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A comprehensive analysis of the environmental performance of the Uruguayan agricultural sector

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ABSTRACT

Ensuring food production while enhancing environmental sustainability is a critical challenge in the 21st century. Quantitative descriptions of environmental performance at the farm level are essential for evaluating agricultural production, aligning with climate and biodiversity goals, and facilitating sustainable transitions. However, many existing indicators and proxies rely on costly field-collected data with limited spatial generalization.

In this study, we assessed the environmental performance of Uruguayan farms larger than 5 ha using seven synoptic indicators derived from remote sensing. These indicators included the proportion of natural habitats (NatHab), diversity of Ecosystem Functional Types (dEFT), supply of regulating and supporting Ecosystem Services (ESSI), their temporal trends (tESSI), Energy Available for Trophic Network (EATN; 1- Human Appropriation of Net Primary Production), Hydrological Yield (HY), and Soil Conservation (SC).

We categorized rural cadastral units from different geomorphological regions into Cropland, Mixed, and Livestock production types. Results showed variations in environmental performance among production systems and regions, with livestock farms generally outperforming and exhibiting less variability. However, cropland farms displayed potential for comparable environmental performance to less intensified areas.

Regional disparities were evident, with the Basaltic region demonstrating higher overall performance. Indicators such as NatHabs, dEFT, and EATN exhibited significant variation, reflecting land-use and management practices. HY also showcased notable regional and land-use differences, influenced by soil characteristics and landscape features. SC varied mainly between geomorphological regions. Interestingly, regional patterns differed among indicators, suggesting low redundancy.

This study provides valuable insights into environmental performance and its spatial dynamics in the Uruguayan agricultural sector, informing land management and policy decisions. Future studies should engage diverse social actors to develop an environmental performance index for agricultural production, enhancing the sustainability of food production systems.

1. Introduction

Maintaining or increasing food production while improving environmental performance is one of humanity's most significant challenges of this century (UN, 2023). In a scenario of increasing food demand, decision-makers, NGOs and Academia suggest that production systems should be transformed towards uses that reduce external inputs and the impact on ecosystems without compromising food security (Foley et al., 2011; Kremen et al., 2012). In this context, the quantitative description of environmental performance plays a critical role in assessing the sustainability of agricultural socio-ecological systems (Pacini et al., 2003; Hayati et al., 2011; Rasmussen et al., 2017; Lynch et al., 2018; Bergez et al., 2022). Environmental evaluations are essential to meet national climate and biodiversity goals and to assess sustainable or

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agroecological transitions at the farm level. From a more operational perspective, local and global markets demand information on the environmental performance for generating certifications or differentiating products. Actually, environmental impacts are a major issue in debating trade agreements among regional blocks (i.e., EU-Mercosur; Kehoe et al., 2020) or become a barrier for some products (e.g. soybean produced in areas deforested after 2020).

The idea of environmental performance seeks to characterize the environmental dimension of the sustainability of agricultural systems. This characterization is not absolute but relative to a reference situation in space (i.e., other areas) or in time (i.e., areas in the past or the future) (Paruelo and Sierra, 2022). Changes in the supply of Ecosystem Services (ES) can be interpreted as a change in the environmental dimension of the sustainability of a given system:

$$\Delta \text{ Environmental Dimension of the Sustainability} = \Delta \text{ ES supply}$$

The capacity to provide services has been linked to the idea of ecosystem health. The concept of ecosystem health is closely related to sustainability (Diaz et al., 2015). A healthy ecosystem has to be stable and sustainable, maintain its organization and autonomy over time, and show resilience to disturbances or factors that generate some degree of stress (Costanza et al., 1992).

The "cascade" model of ES generation (de Groot et al., 2010; Haines-Young and Potschin, 2010), links ecosystem structure and functioning with human well-being. In this scheme, Intermediate ES (Fisher et al., 2009) includes the structural and functional aspects of ecosystems (i.e. primary production, GHG emissions, different taxa biodiversity, runoff, N and P exports) that, in turn, determine the supply of Final ES from which Society derive benefits from a given land unit (de Groot et al., 2010; Haines-Young and Potschin, 2010).

Various ES supply proxies or indicators have been proposed (Stephens et al., 2015). Such proxies and indicators are the building blocks of an environmental characterization of a land unit. Indicators are variables (either modeled, observed, measured, or calculated from other data) that reflect a quantitative or qualitative characteristic of the system under study that are more difficult to assess. Also, they must be sensitive to perturbations, such as land use change. They provide key information to know the current and/or past state of the system and, frequently, to make a decision (Donnelly et al., 2007; Dillon et al., 2016). Such indicators are diagnostic elements and not an end in themselves (Dale and Beyeler, 2001).

Indicators should be connected to specific ES, either intermediate or final, to conduct effective environmental assessments. In such a way, indicators are often integrated into the "production functions" (Daily and Matson, 2008) of ES. For example, the Normalized Difference Vegetation Index (NDVI) is a spectral index derived from remote sensing that allows the calculation of Net Primary Production (NPP), a key ecosystem functional attribute (McNaughton et al., 1989). NPP is an Intermediate ES that determines the supply of several Final services (Richmond et al., 2007), such as meat production or carbon sequestration (provision and regulation ecosystem services, respectively; MEA, 2005). Indicators should also complement each other to offer a systemic and integrative description. Focusing solely on a single dimension, such as the carbon balance, may overlook other significant environmental impacts. In such a way, indicators of C dynamics should be evaluated together with those reflecting water dynamics, nutrient cycling, and biodiversity to have a comprehensive characterization of the environmental performance of a land unit.

Over the last decades and across the Globe a myriad of methods to evaluate environmental performance and agricultural sustainability have been proposed (see Jørgensen et al., 2005; Lattrufe et al., 2016; Gil et al., 2019). Indicators based on field-collected data (i.e. soil samples for soil Carbon stocks, water quality, or biodiversity of specific taxa) are expensive and have a limited potential for spatial generalization. On the other hand, several protocols have been developed to generalize in space ES supply at the plot level using land cover maps (i.e. Burkhard et al., 2012). Some of the most common protocols (e.g. InVEST, ARIES, ECOSER) use a look-up table with fixed parameters for each land cover type to estimate the supply of specific ES (Nelson et al., 2009; Nelson and Daily, 2010; Laterra et al., 2012). Although such models represented a major step in describing and generating scenarios of ES supply, they failed to properly discriminate environmental performance within areas with the same land cover type. This is a key aspect in the environmental assessments of food systems because different agricultural managements might have very contrasting impacts despite being applied on the same cover (Baldassini et al., 2023).

Remotely sensed data not only enable the description of spatial and temporal land cover patterns but also allow for a direct estimation of ecosystem processes (Cabello et al., 2012; Pettorelli et al., 2017), e.g. multiple processes associated with the exchanges of matter and energy between the Earth's surface, the biota and the atmosphere. Remote sensed data on surface reflectance and/or emission allow for estimating the temporal dynamics of surface temperature, albedo, radiation absorption, and radiation use efficiency, among other biophysics variables (Prata et al., 1995; Liang 2000; Pettorelli et al., 2016). Moreover, this information can be integrated into biophysical models to estimate critical ecosystem processes or intermediate services (i.e. evapotranspiration, Moran and Jackson, 1991; or primary production, Potter et al., 1993; Ruimy et al., 1994). Such indicators provide a synoptic view of the system and a full spatial coverage of the territory. They also allow for the environmental performance comparison between land covers using a common observation protocol (Paruelo, 2008).

