019.04.085

IEA, 2018. Market Analysis and Forecast from 2018 to 2023. International Energy Agency.

Ji, X., Long, X., 2016. A review of the ecological and socioeconomic effects of biofuel and energy policy recommendations. Renew. Sustain. Energy Rev. https://doi.org/10.1016/j.rser.2016.03.026

Khan, M.N., Mohammad, F., 2014. Eutrophication: Challenges and solutions, in: Eutrophication: Causes, Consequences and Control. London. Springer, Dordrecht. 1-15.

Kottek, M., Grieser, J., Beck, C., Rudolf, B., Rubel, F., 2006. World map of the Köppen-Geiger climate classification updated. Meteorol. Zeitschrift. 15, 259-263.

Liu, C., Kroeze, C., Hoekstra, A.Y., Gerbens-Leenes, W., 2012. Past and future trends in grey water footprints of anthropogenic nitrogen and phosphorus inputs to major world rivers. Ecol. Indic. 18, 42-49. https://doi.org/10.1016/j.ecolind.2011.10.005

Lovarelli, D., Bacenetti, J., Fiala, M., 2016. Water Footprint of crop productions: A review. Sci. Total Environ. 548, 236-251. https://doi.org/10.1016/j.scitotenv.2016.01.022

Ludeña, C.E., Ryfisch, D., 2015. Uruguay: Mitigación y adaptación al cambio climático. Banco Interamericano de Desarrollo. División de Cambio Climático y Sostenibilidad, Washington D.C. Nota Técnica. 860, 1-28.

Mathioudakis, V., Gerbens-Leenes, P.W., Van der Meer, T.H., Hoekstra, A.Y., 2017. The water footprint of second-generation bioenergy: A comparison of biomass feedstocks and conversion techniques. J. Clean. Prod. 148, 571–582. https://doi.org/10.1016/J.JCLEPRO.2017.02.032

McLauhlin, J., Bouton, D., Conger, B., Ocumpaugh, W., Parrish, D., Taliaferro, C., Vogel, K., Wullschleger, S. 1999. Developing Switchgrass as a Bioenergy Crop. J. Janick (ed.), ASHS Press, Alexandria, VA.

Mekonnen, M.M., Hoekstra, A.Y., 2011. The green, blue and grey water footprint of crops and derived crop products. Hydrol. Earth Syst. Sci. 15, 1577–1600. https://doi.org/10.5194/hess-15-1577-2011

Mekonnen, M.M., Hoekstra, A.Y., 2015. Global Gray Water Footprint and Water Pollution Levels Related to Anthropogenic Nitrogen Loads to Fresh Water. Environ. Sci. Technol. 49, 12860-12868. https://doi.org/10.1021/acs.est.5b03191

Morales, M., Quintero, J., Conejeros, R., Aroca, G., 2015. Life cycle assessment of lignocellulosic bioethanol: Environmental impacts and energy balance. Renew. Sustain. Energy Rev. 42, 1349–1361. https://doi.org/10.1016/J.RSER.2014.10.097

MVOTMA (Ministerio de Vivienda, Ordenamiento Territorial y Medio Ambiente). 1979. Decree 253/79. Código de aguas (p. 14). Montevideo: MVOTMA.

Núñez, M., Pfister, S., Antón, A., Muñoz, P., Hellweg, S., Koehler, A., Rieradevall, J., 2013. Assessing the Environmental Impact of Water Consumption by Energy Crops Grown in Spain. J. Ind. Ecol. 17, 90-102. https://doi.org/10.1111/j.1530-9290.2011.00449.x

R Core Team, 2013. R: A Language and Environment for Statistical Computing.

Scherer, L., Pfister, S. 2015. Modelling spatially explicit impacts from phosphorus emissions in agriculture. Int J Life Cycle Assess 20, 785–795. https://doi.org/10.1007/s11367-015-0880-0

Silva, A., de León, J., García, F., Durán, A., 1988. Aspectos metodológicos en la determinación de la capacidad de retener agua de los suelos del Uruguay. Universidad de la Republica, Facultad de Agronomia.

Siri-Prieto, G., 2013. Bioenergía en el Uruguay ¿cómo hacerla productiva y sustentable? In: Simposio Nacional de Agricultura (3o, 2013, Paysandú, Uruguay). No Se Llega, Si No Se Sabe a Dónde Ir. Pensar En Las Causas y No Sólo Medir Las Consecuencias. Montevideo, Uruguay, Hemisferio Sur. 21-38.

Steduto, P., Hsiao, T.C., Raes, D., Fereres, E., 2009. AquaCrop the FAO crop model to predict yield response to water. I Concepts. Special issue on “Yield Response to Water: Examination of the Role of Crop Models in Predicting Water Use Efficiency”. Agron. J. 101, 426-437. https://doi.org/10.2134/agronj2008.0139s

Su, M.H., Huang, C.H., Li, W.Y., Tso, C.T., Lur, H.S., 2015. Water footprint analysis of bioethanol energy crops in Taiwan. J. Clean. Prod. 88, 132–138. https://doi.org/10.1016/j.jclepro.2014.06.020

Tsakmakis, I.D., Zoidou, M., Gikas, G.D., Sylaios, G.K., 2018. Impact of Irrigation Technologies and Strategies on Cotton Water Footprint Using AquaCrop and CROPWAT Models. Environ. Process. 5, 181-199. https://doi.org/10.1007/s40710-018-0289-4

Uruguay XXI. 2017. Energías renovables: Oportunidades de inversión. Montevideo. Recuperado de: https://www.uruguayxxi.gub.uy/uploads/informacion/Informe%20de%20Energ%C3%A Das%20Renovables%20-%20Setiembre%20de%202017%20-%20Uruguay%20XXI9.pdf. [Consultado: 8/12/2019]

USDA, 1999. Soil Taxonomy: A Basic System of Soil Classification for Making and Interpreting Soil Surveys (Second Edition), Agriculture Handbook Number 436.

Van Heerden, P.S., Crosby, C.T., Grove, B., Benade, N., Theron, E., Schulze, R.E., Tewolde, M.H., 2008. Integrating and upgrading of SAPWAT and PLANWAT to create a powerful and user-friendly irrigation water planning tool. Water Res. Comm. Rep. number TT391/08, Pretoria.

Wu, M., Chiu, Y., Demissie, Y., 2012. Quantifying the regional water footprint of biofuel production by incorporating hydrologic modeling. Water Resour. Res. https://doi.org/10.1029/2011WR011809

Zhong, J., Yu, T.E., Clark, C.D., English, B.C., Larson, J.A., Cheng, C.L., 2018. Effect of land use change for bioenergy production on feedstock cost and water quality. Appl. Energy. https://doi.org/10.1016/j.apenergy.2017.09.070

4. DISCUSIÓN GENERAL Y CONCLUSIONES

Desde el año 2011 en Uruguay se había comenzado a implementar la metodología LCA, principalmente en los principales rubros de la producción agropecuaria (ganadería, agricultura y lechería). Sin embargo, a pesar de la creciente participación de la biomasa en la producción energética nacional, 41% de la matriz primaria (MIEM, 2018), se habían realizado escasos trabajos relacionados a la estimación de huellas ambientales en sistemas de cultivos bioenergéticos. Además, los pocos estudios realizados se centraban en cultivos únicos, y no analizaban sistemas de cultivo en su conjunto. Como lo mencionan Nemecek et al. (2015) este enfoque no es adecuado para cultivos anuales porque cada sistema de cultivo se ve afectado por las operaciones a lo largo de todo el sistema de rotación de cultivos (por ejemplo, control de malezas, aportes de nutrientes, control de erosión, etc.). Por tanto, este trabajo se considera pionero a nivel nacional no solo en la estimación de huella de carbono y huella hídrica de cultivos bioenergéticos; sino, además, porque evalúa sistemas de cultivos teniendo en cuenta todo el sistema de rotación. Además, si bien los sistemas de cultivos evaluados tienen como fin la producción de biomasa para producir etanol, los resultados obtenidos son de utilidad para aquellos sistemas cuyo objetivo son la producción de grano para alimento animal y/o humano. Es decir, puede ser un aporte metodológico y cuantitativo relevante para cadenas de valor de la carne o de producción de cereales.

Esta investigación ha generado información original de utilidad para todos los actores relacionados a la producción y comercialización de biomasa, aportando información importante para lograr una intensificación ecológica y/o sostenible (Tittonell, 2014) de los sistemas bioenergéticos del país. Para dar respuesta a estas preguntas, fueron estimadas la huella de carbono y huella hídrica de cuatro sistemas de producción de biomasa con diferente grado de intensificación. Es relevante que la información generada sea fácilmente comprendida y pueda ser empleada por los tomadores de decisión, por ejemplo, consultores, técnicos, políticos y agentes de los ministerios, entre otros.

La inexistencia de una línea base en sistemas de producción de biomasa para la comparación constituyó una limitante al momento de testear los resultados. Sin embargo la comparación con trabajos internacionales, por ejemplo, Cherubini y Jungmeier (2010), Fazio y Monti (2011) y Gerbens-Leenes (2017), contribuyó de manera importante para contrastar la situación nacional con el contexto internacional. En base a lo anterior, este trabajo si bien no pretende ser una estimación nacional de la huella de carbono y huella hídrica de sistemas bioenergéticos del Uruguay, realiza un aporte importante en la evaluación de las emisiones de GEI y uso y contaminación de agua, contribuyendo a la generación de nuevas hipótesis de trabajo que permitan implementar medidas de mitigación de las emisiones de GEI y disminución de la contaminación de los recursos hídricos.

A nivel nacional la intensificación de los sistemas agrícolas tiene como resultado un aumento de las huellas ambientales evaluadas (HC y HH). El sistema agrícola más intensificado (MWS-R) tuvo la mayor HC y HH por hectárea mientas que el cultivo menos intensivo (Sw) tuvo la menor HC y HH por hectárea y por rendimiento de etanol. Los resultados indican que la intensificación mediante el retiro de residuos (MWS-R) aumenta las emisiones de GEI por hectárea un 60% e incrementa la HH por hectárea 30% con respecto al sistema sin retiro (MWS). En nuestro estudio, el retiro de residuos causó una reducción del COS de 620 kg de C ha^{-1}, en comparación con el no retiro de residuos de cultivos. Estos valores son mayores que los reportados (Gabrielle y Gagnaire, 2008) ya que en nuestro escenario también consideramos la pérdida de C por erosión. Con respecto al incremento de la HH, se debe a un mayor valor de agua gris dado por mayores pérdidas de nutrientes por erosión. Una baja cobertura del suelo después del retiro de residuos conduce a mayores niveles de erosión del suelo y, por lo tanto, a mayores pérdidas de nutrientes.

La intensificación debido a una mayor cantidad de cultivos por año incremento las emisiones GEI y la HH de los sistemas, dado principalmente por un incremento del uso de fertilizantes. En lo que refiere a la HC, el uso de fertilizantes representó un 36%

de las emisiones totales de GEI. Este resultado coincidió con otros estudios donde el fertilizante nitrogenado fue uno de los insumos más costosos, en términos de emisiones de GEI (Monti et al., 2012; Morales et al., 2015). Con respecto a la HH, el uso de fertilizantes incrementó el agua gris como resultado de mayores pérdidas de nutrientes por lixiviación. La literatura mostró que la mayoría de estudios consideraron solo el N para la estimación del agua gris, sin considerar otros nutrientes como el P (Lovarelli et al., 2016). Aunque nuestros resultados mostraron que la pérdida de N fue 30 veces mayor que las pérdidas de P (kg ha^{-1} $año^{-1}$), el agua gris relacionado con P fue 13 veces mayor que N. Los menores valores permisibles para P en la producción de agua potable requieren mayores volúmenes de agua para diluir las exportaciones de este nutriente.