In countries where most of the land is privately-owned, sustainable management alternatives, policies and socioeconomic instruments operate at the farm/ranch level. In such a way, it is critical to have a description of environmental performance at this jurisdictional resolution. Moreover, the evaluation of policies and management plans at the farm level must be able to track changes through time. Indicators based on field data are often impractical due to logistic restrictions.

The agricultural sector of Uruguay has a central economic and social importance. Agricultural products represent more than 80 % of the total exports (Uruguay XXI, 2022). Concerns on the sustainability of the production systems come from both outside (i.e. Kehoe et al., 2020) and inside the country. Internal concerns led the Uruguayan parliament to pass an act aimed to promote agroecological systems (Law N° 19.717/2017) and the government to build the Environmental Footprint of the livestock systems (MA, 2022).

In this article, we provide an assessment of the environmental performance of farms larger than, aprox., 15 ha. We focused on published and already locally tested synoptic indicators and regional products of national coverage: the proportion of natural habitats, the diversity of Ecosystem Functional Types, the supply of regulating and supporting Ecosystem Services and its temporal trend, Energy Available at Trophic Network (1- Human Appropriation of Net Primary Production), the Hydrological Yield, and Soil Conservation. The first 5 indicators listed were incorporated by the Uruguayan government as some of the metrics that define the Environmental Footprint of the livestock systems (MA, 2022). We also studied the correlation among indicators to identify redundancy and complementarities. Our analyses present a within-country evaluation in relative terms. We did not assess the sustainability of farms against a normative framework.

2. Materials and methods

We quantified the seven indicators listed above at the level of rural cadastral units (called "padrones", http://www.catastro.gub.uy), although an actual farm may include more than one cadastral unit. Among the 250.321 rural units of Uruguay, we selected those that completely included at least one pixel of the MODIS satellite images (~5.3 ha), the coarsest spatial resolution product used in this work. This selection resulted in 99.990 units, which were used for calculating the proportion of different land uses based on the land cover map generated by the MapBiomas Pampa initiative (Baeza et al., 2022) for the year 2020 (available at https://pampa.mapbiomas.org/) (Figure S1). This map was made using Landsat images, which have a spatial resolution of 30 m. The vegetation land use and land cover classes of MapBiomas Pampa included: native grasslands, wetlands, native woodlands, tree plantations and agricultural areas (annual crops and sowed pastures). To differentiate croplands from sowed perennial pastures, the agricultural class was combined with a land cover map from Baeza and Paruelo (2020), which identifies a "perennial forage resources" class (native grasslands + sowed pastures). Therefore, the category "sowed pasture" was assigned to the coinciding areas between the MapBiomas "agricultural" class and the perennial forage resources class, and the class "agricultural" to the agricultural area that does not coincide with the perennial forage class of Baeza and Paruelo (2020).

We discarded cadastral units dominated by urban and forests classes (less than 70 % of croplands + native grasslands + sowed pastures), so the final subset included 77.608 cadastral units (median: 71.2 ha, p25 = 36.7 ha, and p75 = 163.8 ha), which were categorized into "Cropland Farms" (>= 70 % of the area devoted to annual crops), "Mixed Farms" (annual crops cover between 30 and 70 % of the unit) and "Livestock Farms" (native grasslands and sowed pastures cover >=70 % of the unit). These thresholds were defined based on the descriptive statistics of livestock farms in Uruguay as presented by Modernel et al. (2018). We

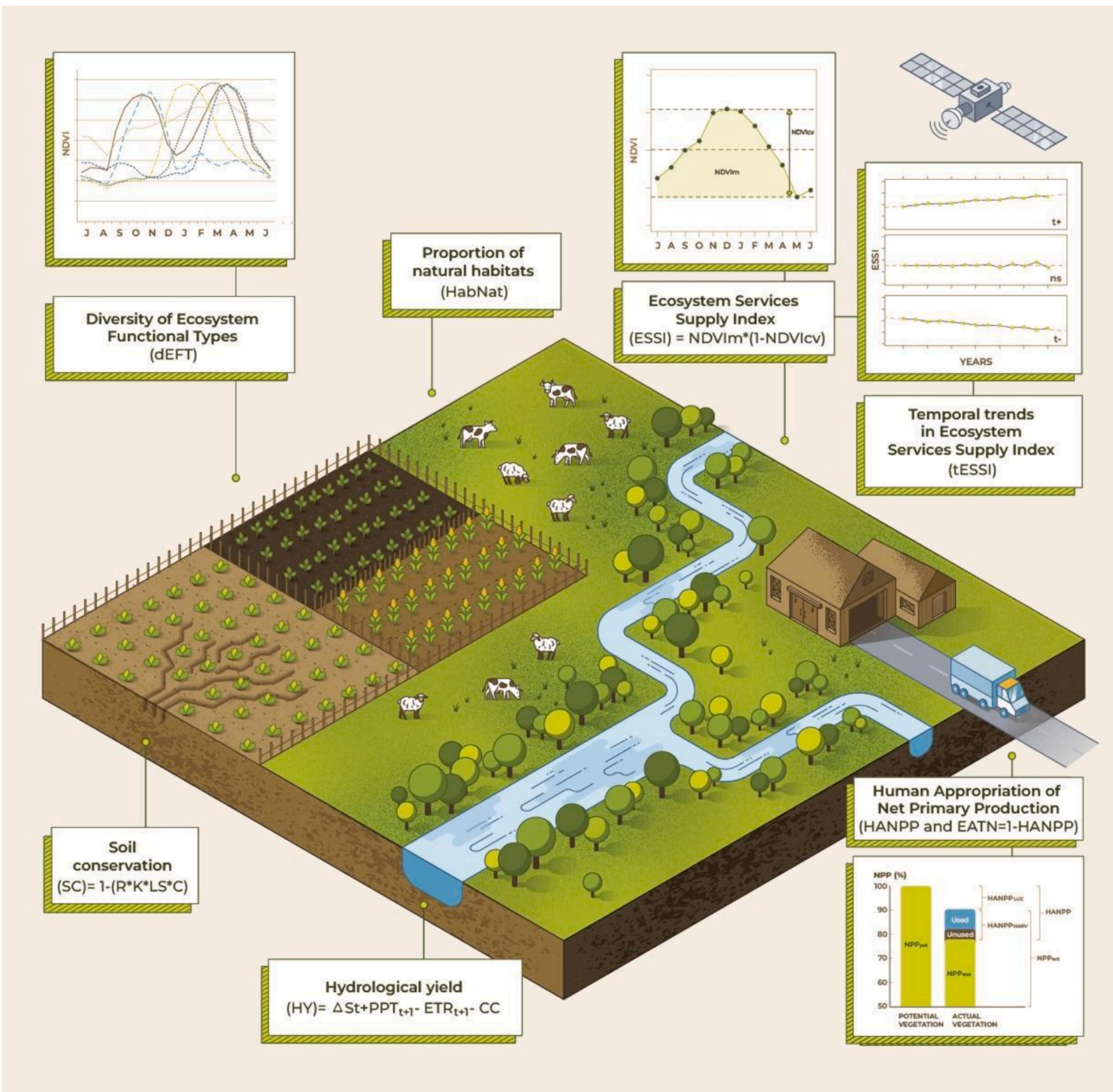

Fig. 1. Remote sensed indicators evaluated. dEFT: diversity of Ecosystem Functional Types; HABNAT: Proportion of Natural Habitats; ESSI: Ecosystem Services Supply Index; tESSI: temporal trend in the Ecosystem Services Supply Index; EATN: Energy Available to Trophic Network (1-Human Appropriation of Net Primary Production); HY: Hydrological Yield; SC: Soil Conservation (1 – scaled RUSLE). While for certain indicators, like RUSLE, the HANPP or the diversity of Ecosystem Functional Types, a formal evaluation at broad scale are not feasible, the other indicators derived from remote sensing used have undergone local testing. For the Ecosystem Services Supply Index (and its temporal trend) such evaluation is reported in Paruelo et al (2016) & Staiano et al (2021). In the case of the proportion of Natural Habitats, Baeza et al. (2022) performed an formal evaluation of the land cover classification. Gallego et al (2023b) analyzed the performance of the procedure used to calculate hydrological yield for two watersheds.

also utilized a map of the geomorphological regions of Uruguay (Panario et al., 2014) to determine the region in which each cadastral unit is located. The "Santa Lucía graben'' region was included into the "Crystalline shield" due to its size and characteristics. We ended up with 6 regions: Western sediment basin, Crystalline shield, Basaltic region, Gondwanic Sediment basin, Eastern hills, and Lagoon Merin graben.

2.1. Indicators evaluated

We calculated the proportion of natural habitats, the diversity of Ecosystem Functional Types, the supply of regulating and supporting Ecosystem Services and its temporal trend, the Energy Available to the Trophic Network, the hydrological yield, and soil conservation for each cadastral unit of the processed cadastre described above (Fig. 1). These indicators were selected based on the following aspects: a. supporting scientific evidence and local evaluation (see Volante et al., 2012; Paruelo et al, 2016, Staiano et al., 2021, Verón et al., 2018, Baeza and Paruelo, 2018, Baldassini et al., 2023, Storkes et al., 2024), b. estimation/calculation protocol including data requirements, d. environmental processes evaluated (i.e. biodiversity, ecosystem services supply, C dynamics), e. expected limitations for its generalization over the study regions, f. level of legitimization by different stakeholders. The last point was evaluated based on the list of synoptic indicators included into Environmental Footprint of the Livestock Sector (Huella Ambiental de la Ganadería in UY, https://www.gub.uy/ministerio-ambiente/comunicacion/noticias/huella-ambiental-ganaderia-uruguay). Five out of seven indicators were part of the set defined by this panel. Soil Conservation was based on the same model used by the uruguayan government to evaluate Agricultural Management Plans (https://www.gub.uy/ministerio-ganaderia-agricultura-pesca/politicas-y-gestion/planes-uso-manejo-suelos). The only indicator that is not part of an official list yet is the hydrological yield.

2.1.1. Proportion of natural habitats (HABNAT)

The proportion of natural habitats was calculated as the sum of the area covered by native grasslands, wetlands and woodlands (native forests) divided by the area of the cadastral unit. Land use/land cover data were derived from the MapBiomas Pampa initiative (Baeza et al., 2022, MapBiomas Pampa, 2023). MapBiomas land cover maps were formally tested based on ground truth (Baeza et al., 2022).

2.1.2. Diversity of Ecosystem Functional Types (dEFT)

Ecosystem Functional Types (EFT) are groups of ecosystems or patches of the land surface that share similar dynamics of matter and energy exchanges between the biota and the physical environment (Paruelo et al., 2001; Alcaraz-Segura, 2006, Baeza et al., 2006; Cazorla et al., 2021, Bagnato et al., 2024). EFTs result from the combination of three attributes of the annual dynamics of a spectral index: the annual mean, the intra-annual coefficient of variation and the month in which the highest EVI value occurs. Specifically we based calculations on the Enhanced Vegetation Index (EVI) average of the analyzed period (derived from MODIS sensor images, MOD13Q1 product with a spatial resolution of 230 m and a temporal resolution of 16 days). The diversity of ecosystem functional types (EFT) evaluates the functional biodiversity of the territory in terms of the seasonal dynamics of carbon gains (described by the Shannon index, Alcaraz-Segura et al., 2013; Gallego et al., 2023a). In our case, EFT diversity was computed for the area occupied by croplands and pastures and, then, it is a measure of the multifunctionality of the non-natural portion of the agricultural landscapes (Staiano et al., 2022). Greater diversity and spatial heterogeneity is associated with a greater supply of habitats (Fahrig et al., 2011; Tscharntke et al., 2012; Bommarco et al., 2013). Thus, a more diverse and heterogeneous non-natural system would present greater resilience and a more efficient use of resources (Davis et al., 2012; Gaudin et al., 2015; Gurr et al., 2016).

Since EFT diversity depends on cadastral unit size, the residuals of a model that relates EFT diversity with the unit size were estimated to obtain an estimate of this indicator that was independent of the plot size (see the supplementary material for more details). The Michaelis-Menten model was used to describe a saturation curve where EFT diversity (y-axis) increases with the area of the farm until reaching a plateau. From this model, residuals were obtained as the difference between the observed values and those estimated by the model. The residuals were scaled to have a range of variation between 0 and 1. The scaled residuals of the EFT diversity ratio as a function of land area are the indicator of diversity used.

2.1.3. Supply of regulating and supporting Ecosystem Services (ESSI)

We computed the Ecosystem Service Supply Index (ESSI) (Paruelo et al., 2016). This index is positively associated with specific regulating and supporting services related to carbon and water dynamics (Staiano et al., 2021; Baldassini et al., 2023; Gallego et al., 2023a). Particularly important is the linear relationship of the ESSI with soil carbon stocks (Staiano et al., 2021; Baldassini et al., 2023). The ESSI combines two attributes of the seasonal dynamics of the Normalized Difference Vegetation Index (NDVI), its annual mean (NDVIm), and its coefficient of seasonal variation (NDVIcv): ESSI=NDVIm * (1 − NDVIcv).

NDVI was obtained from the MOD13Q1 MODIS product, which has a spatial resolution of 230 m and a temporal resolution of 16 days. The annual mean NDVI is an estimator of total carbon gains, while the NDVI coefficient of variation indicates the temporal stability or seasonality of those gains. High values of ESSI are associated with covers with high and stable productivity during the growing season, while low values correspond to covers with lower productivity and/or more variability. The average ESSI per cadastral unit was calculated for the 2017–2018 and 2018–2019 growing seasons (each period taken from July to June).

2.1.4. Temporal trends of supply of regulating and supporting Ecosystem Services (tESSI)

The temporal trends of the ESSI (tESSI) (Paruelo et al., 2016) were calculated for each MODIS pixel, obtaining the slope of the regression of the ESSI over the 2000–2019 period. The information at the cadastral unit was summarized as the proportion of negative, positive, and neutral (i.e. statistically non-significant) temporal trends of the index. We used the sum of the proportion of pixels with neutral and positive ESSI trends as the environmental performance indicator for this dimension.