En base a estos resultados, en coincidencia con Hillier et al. (2009), es posible definir al manejo de los nutrientes como un factor clave para reducir las emisiones GEI y los impactos en la calidad del agua. Cualquier estrategia que permita reducir el uso de fertilizantes y/o aumentar su eficiencia dará como resultado una reducción de las emisiones de GEI y una menor contaminación de los recursos hídricos. Sin embargo, los suelos uruguayos naturalmente se caracterizan por una baja disponibilidad de P (Berretta et al., 2000), y la capacidad del suelo para aportar nitrógeno está disminuyendo (Morón et al., 2012). Por lo cual, una reducción de la fertilización conduciría a menores rendimientos y un incremento de las huellas estimadas. Entre algunas medidas para mitigar estos efectos, la inclusión de una fase de pasturas con componentes gramíneas y leguminosas en los sistemas de cultivo agrícola para aumentar el secuestro de C en el suelo y promover la fijación biológica de nitrógeno podría ayudar a reducir los requisitos de fertilizante nitrogenado (Köpke y Nemecek, 2010). De acuerdo con Herath et al. (2014), la aplicación de fertilizantes puede minimizarse adoptando prácticas de manejo de fertilizantes, por ejemplo, análisis de suelos previo a fertilizar, ajuste de fertilización según requerimientos del cultivo, aplicación de enmiendas orgánicas, etc. Esto requiere que los agricultores estén familiarizados con las características del suelo, especialmente el contenido de nutrientes y los requisitos de nutrientes en varias etapas de crecimiento, a fin de determinar el momento óptimo y la cantidad de fertilizante.

Finalmente, se debe destacar la importancia de haber considerado los procesos de mineralización/secuestro de C por cambios en el uso de suelo (LUC) en el balance de GEI así como las pérdidas de C y nutrientes por erosión del suelo. Esta consideración cumple con el reciente Reglamento 2018/841 de la directiva de la Unión Europea sobre Energías Renovables (RED) (2009/28/EC) el cual determina la necesidad de cuantificar las emisiones respecto a LUC y la pérdida de erosión del suelo en el balance de GEI en los estudios de LCA. Por lo tanto, este trabajo se adelantó al protocolo de cálculo recién acordado por la UE. Estas fuentes afectaron significativamente las emisiones globales de GEI de los sistemas de cultivo evaluados. En el sistema MWS-R y Ss con una cosecha total de biomasa, el COS perdido por la erosión del suelo y LUC correspondió al 37 y 20%, respectivamente, de las emisiones totales de GEI de estos sistemas. Además, en el sistema MWS-R, la pérdida de COS fue 70% mayor que el sistema MWS. La baja HC de Sw encontrada en este trabajo se debe en gran parte a una tasa anual de secuestro de C de 1,47 Mg $CO_{2\text{-eq}}$ ha^{-1}, un valor que corresponde al 70% de las emisiones netas totales de este sistema. Al igual que Schmer et al. (2015) y Cherubini et al. (2009) este trabajo muestra que la cuantificación de los cambios en el almacenamiento de COS y erosión son factores importantes para estimar con precisión las emisiones de GEI en las evaluaciones LCA de bioenergía. A pesar de no existir una metodología común para considerar los cambios de SOC en LCA, los modelos utilizados en este trabajo, AMG para cambios en el uso del suelo y EROSION 6.0 para cuantificar la erosión, mostraron ser modelos válidos para permitir cuantificar las pérdidas y/o ganancias de C de los sistemas. Esto como consecuencia de ser modelos calibrados para las condiciones de Uruguay y utilizar datos sitio-específico de suelo y clima representativos de la región.

4.1. COMENTARIOS PARA INVESTIGACIONES FUTURAS

A pesar de los avances logrados en la estimación de las huellas ambientales de sistemas bioenergéticos, de las fuentes que explican las emisiones de GEI y de las pérdidas de nutrientes que llevan a altos valores de agua gris, numerosos aspectos de interés no han sido totalmente dilucidados y al mismo tiempo surgen nuevos interrogantes a resolver en investigaciones futuras. Algunos de estos aspectos se plantean a continuación.

Trabajar con modelos de simulación implica conocer el funcionamiento de los cultivos y que la herramienta a emplear esté calibrada, con el fin de representar de la manera más fiel posible el crecimiento y desarrollo de los cultivos a estudiar. Pero además, requiere de utilizar la información más exacta posible de los sistemas que se van a simular, para que estas simulaciones sean representativas de la realidad. En éste trabajo la utilización del modelo CROPWAT para la estimación del agua verde mostró que dicho modelo es limitado ya que no considera diferencias en rendimiento de biomasa para determinar los valores de agua evapotranspirada. Esto es consecuencia de que el modelo es insensible al rendimiento del cultivo. CROPWAT no es un modelo de crecimiento de cultivos y no proporciona asesoramiento de programación en tiempo real, sino que se desarrolló como una herramienta para estimar los requisitos de riego de cultivos (Van Heerden et al., 2008). La intensificación de los sistemas de cultivo de bioetanol al aumentar la biomasa cosechada (grano y residuos) demuestra la necesidad de calibrar modelos para Uruguay que estimen el consumo de agua por unidad de producto (biomasa total). Con respecto a esto, AquaCrop es un modelo que puede simular el ciclo anual de crecimiento de los cultivos e incorpora el concepto de productividad del agua del cultivo para transformar la evapotranspiración estimada del cultivo en el rendimiento final del cultivo (Steduto et al., 2009). Los requisitos de entrada climática son muy similares a los de CROPWAT, pero AquaCrop es un modelo más sofisticado que requiere información más detallada sobre el suelo y los cultivos (Tsakmakis et al., 2018).