2.1.5. Human Appropriation of Net Primary Production and Energy Available to the Trophic Network (HANPP and EATN)

This indicator reports the human impact on ecosystems and it is an estimator of land use intensity (Haberl et al., 2004, 2007). Human appropriation of net primary production (HANPP) results from the difference between the net primary production (NPP) in the absence of human influence (NPP of potential vegetation: NPP_0) and the NPP of current vegetation remaining after harvest (NPP_{REM}). NPP_{REM} is calculated as the NPP of current vegetation (NPP_{ACT}) minus the NPP harvested (NPP_{HAR}), directly appropriated by humans as agricultural products (grain, wood, meat, etc.) or destroyed during harvesting. NPP_0 was assumed equal to NPP of native grassland, the original vegetation type of most of Uruguayan territory (see Baeza and Paruelo, 2018 for details). Although grasslands in Uruguay are grazed by domestic livestock, there is no consistent effect of grazing on NPP productivity in this region (Oesterheld et al., 1999; Rusch and Oesterheld, 1997; Altesor et al., 2005). The approximation used takes into account regional variations in NPP_0, a value of NPP_0 was obtained for each geomorphological region, taking the 95th percentile of the NPP of all remaining grasslands, calculated according to Monteith's model (1972). We have made this decision under the assumption that NPP_0 should resemble, as closely as possible, that of the best conserved and most productive grasslands.

HANPP was estimated for the 2017–2018 and 2018–2019 growing seasons for each cadastral unit by combining land cover maps, crop yield estimates, forage production derived from remote sensing data, and correlative models based on Baeza and Paruelo (2018). The values obtained for each growing season were then averaged. We reported the complement of the HANPP, which quantifies the energy available to the trophic network (EATN=1-HANPP).

2.1.6. Hydrological yield (HY)

We calculated the annual hydrological yield at the cadastral unit level for the 2017–2018 and 2018–2019 growing seasons. This indicator is related to the water production (Salemi et al., 2012) that supports wildlife, stream functioning, agricultural irrigation, drinking water supply, and other ecosystem services. The hydrological yield estimation considered soil water content (ΔS), precipitation (PPT) obtained from the Climate Hazards Group InfraRed Precipitation with Station product (CHIRPS, Funk et al., 2015), actual evapotranspiration derived from the MOD16A2 MODIS product (ETR), and the field water capacity up to 1-meter (FWC) derived from the Hengl and Gupta (2019) product: $HY=\Delta S+PPT - ETR - FWC$. The hydrological yield was estimated daily using a function that iterates the calculation, taking into account the soil water content from the previous day. Annual values of this indicator were averaged (see Gallego et al., 2023b for more details).

2.1.7. Soil conservation (SC)

Soil conservation was estimated based on the revised version of the "Universal Soil Loss Equation" (RUSLE, Renard, 1997). The estimation of the RUSLE involves multiplying the precipitation erosivity (R), soil erodibility (K), topographic (T, associated with slope length and gradient), land cover (C), and conservation practice (P) factors: $RUSLE=R \times K \times T \times C \times P$. Higher RUSLE values indicate more soil erosion or less soil conservation. Elnashar et al. (2021) developed a framework for estimating the RUSLE with remote sensing data in the Google Earth Engine cloud platform (Gorelick et al., 2017). In this framework, each factor of the RUSLE is estimated with a combination of available satellite data, spatially explicit information, and biophysical models. We adapted this procedure to estimate the RUSLE of cropland pixels of Uruguay. The R factor of the RUSLE was calculated from the model proposed by Renard and Freimund (1994) which is appropriate for regions with annual precipitation over 850 mm (see equation 12 in Renard and Freimund, 1994). This model is empirical and it only needs the mean annual precipitation data for a climatic series (at least 30 years), so we used the Climate Hazards Group InfraRed Precipitation with Station data (CHIRPS; Funk et al., 2015) which provides daily rainfall data from the combination of satellite and in situ station data. We estimated the annual mean precipitation for a 30-year series from 1990 to 2019.

The K factor was estimated for the upper 30 cm of the soil profile using the Soil Erodibility Nomograph (Wishmeier & Smith 1978; Renard, 1997), which estimates the soil sensitivity to erosion as a function of the soil's structure, permeability, and texture. For this, we used the Soil Erodibility Nomograph equation (see equation 4 in Elnashar et al., 2021) and followed the procedure from Elnashar et al. (2021) for assigning soil structure and permeability values (see Table 1 in Elnashar et al., 2021) from soil texture and soil organic carbon data. We used the Soil Texture Class, Soil Clay Fraction, Soil Sand Fraction, and Soil Organic Carbon from the Open Land Map database (Hengl and Wheeler, 2018; Hengl, 2018). For estimating the % of very fine sand, we followed the assumptions from Elnashar et al. (2021) who estimated this fraction as 20 % of the sand fraction. The soil silt content was estimated as $100 - \%clay + \%sand$. The Organic Matter (OM) content was estimated as the soil organic carbon (SOC) content multiplied by 1.724 which assumes that 58 % of the OM is composed of SOC (Van Bemmelen, 1891; Tabatabai, 1996). We set 70 % as the upper limit of silt plus very fine sand contents and 4 % for OM content as defined by Elnashar et al. (2021) to prevent the underestimation of K values.

The topography factor (T) was estimated following the procedure from Elnashar et al. (2021), who used a set of equations to estimate the slope length and steepness (see equations 5 to 7.3 in Elnashar et al., 2021) from the NASA SRTM Digital Elevation Model data. We adapted this procedure using the Multi-Error-Removed Improved-Terrain DEM (Yamazaki et al., 2017), an enhanced Digital Elevation Model (DEM) produced by eliminating the error components of the most used DEM's (NASA SRTM and JAXA AW3D among others). The land cover factor (C factor) was estimated from the equation used by Almagro et al. (2019) and Sone et al. (2019) who used a calibration model of the Normalized Difference Vegetation Index (NDVI) and C factor values derived from soil erosion experimental plots developed in the Southern East of Brazil (see equation 2 in Almagro et al., 2019). We used the NDVI for the 2017–2018 and 2018–2019 growing seasons and then averaged for calculating the C factor. The factor P, associated with conservation practices such as the construction of crop terraces or the conservation of uncultivated edges, was not estimated in this study (was taken as 1) due to the impossibility of detecting such practices at the scale of the analysis. Furthermore, there are no reports of the widespread use of this kind of practices in Uruguay (Carrasco-Letelier and Beretta, 2017). Finally, the RUSLE was calculated by multiplying each described factor at the pixel level and then averaged to obtain a single estimation for each cadastral unit.

2.2. Comparative analysis and visualization

To ensure comparability among environmental performance indicators, they were scaled within a range of values from 0 to 1. This scaling process involved calculating the difference between each value of a cadastral unit and the minimum value for each indicator, which was then divided by the range (the difference between the maximum and minimum values). The 5th (p5) and 95th (p95) percentiles of each indicator along all cadastral units were considered as the minimum and maximum values, respectively. For a better interpretation of the results, we considered that the highest scaled values of the indicators represent the best environmental performance. Thus, to obtain the EATN, we calculated the complement of the HANPP and to obtain SC the complement of the scaled RUSLE equation. For facilitating comparative analysis of environmental performance across different uses and regions, we computed the median value for each indicator based on the productive system type (i.e. Cropland Farms, Mixed Farms, and Livestock Farms) and region (6 in total). We determined the median for each type of productive system type to compare the environmental performance. To represent the variability around the median for each combination of productive system and region we calculated the 25th (p25) and 75th (p75) percentiles. To visualize the environmental performance among regions and productive system types, we generated "flower plots". In these plots, each petal represents the performance of a specific environmental indicator. As the analysis involved the entire set of cadastral units (a census, not a sample), no statistical tests were conducted. A flowchart including in the Supplementary Material (Figure S2) summarizes the processes performed to characterize the environmental performance at the cadastral unit level.

Finally, we performed a Pearson's correlation analysis for assessing the level of complementation among the indicators calculated at the cadastral unit level. We performed the analysis for the whole data set ($n = 77.608$ cadastral units) and for a subset where we took into account spatial correlation. To consider the spatial correlation among cadastral units, semi-variograms for each indicator were made in order to find a threshold distance where correlations among farms diminished. For five out of seven indicators, a plateau in semi-variance values was reached at a distance of 25 km (see Supplementary Material, Figure S3).

3. Results and discussion

Across the whole country, it was possible to identify spatial patterns for each indicator of environmental performance considered (Fig. 2). Such patterns showed a partial correspondence with the geomorphological units defined by Panario et al. (2014). The boundaries among

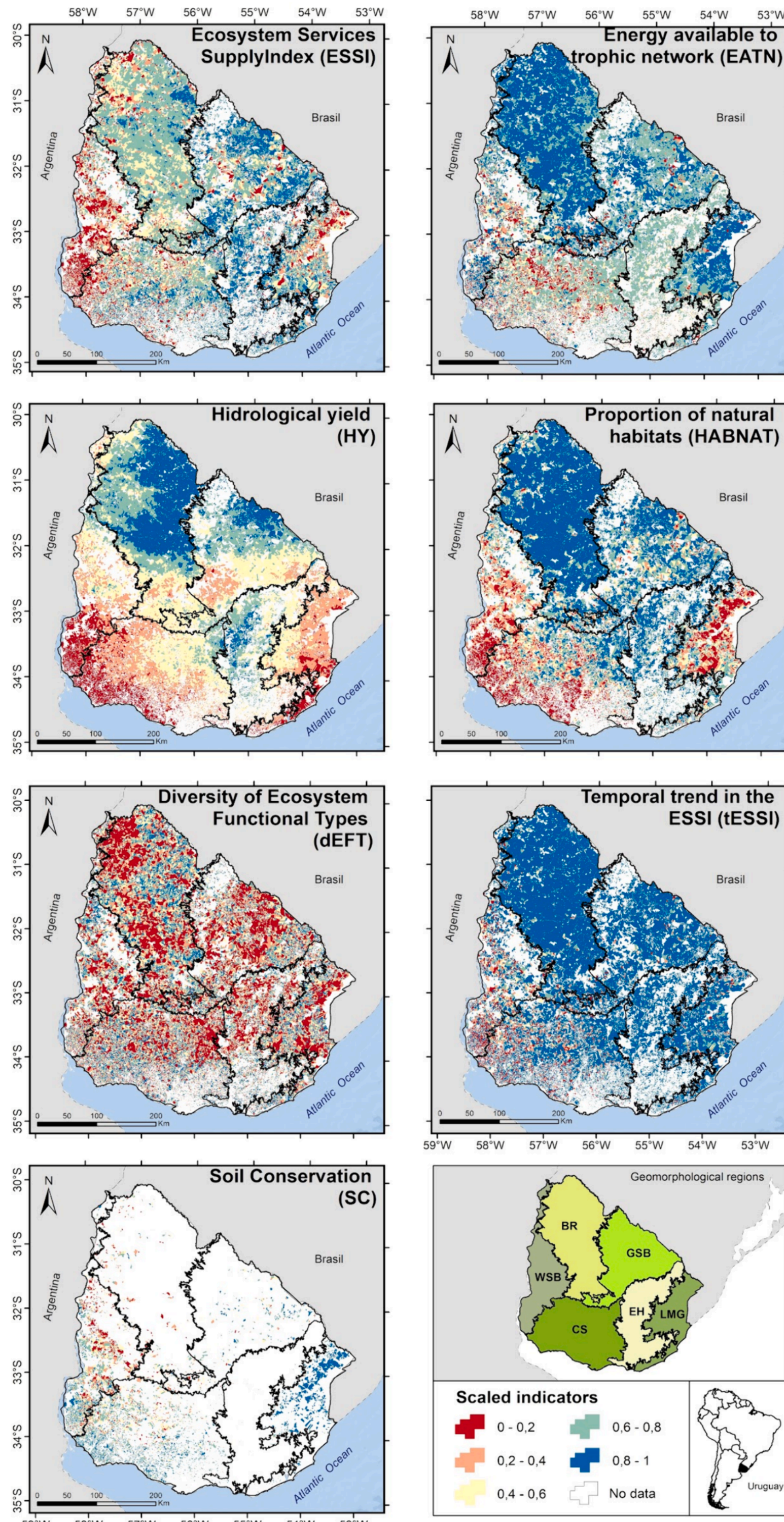

Fig. 2. Maps showing the spatial patterns of the scaled values (0–1) of the 7 indicators studied: dEFT: diversity of Ecosystem Functional Types; HABNAT: Proportion of Natural Habitats; ESSI: Ecosystem Services Supply Index; tESSI: temporal trend in the Ecosystem Services Supply Index; EATN: Energy Available to Trophic Network (1-Human Appropriation of Net Primary Production); HY: hydrological yield; SC: Soil Conservation. Inset showed the distribution of the 6 geomorphological regions considered: Western sediment basin (WSB), Crystalline shield (CS), Basaltic region (BR), Gondwanic Sediment basin (GWB), Eastern hills (EH), and Lagoon

Merin graben (LMG). White areas correspond to cadastral units with no data due to the proportion of tree plantations and/or non-agricultural soils (for the SC indicator).

some of the regions were particularly evident (i.e. for natural habitats between the Basaltic region and the Western Sediment Basin and Gondwanic Sediment Basin or between this last unit and the Lagoon Merin Graben). White areas in the maps corresponded to cadastral units dominated by tree plantations or, in the case of Soil Conservation, to non-agricultural soils (Fig. 2).

Interestingly, the regional patterns differ among indicators suggesting a low redundancy. The low redundancy was evident when the spatial correlation between indicators was analyzed (Fig. 3). The highest correlation among indicators observed (HABNAT vs HY) was 0.62, indicating that only 38 % of the variance of one of the indicators was associated with the other. When the correlation among indicators was analyzed for each geomorphological region, the pattern was similar to that observed for the whole country. Only in three regions (Basaltic Region, Crystalline Shield, and Western Sediment Basin) correlation coefficients higher than 0,50 were observed (between HABNAT vs tESSI, ESSI vs tESSI, HY vs HABNAT and HABNAT vs ESSI) (Figure S4, Supplementary material). We performed an analysis on a subset of the data to avoid potential effects of spatial correlation and the results showed basically the same patterns (Figure S5, Supplementary material).