Los resultados mostraron que el proceso de secuestro de C jugó un papel importante en el balance de GEI, principalmente en el sistema de Sw donde el secuestro de C representó el 70% de las emisiones del sistema. Sin embargo, el valor de secuestro utilizado para switchgrass de 0,4 Mg (ha año)$^{-1}$ fue tomado de bibliografía (Anderson-Teixeira et al., 2009) dado que en Uruguay, existe un vacío de información acerca de las tasas de secuestro de C de los cultivos perennes. Si bien existe basta información internacional, no existe información públicamente disponible que permita conocer las tasas de secuestro bajo las condiciones de suelo y clima del país. Respecto a esto, actualmente hay instalados ensayos de largo plazo cuyo objetivo es evaluar el efecto de distintos cultivos perennes sobre la calidad de suelo (Siri-Prieto, 2013). Por tanto, nuestro trabajo puede ser fácilmente actualizado una vez que esta información esté disponible. Pero creemos importante resaltar, la necesidad de conocer con mayor precisión las tasas de secuestro de C de cultivos perennes como switchgrass.

La estimación de huellas ambientales (carbono y agua), aplicando metodologías ampliamente utilizadas a nivel mundial condujo al uso de modelos y base de datos con información pertenecientes a otras regiones del mundo. Así, para la estimación de las emisiones de GEI, se utilizó la base de datos de USDA, la cual posee valores que no sabemos cuan representativos son para esta región. Por lo que sería deseable el desarrollo de una base de datos representativa de la región como el MERCOSUR, para un cálculo más exacto. Información más representativa con datos de emisión locales, puede permitir estimar con mayor precisión las emisiones de los sistemas evaluados. Una vez que esta información se encuentre disponible, puede ser empleada para actualizar este trabajo en la medida de que implique una mejora de las estimaciones de las huellas ambientales de sistemas bioenergéticos de nuestro país.

Este trabajo de investigación es un aporte metodológico para lograr la intensificación ecológica de sistemas bioenergéticos, ya que provee información local aplicando metodología transparente y ampliamente reconocida, de las huellas ambientales de carbono y agua. Este aporte habilita la evaluación de otras dimensiones del impacto ambiental como: huella de nutrientes, huella de biodiversidad, ecotoxicidad, acidificación, agotamiento de recursos. Las herramientas para aplicar este tipo de análisis se encuentran disponibles y creemos que indudablemente es un paso que debe ser dado en el menor tiempo posible.

Finalmente, creemos que la información generada con el propósito de contribuir a la mejora de los sistemas agrícolas de nuestro país, informando a todos los actores comprometidos con la producción, todos estos esfuerzos son en vano si no logramos que sea efectivamente transmitida a la sociedad. Hoy sobran las herramientas para acortar distancias entre las universidades y los centros de investigación y los tomadores de decisiones, por lo tanto, debemos hacer el mejor uso posible de las mismas para que el conocimiento generado llegue a los diferentes actores y por ende repercuta positivamente en la producción nacional.

4.2. RECOMENDACIONES Y SUGERENCIAS

Si bien el objetivo principal de la tesis era de carácter científico tecnológico, aportando una primera estimación de las huellas ambientales de carbono y agua de sistemas de cultivos bioenergéticos, es de suma importancia utilizar dichos valores para incidir en una toma de decisiones sobre que sistemas bioenergéticos sería conveniente utilizar a nivel nacional. Utilizando los criterios de sostenibilidad de la Directiva Europea de Energía Renovable (EC Directive, 2009) sobre los biocombustibles, en lograr una reducción del 60% en GEI en relación con los combustibles fósiles, solamente Sw cumpliría con estos requisitos dado que presenta una disminución de 90% de emisiones GEI. Los sistemas de cultivos anuales (MWS y MWS-R) solo generaron reducciones de 9,3 y 3,4%, respectivamente en comparación con los combustibles fósiles, lo que los hace no aptos para la producción sostenible de biocombustibles. Además, Ss tuvo el peor desempeño al generar un incremento en las emisiones de GEI del 60% en comparación con los combustibles fósiles.

La utilización de una base de datos de 8 años de rendimiento permitió considerar la variación meteorológica anual la cual produjo diferentes rendimientos en cada sistema y, por lo tanto, huellas ambientales específicas cuando estas se expresaron por unidad de producto (rendimiento en etanol). Los sistemas de cultivos anuales mostraron mayores valores de HC y HH dado que los rendimientos generales fueron más bajos en comparación con Sw, lo que condujo a aumentos significativos en las huellas evaluadas. Esto es más importante en años donde las condiciones climáticas son adversas para el crecimiento de los cultivos. Switchgrass tuvo una mayor variabilidad en el rendimiento de 4,2 a 30,4 Mg (ha^{-1} $año^{-1}$), sin embargo, las emisiones de Sw tuvieron un patrón más estable. Si se compara con los criterios de sostenibilidad del Reglamento 2009/28/EC (GEI igual o al menos inferior al 40% de las emisiones de gasolina), solo Sw puede satisfacer esta condición en todas las condiciones meteorológicas ocurridas. Además, dada la gran variabilidad climática a la que están expuestos los cultivos, Sw sería el cultivo más recomendado para mantener bajas emisiones de GEI y valores de HH.

Las gramíneas perennes como switchgrass (Sw) se han propuesto como posibles candidatos para la producción de bioetanol debido a los beneficios prometedores de productividad y calidad del agua. En comparación con los sistemas de cultivos anuales, Sw mostró las emisiones más bajas de GEI y la menor HH. Además, Sw aumentó el secuestro de C del suelo y presentó niveles de erosión muy por debajo de la tolerancia de perdida de suelo. En este contexto, se pueden hacer dos sugerencias: en primer lugar, Sw es una opción factible para suelos con baja aptitud agrícola (Fritsche et al., 2010; Cai et al., 2011) donde los cultivos anuales tienen bajo rendimiento y rentabilidad de forma de evitar la competencia con cultivos para alimento (Gelfand et al., 2013). Sin embargo, dado que los suelos en nuestro estudio son de aptitud agrícola media, los resultados de este estudio probablemente tengan un rendimiento más alto que los posibles en suelos con baja aptitud agrícola. Por estas razones, la productividad de Sw debería estudiarse en condiciones de baja aptitud agrícola. Segundo, Sw podría ser una buena herramienta para mejorar las propiedades físicas y químicas (Anderson-Teixeira et al., 2009; Monti et al., 2012) de los suelos degradados por la intensificación de la agricultura, según lo descrito por Beretta-Blanco et al. (2019) y Carrasco-Letelier y Beretta-Blanco (2017).