For each region and type of farm, it is possible to define the median environmental performance (Fig. 4). Each farm had also an individual environmental "signature" (data not shown). The p25 and p75 percentile bars of each petal in Fig. 4 exhibited significant variability across all the dimensions considered. Such differences are mainly associated with the dominant land use (cropping vs livestock production) (Fig. 4, see table in Supplementary material with the non-scaled values of the indicators). Livestock farms showed the best performance and also the lowest variability for most indicators (Fig. 4). Though cropland farms presented lower median values than livestock farms, the widest variability range displayed for, particularly ESSI, tESSI, EATN, and dEFT indicates that such cadastral units have the potential of reaching similar environmental performances than less intensified (Livestock and Mixed farms) areas. The performance of the different types of farms was similar among geomorphological regions, except for the Cropland Farms of the Lagoon Merin graben where the dominance of irrigated rice fields and a flat landscape determined important differences with the rainfed agriculture that dominated the rest of the country (Fig. 4). When all the cadastral units within a region are considered, the Basaltic Region, the area of the country with the highest proportion of natural grassland (Baeza et al., 2022) and the lowest level of habitat fragmentation (Mello et al., 2023), showed the best performance (Fig. 4), mainly due to the high proportion of "livestock farms" (LF) on it (96 % of the cadastral units were classified as LF). The region where croplands were dominant, the Western Sedimentary Basin, showed the lowest performance in environmental terms due, mainly, to the lowest proportion of natural habitats and the low values of the ESSI.

In absolute terms, the proportion of Natural Habitats (HABNAT) varied between 9.2 (p5) and 99.8 % (p95). The median value for Livestock farms was 86 %, substantially higher than the median of Cropland Farms (15.4 %) and Mixed farms (33.1 %). As expected, the diversity of Ecosystem Functional Types (dEFT) augmented with the size of the cadastral unit and reached a plateau around 2500 ha (Figure S6, Supplementary Material). A Michaelis Menten function was able to capture 62 % of the variability. Livestock and mixed farms showed similar dEFT values, being mixed farms more diverse for units lower than 500 ha and livestock farms more diverse in farms above this size. On the other hand, cropland farms showed lower dEFT values for all farm sizes (Figures S6 and S7). The dispersion around these models provides a relative measure of the functional diversity of a cadastral unit. Units above the fitted

Fig. 3. Pearson coefficient of correlation between indicators considering the whole cadastral units. ESSI: Ecosystem Services Supply Index; EATN: Energy Available to Trophic Network; HY: Hydrological yield; HABNAT: proportion of natural habitats; dEFT: Diversity of Ecosystem Functional Types; tESSI: temporal trend in the Ecosystem Services Supply Index; SC: Soil conservation.

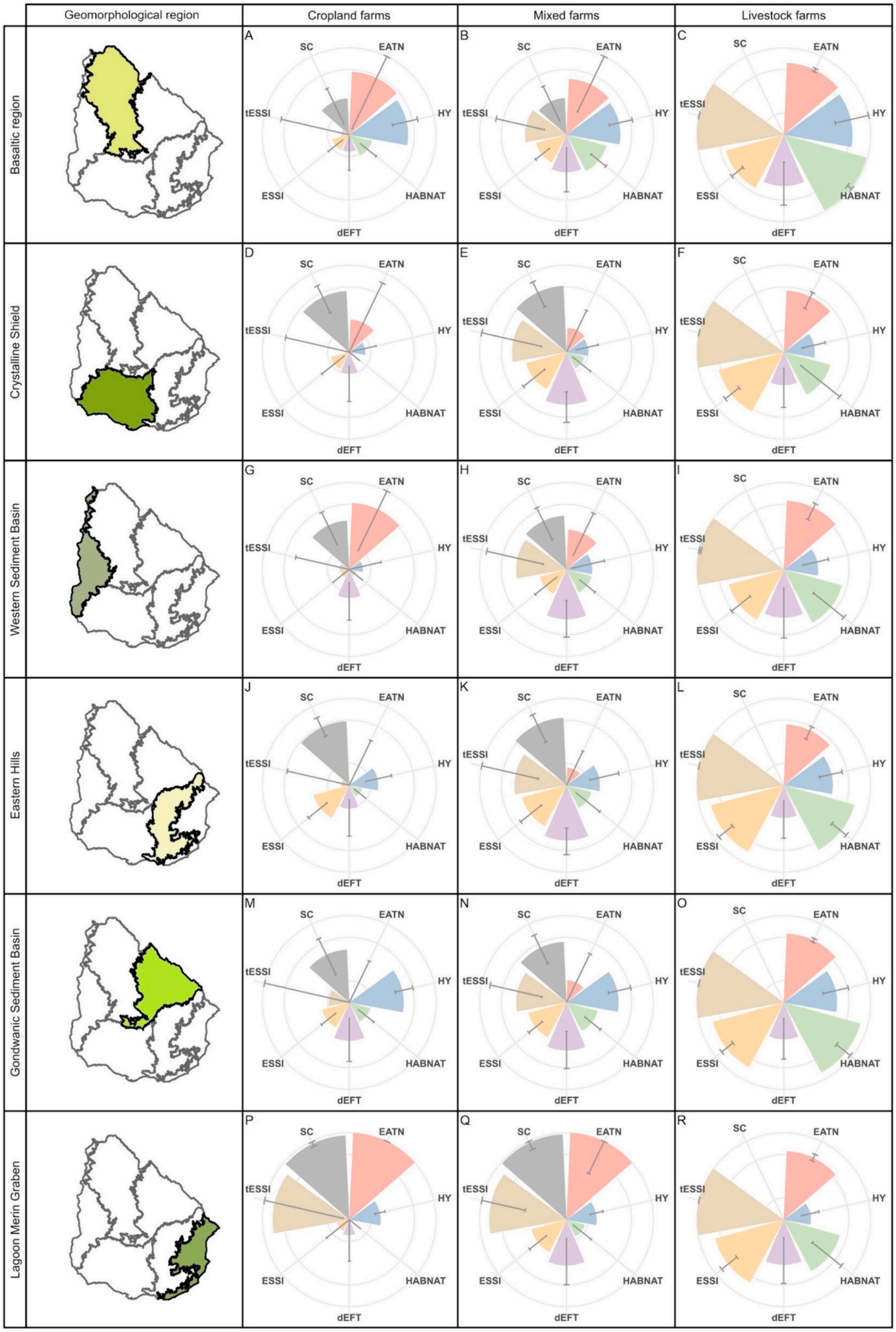

Fig. 4. Flower plots showing the environmental performance of cadastral units grouped by productive use (Livestock, mixed and cropland farms) in the columns and geomorphological region (also shown with maps) in the rows. The median value of each indicator is shown in each petal of the flowers, where the error bars represent the 25th (lower bar) and 75th (upper bar) percentile values of the corresponding indicator. The soil conservation (SC) indicator was not calculated for grassland areas, thus the SC petal is absent in Livestock farm plots. ESSI: Ecosystem Services Supply Index; EATN: Energy Available to Trophic Network; HY: Hydrological yield; HABNAT: proportion of natural habitats; dEFT: Diversity of Ecosystem Functional Types; tESSI: temporal trend in the Ecosystem Services Supply Index; SC: Soil conservation.

curve were more diverse than the mean for a given size. For units between 400 and 500 ha, only 2 % of the farms above the mean dEFT value were Cropland farms (Figure S7).

ESSI varied in absolute values between 0.55 and 0.78 (p5 and p95, respectively). The highest values, again, correspond to Livestock farms. However, Cropland Farms may reach high values of this indicator mainly in areas with double cropping systems, service (cover) crops or a high proportion of sowed pastures. tESSI were also lower in Cropland Farms than in Livestock farms, probably because during the period analyzed an important land cover change (grasslands transformed into croplands and/or reduction of the proportion of sowed pastures) took place across the country (Baeza and Paruelo, 2020; Baeza et al., 2022). However, the dispersion of tESSI values (p75-p25) in Cropland Farms was particularly high, meaning that some cropland farms experienced positive changes in ES supply. A survey of the management actions taken in the Cropland farms with higher ES supply would guide sustainability transitions for the agricultural sector.

The Energy Available for the trophic network (EATN), the complement of the widely used HANPP, presents interesting patterns. It varied from 52 (p5) to 90 % (p95) for the whole data set. In general, the higher proportion of crops in the cadastral units was associated with a lower EATN (Cropland < Mixed < Livestock Farms), due to the appropriation of part of the PPN as harvest. In some cases, due to management (mainly fertilization and irrigation), the actual NPP in cropland areas may be greater than the potential NPP. Consequently the remaining biomass in the system, even with a high harvest index, can be large.

The Hydrological Yield showed large differences among regions, due mainly to differences in soil depth and slopes. Median absolute values for the Basaltic region were 32 % and 22 % for the Western Sedimentary region. Within each region, land use also generates differences with higher median values of HY for Livestock farms than for Cropland farms (ranging from 32-28 % for the Basaltic region and 24–19 % for the Western Sedimentary region, respectively).

Finally, the Soil Conservation indicator for cropland areas (Cropland and mixed farms) showed more variation between geomorphological regions than among productive uses. It had the highest values (least soil loss) in the Lagoon Merin, followed by Eastern Hills and Crystalline Shield. On the other hand, the SC values were lower in the Gondwanic Sediment Basin, followed by the Western Sediment Basin and the Basaltic region, with the lowest values (Fig. 4). In absolute terms, the soil loss estimations from the RUSLE calculation ranged from 0 to 16.7 (tn/ha.year), with a median value of 1.4 (tn/ha.year). In the region where croplands are dominant, the Western Sediment Basin, the median soil loss rate was 1.9 (tn/ha/year) and ranged from 1.25 (p25) to 3 (p75) (tn/ha.year) in cropland farms ($n = 2693$). For the Gondwanic Sediment Basin, the median soil loss rate in cropland farms ($n = 149$) was 1.74 (tn/ha.year) and ranged from 0.84 (p25) to 2.8 (p75) (tn/ha.year). Despite not having many cropland farms ($n = 136$), the Basaltic Region had the highest soil loss rate in these fields, with a median soil loss rate of 2.5 (tn/ha.year) ranging from 1.8 (p25) to 3.7 (p75) (Table S1), mainly due to the combination of shallow soils and high slopes which make the soils more susceptible to erosion.

4. Applications and limitations

Describing the environmental performance at the farm level based on synoptic indicators may represent a major step in assessing the sustainability of the agricultural sector. Some innovative aspects of our study include: a. the extension of the analysis (the entire country), b. the resolution of both the environmental description and the summary of the information (a Landsat or MODIS pixel and the cadastral unit), c. the multidimensional characterization of environmental performance (by including biodiversity, C and water dynamics, and ecosystem services supply), d. the quantification of the complementary/redundancy of the indicators evaluated at the resolution of management units, e) the dynamic nature of the indicators and their sensitivity to human interventions (can be calculated every growing seasons and can track changes in land uses/management) and f) the comprehensive characteristic of the study that allow to use the same protocols to calculate the indicators across regions and production systems. Several of the indicators proposed in the literature to measure sustainability are not scalable (it is impossible to spatially cover the whole territory), depend on secondary information (e.g. amount of products and applied dose) or are difficult to monitor over time (Robling et al., 2023). Our results presented spatially explicit indicators, that are independent of secondary information and are updatable over time. Moreover, they are clearly linked to ecological features that determine environmental sustainability (Storkey et al., 2024).

Obviously the approach we took to describe the environmental performance has limitations. Such limitations are, primarily associated to the resolution (both spatial and conceptual) of the remote sensing products, to the correspondence between the cadastral unit and actual farms and with the conceptual models used to derive the indicators. The spatial resolution of those products derived from MODIS images determined that small cadastral units (smaller than aprox. 15 ha) were not considered in the characterization. This may generate biases associated to, for example, the particular types of farmers included due to a potential underrepresentation of the smallholders. Moreover, the cadastral unit does not necessarily represent a farm (a management and commercial unit). Some farms may include more than one cadastral unit or "padrón", but the information on which cadastral unit defines a farm is not available because it is protected by the Statistical Secrecy Law (Law No. 16.616). The conceptual resolution of some of the indicators may also represent a limitation. The product used to derive the percentage of natural habitat (MapBiomas) do not discriminate between types of woodlands or native grassland communities (Lezama et al., 2019). The present study relied only on one descriptor of the temporal changes (tESSI). Some of the trends of other indicators (for example HabNat or EATN) are reflected in changes in the Ecosystem Services Supply Index. However, to include a description of the temporal trends for the whole set of indicators may provide a more comprehensive description of the environmental performance.

We emphasize the importance of describing sustainability in relative terms. The indicators proposed permit the comparison of a farm with its surroundings and rank its performance within a given administrative unit or groups of farms. Moreover, it is possible to track the changes of the different dimensions on a single farm to evaluate the impacts of practices and climatic events or trends. We are not proposing at this stage normative categories, such as the evaluative (good, bad, better, and so on) and the deontic (required, permitted, forbidden, and so on). Defining such categories would require identifying reference situations and a characterization of their temporal and spatial variability. Moreover, actual boundaries or thresholds are a social construction and require a political discussion. It requires the construction of agreements that must incorporate technical aspects, international commitments, the intergenerational perspective, and the vision of social actors with diverse values and interests, among other aspects.

Relative analyses, though, can substantially improve decision-making. We list three potential applications of the indicators presented that are currently in the process of being implemented in Uruguay:

a) Native Grasslands conservation and land use planning. Two Law projects are under discussion at the Representative Chamber of the Uruguayan Parliament. Both are aimed to reduce the transformation, to control and reverse degradation, and to restore native grasslands; the most widespread biome of the Country. HabNat is a critical piece of information to identify native grassland areas, to track temporal changes and to identify critical patches to preserve the connectivity. The ESSI temporal trends (tESSI) become essential to monitor changes in the conservation status and to identify degradation spots. These two indices have been used with high weightings in an exercise aimed to define the conservation status of grasslands in a subregion particularly threatened by land use changes (Staiano et al., 2022). The functional diversity (dEFT) of the grassland surroundings was also used, although with the least weight in the final value.

b) Agroecological Transitions. Uruguay issued in 2017 the Agroecological Act (No. 19.717, https://www.impo.com.uy/bases/leyes/19717–2018). The objective of the norm is to promote agroecological systems of production, distribution and consumption. A critical issue is to identify effective agroecological transition and to devise indicators to monitor them. This is particularly critical for extensive production systems (croplands, livestock, and dairy production). García-Inza et al. (2023) proposed to identify transitions based on changes on 14 dimensions that include, among other, natural habitats conservation, ecosystem services supply, biodiversity conservation, soil conservation and GHG emissions. As a related initiative, a joint effort of the Ministries of Agriculture and Environment, set a working group including the public sector and the academia to define a set of indicators to characterize the Environmental Footprint of the Livestock sector (MA, 2022). This group included in the definition of the Environmental Footprint 5 out of the 7 indicators presented here. A program on Agroecological Transitions in different production systems instrumented by the Ministry of Agriculture and funded by the World Bank ("Sendas Agroecológicas", https://www.gub.uy/ministerio-ganaderia-agricultura-pesca/comunicacion/convocatorias/senda-agroecologica) also incorporate 5 of the indicators proposed here to characterize the environmental baseline (2023) and, later, the magnitude of the changes (2026) on the 146 farms selected.

c) Product typification: The environmental performance of a territorial unit (i.e. a farm or group of farms), evaluated in its multiple dimensions, can be linked to the production generated in that unit (Fig. 5). The combination of the description of environmental performance with the traceability systems in operation in Uruguay in the livestock industry makes it possible to typify products from the point of view of their environmental sustainability. Product typification allows an effective and explicit connection with consumption and, therefore, with the possibility of modifying patterns by differentiating products based on their environmental and social performance.