En cuanto a los sistemas de cultivos anuales, de la forma en que están planteados en este trabajo no serían recomendables para ser utilizados como sistemas de producción de biomasa para la producción de etanol. Sin embargo, sería recomendable evaluar estos sistemas con algunos cambios que permitan mejorar el desempeño ambiental de los mismos. Desde un punto de vista agronómico la inclusión de una fase de pasturas en los sistemas de rotación permitiría incorporar el secuestro de C en el suelo mediante las especies gramíneas y promover la fijación biológica de N mediante las especies leguminosas. Esto permitiría mejorar el balance de C de los sistemas y además disminuir la utilización de fertilizantes sintéticos durante la fase de cultivos.

Los resultados de este trabajo constituyen una primera evaluación de la relación entre la intensidad agrícola y las emisiones de GEI y huella hídrica en sistemas bioenergéticos en el Uruguay. Pese a que uno de los desafíos primordiales de la producción agrícola es incrementar la rentabilidad económica de los sistemas, la respuesta debe estar dirigida hacia sistemas más sustentables. Las influencias de las políticas ambientales son dignas de atención en Uruguay a pesar de que todavía no es el foco principal en el país, por tal motivo el desarrollo de herramientas para estimar impacto ambiental, controlar el progreso y la eficacia de las opciones de mitigación sería un acierto que generaría ventajas considerando la aparición de futura regulación.

5. BIBLIOGRAFÍA

Anderson-Teixeira KJ, Davis SC, Masters MD, Delucia EH. 2009. Changes in soil organic carbon under biofuel crops. GCB Bioenergy, 1(1): 75-96. https://doi.org/10.1111/j.1757-1707.2008.01001.x.

Arodudu OT, Helming K, Voinov A, Wiggering H. 2017. Integrating agronomic factors into energy efficiency assessment of agro-bioenergy production - A case study of ethanol and biogas production from maize feedstock. Applied Energy, 198: 426–439. https://doi.org/10.1016/J.APENERGY.2017.02.017

Becoña G, Astigarraga L, Picasso VD. 2014. Greenhouse gas emissions of beef cow–calf grazing systems in Uruguay. Sustainable Agriculture Research, 3(2): 89-105. http://dx.doi.org/10.5539/sar.v3n2p89.

Becoña G, Oyantcabal W, Astigarraga L, Roel A, Saizar C. 2013. Primer Estudio de la Huella de Carbono de Tres Cadenas Agroexportadoras del Uruguay: Carne vacuna, Lácteos y Arroz. Informe Final. Ministerio de Ganadería Agricultura y Pesca, Facultad de Agronomía, Instituto Nacional de Investigación Agropecuaria, Laboratorio Tecnológico del Uruguay. Montevideo. 54 p.

Becoña G. 2012. Emisiones de gases de efecto invernadero en sistemas de cría vacuna del Uruguay. Tesis de maestría. Montevideo, Uruguay. Facultad de Agronomía. 69 p.

Beretta-Blanco A, Pérez O, Carrasco-Letelier L. 2019. Soil quality decrease over 13 years of agricultural production. Nutrient Cycling in Agroecosystems, 114(1): 45-55. https://doi.org/10.1007/s10705-019-09990-3.

Berretta E, Risso D, Montossi F, Pigurina G. 2000. Campos in Uruguay. In: Grassland Ecophysiology and Grazing Ecology. Ed. Lemaire G, Hogdson J, Moraes A, Nabinger C, Carvalho PC. New York, USA. CABI Publishing. 377–394.

Brandão M, Milà i Canals L, Clift R. 2011. Soil organic carbon changes in the cultivation of energy crops: Implications for GHG balances and soil quality for use in LCA. Biomass and Bioenergy, 35(6): 2323-2336. https://doi.org/10.1016/j.biombioe.2009.10.019.

Cai X, Zhang X, Wang D. 2011. Land availability for biofuel production. Environmental Science & Technology, 45(1): 334-339. https://doi.org/10.1021/es103338e.

Carrasco-Letelier L, Beretta-Blanco A. 2017. Soil erosion by water estimated for 99 Uruguayan basins. Ciencia e Investigación Agraria, 44(2): 184-194. https://doi.org/10.7764/rcia.v44i2.1717.

Carrasco-Letelier L, Blanco AB, Eguren G. 2014. Primer mapa nacional de la calidad del agua de Uruguay. Montevideo, Uruguay. Revista INIA, 39: 67-70.

Carrasco-Letelier L, Vázquez D, D'Ottone F, Resquín F, Scoz R. Vilaró F, Rodríguez, G, Vicente E, Terra J. 2013. Balance energético de cadenas agro-industriales de interés para la producción de bioenergías. Montevideo, Uruguay. Revista INIA, 32: 46-50.

Cherubini F, Jungmeier G. 2010. LCA of a biorefinery concept producing bioethanol, bioenergy, and chemicals from switchgrass. International Journal of Life Cycle Assessment, 15(1): 53-66. https://doi.org/10.1007/s11367-009-0124-2.

Cherubini F, Bird ND, Cowie A, Jungmeier G, Schlamadinger B, Woess-Gallasch, S. 2009. Energy- and greenhouse gas-based LCA of biofuel and bioenergy systems: Key issues, ranges and recommendations. Resources, Conservation & Recycling, 53(8): 434-447. https://doi.org/10.1016/j.resconrec.2009.03.013

Darré E, Cadenazzi M, Mazzilli SR, Rosas JF, Picasso VD. 2019. Environmental impacts on water resources from summer crops in rainfed and irrigated systems. Journal of Environmental Management, 232: 514-522.

EC Directive. 2009. Directive 2009/28/EC of The European Parliament and of The Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing directives 2001/77/EC and 2003/30/EC. Official Journal of the European Union.

EU Regulation. 2018. Regulation 2018/841 of the European parliament and of the council of 30 May 2018 on the inclusion of greenhouse gas emissions and removals from land use, land use change and forestry in the 2030 climate and energy framework, and amending Regulation (EU) No 525/2013 and

Decision No 529/2013/EU. Official Journal of the European Union 19 June 2018, L 156/1–25.

Fazio S, Monti A. 2011. Life cycle assessment of different bioenergy production systems including perennial and annual crops. Biomass and Bioenergy, 35(12): 4868–4878. https://doi.org/10.1016/J.BIOMBIOE.2011.10.014.

Follett RF, Vogel KP, Varvel GE, Mitchell RB, Kimble J. 2012. Soil Carbon Sequestration by Switchgrass and No-Till Maize Grown for Bioenergy. BioEnergy Research, 5(4): 866–875. https://doi.org/10.1007/s12155-012-9198-y.

Fritsche UR, Sims RE, Monti A. 2010. Direct and indirect land-use competition issues for energy crops and their sustainable production–an overview. Biofuels, Bioproducts and Biorefining, 4(6): 692-704. https://doi.org/10.1002/bbb.258.

Gabrielle B, Gagnaire N. 2008. Life-cycle assessment of straw use in bio-ethanol production: A case study based on biophysical modelling. Biomass and Bioenergy, 32(5): 431-441. https://doi.org/10.1016/j.biombioe.2007.10.017.

Gelfand I, Sahajpal R, Zhang X, Izaurralde RC, Gross KL, Robertson GP. 2013. Sustainable bioenergy production from marginal lands in the US Midwest. Nature, 493(7433): 514-517. https://doi.org/10.1038/nature11811.

Gerbens-Leenes PW. 2017. Bioenergy water footprints, comparing first, second and third generation feedstocks for bioenergy supply in 2040. European Water, 59: 373-380. https://doi.org/10.1073/pnas.0812619106.

Gerbens-Leenes W, Hoekstra AY, van der Meer TH. 2009. The water footprint of bioenergy. Proceedings of the National Academy of Sciences, 106(25): 10219-10223. https://doi.org/10.1073/pnas.0812619106.

Gnansounou E, Dauriat A, Villegas J, Panichelli L. 2009. Life cycle assessment of biofuels: Energy and greenhouse gas balances. Bioresource Technology, 100(21): 4919-4930. https://doi.org/10.1016/j.biortech.2009.05.067.

Halvorsen KE, Barnes JR, Solomon BD. 2009. Upper Midwestern USA Ethanol Potential from Cellulosic Materials. Society and Natural Resources, 22(10): 931-938. https://doi.org/10.1080/08941920902755382.

Heaton E, Voigt T, Long SP. 2004. A quantitative review comparing the yields of two candidate C4 perennial biomass crops in relation to nitrogen, temperature and water. Biomass and Bioenergy, 27(1): 21-30. https://doi.org/10.1016/j.biombioe.2003.10.005.

Herath I, Green S, Horne D, Singh R, Clothier B. 2014. Quantifying and reducing the water footprint of rain-fed potato production, part I: Measuring the net use of blue and green water. Journal of Cleaner Production, 81: 111-119. https://doi.org/10.1016/j.jclepro.2014.06.026.

Herrera I, Gamarra AR, Lechón Y, de la Rúa C, Caldés N. 2017. Análisis de ciclo de vida del proceso de produccion de etanol a partir de sorgo granífero en la planta de Paysandú de la empresa Alcoholes del Uruguay (ALUR). Informe Técnico. Madrid. CIEMAT. 69 p.

Herrera I, Lechón Y, de la Rúa C, Caldés N. 2016. Análisis de ciclo de vida del proceso de produccion de biodiesel a partir de un mix de materias primas grasas en la empresa alcoholes del Uruguay (ALUR). Informe Técnico. Madrid. CIEMAT. 68 p.

Herrera I, de la Rúa C, Lechón Y. 2015. Análisis de ciclo de vida del proceso de transformación de la caña de azúcar para la producción de bioetanol en la planta de Bella Unión de la empresa ALUR. Informe Técnico. Madrid. CIEMAT. 69 p.

Hillier J, Hawes C, Squire G, Hilton A, Wale S, Smith P. 2009. The carbon footprints of food crop production. International Journal of Agricultural Sustainability, 7(2): 107-118. https://doi.org/10.3763/ijas.2009.0419.

Hoekstra AY, Chapagain AK, Mekonnen MM, Aldaya MM. 2011. The water footprint assessment manual: Setting the global standard. London: Routledge. 204 p.

IEA (International Energy Agency). 2018. Renewables 2018: Analysis and Forecasts to 2023. OECD Publishing. Paris. 211 p.

IEA (International Energy Agency). 2017. World Energy Outlook 2017. OECD Publishing. Paris. 763 p.

ISO (International Organization for Standardization). 2017. ISO/TR 14073:2017.

Environmental Management - Water Footprint - Illustrative Examples on How to Apply ISO 14046. Geneva, Switzerland.

ISO (International Organization for Standardization). 2014. ISO 14046: Water Footprint - Principles, Requirements and Guidelines, in: Environmental Management. Geneva, Switzerland.

ISO (International Organization for Standardization). 2006a. ISO 14040: Environmental Management e Life Cycle Assessment e Principles and, Framework. International Organization for Standardization. Geneva, Switzerland.

ISO (International Organization for Standardization). 2006b. ISO 14044: Environmental Management e Life Cycle Assessment e Requirements and Guidelines. International Organization for Standardization. Geneva, Switzerland.

Ji X, Long X. 2016. A review of the ecological and socioeconomic effects of biofuel and energy policy recommendations. Renewable and Sustainable Energy, Reviews, 61: 41-52. https://doi.org/10.1016/j.rser.2016.03.026.

Khan MN, Mohammad F. 2014. Eutrophication: Challenges and solutions. In: Eutrophication: Causes, Consequences and Control. London. Springer, Dordrecht. 1-15.

Kim S, Dale BE. 2005. Life cycle assessment of various cropping systems utilised for producing biofuels: Bioethanol and biodiesel. Biomass and Bioenergy, 29(6): 426-439. https://doi.org/10.1016/j.biombioe.2005.06.004.

Köpke U, Nemecek T. 2010. Ecological services of faba bean. Field Crops Research, 115(3): 217–233.

Kruk C, Segura A, Nogueira L, Carballo C, Martínez de la Escalera G, Calliari D, Ferrari G, Simoens M, Cea J, Alcántara I, Vico P, Míguez D, Piccini C. 2015. Monitoring tools and early warning system for harmful cyanobacterial blooms: Rio Uruguay and Rio de la Plata. Innotec. 10: 23-39.

Kruk C, Suárez C, Ríos M, Zaldúa N, Martino D. 2013. Ficha: análisis calidad de agua en Uruguay. Asesoramiento Ambiental Estratégico, Vida Silvestre, IUCN. Montevideo, Uruguay.

Lal R. 2009. Soil quality impacts of residue removal for bioethanol production. Soil Tillage Research, 102(2): 233-241. https://doi.org/10.1016/j.still.2008.07.003.

Lal R. 2004. Soil carbon sequestration to mitigate climate change. Geoderma, 123(1-2): 1-22. https://doi.org/10.1016/j.geoderma.2004.01.032.

Lemus R, Lal R. 2005. Bioenergy Crops and Carbon Sequestration. Critical Reviews in Plant Sciences, 24(1): 1-21. https://doi.org/10.1080/07352680590910393.

Lewandowski I, Schmidt U. 2006. Nitrogen, energy and land use efficiencies of miscanthus, reed canary grass and triticale as determined by the boundary line approach. Agriculture, Ecosystems & Environment, 112(4): 335-346. https://doi.org/10.1016/j.agee.2005.08.003.

Liska AJ, Yang H, Milner M, Goddard S, Blanco-Canqui H, Pelton MP, Fang XX, Zhu H, Suyker AE. 2014. Biofuels from crop residue can reduce soil carbon and increase CO_2 emissions. Nature Climate Change, 4(5): 398-401. https://doi.org/10.1038/nclimate2187.

Liska AJ, Yang HS, Bremer VR, Klopfenstein TJ, Walters DT, Erickson GE, Cassman KG. 2009. Improvements in Life Cycle Energy Efficiency and Greenhouse Gas Emissions of Corn-Ethanol. Journal of Industrial Ecology, 13(1): 58-74. https://doi.org/10.1111/j.1530-9290.2008.00105.x.

Lizarralde C, Picasso V, Rotz CA, Cadenazzi M, Astigarraga L. 2014. Practices to reduce milk carbon footprint on grazing dairy farms in southern Uruguay: Case studies. Sustainable Agriculture Research, 3(2): 1-15. http://dx.doi.org/10.5539/sar.v3n2p1.

Lizarralde PC. 2013. Relación entre la huella de carbono y las prácticas de manejo de predios lecheros de Uruguay. Tesis de Maestría. Montevideo, Uruguay. Facultad de Agronomía. 81 p.

Lovarelli D, Bacenetti J, Fiala M. 2016. Water Footprint of crop productions: A review. Science of the Total Environment, 548: 236-251. https://doi.org/10.1016/j.scitotenv.2016.01.022.

Ludeña CE, Ryfisch D. 2015. Uruguay: Mitigación y adaptación al cambio climático. Banco Interamericano de Desarrollo. División de Cambio Climático y Sostenibilidad, Washington D.C. Nota Técnica, 860: 1-28.

Lugato E, Jones A. 2015. Modelling Soil Organic Carbon Changes Under Different Maize Cropping Scenarios for Cellulosic Ethanol in Europe. Bioenergy Research. 8(2): 537-545. https://doi.org/10.1007/s12155-014-9529-2.

Macedo YI, Terra JA, Siri G, Carrasco-Letelier L, Velazco JI. 2020. Intensification of rice-pasture production systems: changes in energy performance. [Datos sin publicar].

Macedo YI. 2018. Calidad de suelos y eficiencia de uso de energía en rotaciones arroceras contrastantes. Tesis de maestría. Montevideo, Uruguay. Facultad de Agronomía. 70 p.

Mathioudakis V, Gerbens-Leenes PW, Van der Meer TH, Hoekstra AY. 2017. The water footprint of second-generation bioenergy: A comparison of biomass feedstocks and conversion techniques. Journal of Cleaner Production, 148: 571-582. https://doi.org/10.1016/j.jclepro.2017.02.032.

Mekonnen MM, Hoekstra AY. 2011. The green, blue and grey water footprint of crops and derived crop products. Hydrology and Earth System Sciences, 15: 1577-1600. https://doi.org/10.5194/hess-15-1577-2011.

MIEM (Ministerio de Industria, Energía y Minería). 2018. Balance energético 2018. Consultado 18 enero 2020. Disponible en: https://ben.miem.gub.uy/descargas/1balance/1-1-Libro-BEN2018.pdf

Milà I Canals L, Bauer C, Depestele J, Dubreuil A. Knuchel RF, Gaillard G, Michelsen O, Müller-Wenk R, Rydgren B. 2007. Key elements in a framework for land use impact assessment within LCA. International Journal of Life Cycle Assessment, 12(1): 5-15. https://doi.org/10.1065/lca2006.05.250.

Modernel P, Astigarraga L, Picasso V. 2013. Global versus local environmental impacts of grazing and confined beef production systems. Environmental Research Letters. 8(3): 035052. http://dx.doi.org/10.1088/17489326/8/3/035052.

Modernel P. 2011. Emisiones de gases de efecto invernadero en invernada vacuna del Uruguay. Tesis de Maestría. Montevideo, Uruguay. Facultad de Agronomía. 65 p.

Monti A, Barbanti L, Zatta A, Zegada-Lizarazu W. 2012. The contribution of switchgrass in reducing GHG emissions. GCB Bioenergy, 4(4): 420-434. https://doi.org/10.1111/j.1757-1707.2011.01142.x.

Morales M, Quintero J, Conejeros R, Aroca G. 2015. Life cycle assessment of lignocellulosic bioethanol: Environmental impacts and energy balance. Renewable and Sustainable Energy Reviews, 42: 1349-1361. https://doi.org/10.1016/j.rser.2014.10.097.

Morón A, Quincke A, Molfino J, Ibáñez W, García A. 2012. Soil quality assessment of Uruguayan agricultural soils. Agrociencia, 16(3): 135-143.

Nemecek T, Hayer F, Bonnin E, Carrouée B, Schneider A, Vivier C. 2015. Designing eco-efficient crop rotations using life cycle assessment of crop combinations. European Journal of Agronomy, 65: 40-51. https://doi.org/10.1016/j.eja.2015.01.005.

Picasso V, Modernel P, 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 Science, 98(3): 346-354. https://doi.org/10.1016/j.meatsci.2014.07.005.

Rodríguez-Gallego L, Achkar M, Defeo O, Vidal L, Meerhoff E, Conde D. 2017. Effects of land use changes on eutrophication indicators in five coastal lagoons of the Southwestern Atlantic Ocean. Estuarine, Coastal and Shelf Science, 188: 116-126. https://doi.org/10.1016/j.ecss.2017.02.010.

Roel A, Bonilla C, Saizar C, Martinez G. 2013. Huella de carbono del sector arrocero. Montevideo, Uruguay. Arroz (ACA), 15(74): 12-17.

Schmer MR, Jin VL, Wienhold BJ. 2015. Sub-surface soil carbon changes affects biofuel greenhouse gas emissions. Biomass and Bioenergy, 81: 31-34. https://doi.org/10.1016/j.biombioe.2015.05.011.

Siri-Prieto G. 2013. Bioenergía en el Uruguay ¿cómo hacerla productiva y sustentable? In: Simposio Nacional de Agricultura (3o, 2013, Paysandú, Uruguay). No Se Llega, Si No Se Sabe a Dónde Ir. Pensar En Las Causas y No Sólo Medir Las Consecuencias. Montevideo, Uruguay, Hemisferio Sur. 21-38.

Steduto P, Hsiao TC, Raes D, Fereres E. 2009. AquaCrop the FAO crop model to predict yield response to water. I Concepts. Special issue on "Yield Response to Water: Examination of the Role of Crop Models in Predicting Water Use Efficiency". Agronomy Journal, 101: 426-437. https://doi.org/10.2134/agronj 2008.0139s.

Tittonell P. 2014. Ecological intensification of agriculture-sustainable by nature. Current Opinion in Environmental Sustainability, 8: 53-61. https://doi.org/10.1016/j.cosust.2014.08.006.

Tsakmakis ID, Zoidou M, Gikas GD, Sylaios GK. 2018. Impact of Irrigation Technologies and Strategies on Cotton Water Footprint Using AquaCrop and CROPWAT Models. Environmental Processes. 5(1): 181-199. https://doi.org/10.1007/s40710-018-0289-4.

Uruguay. Poder Legislativo. 2007. 18.195/007. Agrocombustibles. [En línea]. Consultado 15 noviembre 2019. Disponible en: https://legislativo.parlamento.gub.uy/temporales/leytemp7718815.htm

Uruguay. Poder Legislativo. 2002. 17.567/002. Combustibles Alternativos, Renovables y Sustitutivos de los Derivados del Petróleo elaborados con material prima nacional de origen animal o vegetal. [En línea]. Consultado 10 noviembre 2019. Disponible en: https://www.miem.gub.uy/sites/default/files/ley_17567_-_combustibles_alternativos.pdf

Uruguay XXI, 2017. Energías renovables: Oportunidades de inversión. Montevideo. [En línea]. Consultado 8 diciembre 2019. Disponible en: http://www.ps.org.uy/wp-content/uploads/2017/10/Informe-de-Energ%C3%ADas-Renovables-Setiembre-de-2017-Uruguay-XXI-1.pdf

Van Heerden PS, Crosby CT, Grove B, Benade N, Theron E, Schulze RE, Tewolde MH. 2008. Integrating and upgrading of SAPWAT and PLANWAT to create a powerful and user-friendly irrigation water planning tool. Water Research Commission, Report number: TT391/08, Pretoria.

Vázquez D, Carrasco-Letelier L, D’ Ottone F, Resquín F, Scoz R., Vilaró F, Rodríguez G, Vicente E, Terra J. 2013. Evaluating the sustainability of potential agro-industrial chains (sweet sorghum, grain sorghum, sweet potato and forestry) for agroenergy production. In: USDA-INIA Workshop of ECPA Project "Evaluating the Sustainability of Potential Agro-Industrial Chains for Agroenergy Production".

Wang M, Chen Y, Xia X, Li J, Liu J. 2014. Energy efficiency and environmental performance of bioethanol production from sweet sorghum stem based on life cycle analysis. Bioresource Technology, 163: 74-81. https://doi.org/10.1016/j.biortech.2014.04.014.

Zhong J, Yu TE, Clark CD, English BC, Larson JA, Cheng CL. 2018. Effect of land use change for bioenergy production on feedstock cost and water quality. Applied Energy, 210: 580-590. https://doi.org/10.1016/j.apenergy.2017.09.070.