The use of indicators requires a complex process to legitimize them. A first step is the academic evaluation through the peer review processes and publication in peer-reviewed scientific journals. A second step involves the evaluation of the indicators as technological developments. Such evaluation requires a multidimensional and situated approach. The National Institute of Agricultural Research (INIA by its Spanish acronym) in Uruguay stated a program to evaluate and certify technologies for the agricultural sector (Vasen et al., 2021; (https://www.inia.uy/productos-y-servicios/Productos/Certificacion-de-tecnologias). The process involves the opinion of a panel of stakeholders linked to the academia, the public and private sectors and the users. Five of the indicators presented here have been already certified by INIA (Sierra et al., 2023). However, a critical step in building legitimacy is related to final users' opinion and adoption, an ongoing process in this case.

CRediT authorship contribution statement

José M. Paruelo: Writing – review & editing, Writing – original draft, Methodology, Funding acquisition, Formal analysis, Conceptualization. **Gonzalo Camba Sans:** Writing – review & editing, Writing – original draft, Methodology, Formal analysis, Data curation, Conceptualization. **Federico Gallego:** Writing – review & editing, Writing –

Fig. 5. A scheme of product typification at the farm level based on the environmental performance assessment. The Ministry of Livestock, Agriculture and Fisheries, through the National Livestock Information System (SNIG, for its acronym in Spanish), has a traceability protocol that associates livestock stocks with the farm (and farmer) from which they came. Combining the traceability systems with indicators described would allow one to label the final product making it possible to associate the consumption phase with the specific environmental performance at farm. Such association would generate a signal that may allow for a retrocontrol on the production phase (red arrow).

original draft, Methodology, Formal analysis, Data curation, Conceptualization. **Pablo Baldassini:** Writing – review & editing, Writing – original draft, Methodology, Formal analysis, Data curation, Conceptualization. **Luciana Staiano:** Writing – review & editing, Methodology, Formal analysis, Data curation, Conceptualization. **Santiago Baeza:** Writing – review & editing, Methodology, Funding acquisition, Formal analysis, Conceptualization. **Hernán Dieguez:** Writing – review & editing, Formal analysis, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data area available at https://redata.anii.org.uy/dataset.xhtml?persistentId=doi:10.60895/redata/UCZZKW.

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Appendix A. Supplementary data

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References

Alcaraz-Segura, D., Paruelo, J.M., Cabello, J., 2006. Identification of current ecosystem functional types in the Iberian Peninsula. Glob. Ecol. Biogeogr. 15, 200–212. https://doi.org/10.1111/j.1466-822X.2006.00215.x.

Alcaraz-Segura, D., Paruelo, J.M., Epstein, H.E., Cabello, J., 2013. Environmental and human controls of ecosystem functional diversity in temperate South America. Remote Sens. (Basel) 5, 127–154. https://doi.org/10.3390/rs5010127.

Almagro, A., Thomé, T.C., Colman, C.B., Pereira, R.B., Junior, J.M., Rodrigues, D.B.B., Oliveira, P.T.S., 2019. Improving cover and management factor (C-factor) estimation using remote sensing approaches for tropical regions. Int. Soil Water Conserv. Res. 7, 325–334. https://doi.org/10.1016/j.iswcr.2019.08.005.

Baeza, S., Paruelo, J. M., Altesor, A., 2006. Caracterización funcional de la vegetación del Uruguay mediante el uso de sensores remotos. Interciencia. 31: 382-388. Available at: http://ve.scielo.org/scielo.php?script=sci_arttext&pid=S0378-18442006000500011&lng=es&nrm=iso>. ISSN 0378-1844.

Altesor, A., Oesterheld, M., Leoni, E., Lezama, F., Rodríguez, C., 2005. Effect of grazing exclosure on community structure and productivity of a Uruguayan grassland. Plant Ecology 179, 83–91.

Baeza, S., Paruelo, J.M., 2018. Spatial and temporal variation of human appropriation of net primary production in the Rio de la Plata grasslands. ISPRS J. Photogramm. Remote Sens. 145, 238–249. https://doi.org/10.1016/j.isprsjprs.2018.07.014.

Baeza, S., Paruelo, J.M., 2020. Land use/land cover change (2000–2014) in the Rio de la Plata grasslands: an analysis based on MODIS NDVI time series. Remote Sens. (Basel) 12, 381. https://doi.org/10.3390/rs12030381.

Baeza, S., Vélez-Martin, E., De Abelleyra, D., Banchero, S., Gallego, F., Schirmbeck, J., Verón, S., Vallejos, M., Weber, E., Oyarzabal, M., Barbieri, A., Petek, M., Guerra Lara, M., Sarrailhé, A., Baldi, G., Bagnato, C., Bruzzone, L., Ramos, S., Hasenack, H., 2022. Two decades of land cover mapping in the Río de la Plata grassland region: The MapBiomas Pampa initiative. Remote Sens. Appl.: Soc. Environ. 28, 100834 https://doi.org/10.1016/j.rsase.2022.100834.

Bagnato, C., Alcaraz-Segura, D., Cabello, J., Berbery, H., Epstein, H., Jobbágy, E., Paruelo, J., 2024. Global Ecosystem Functional Types..

Baldassini, P., Baethgen, W., Camba-Sans, G., Quincke, A., Pravia, V.M., Terra, J., Macedo, I., Piñeiro, G., Paruelo, J.M., 2023. Carbon stocks and potential sequestration of Uruguayan soils. A road map to a comprehensive characterization of temporal and spatial changes to assess Carbon footprint. Front. Sustain. Food Syst. 7, 1045734. https://doi.org/10.3389/fsufs.2023.1045734.

Bergez, J.E., Bethinger, A., Bockstaller, C., Cederberg, C., Ceschia, E., Guilpart, N., Lange, S., Müller, F., Reidsma, P., Riviere, C., Schader, C., Therond, O., van Der Werf, H.M., 2022. Integrating agri-environmental indicators, ecosystem services assessment, life cycle assessment and yield gap analysis to assess the environmental sustainability of agriculture. Ecol. Ind. 141, 109107 https://doi.org/10.1016/j.ecolind.2022.109107.

Bommarco, R., Kleijn, D., Potts, S.G., 2013. Ecological intensification: harnessing ecosystem services for food security. Trends Ecol. Evol. 28, 230–238. https://doi.org/10.1016/j.tree.2012.10.012.

Burkhard, B., Kroll, F., Nedkov, S., Müller, F., 2012. Mapping ecosystem service supply, demand and budgets. Ecol. Ind. 21, 17–29. https://doi.org/10.1016/j.ecolind.2011.06.019.

Cabello, J., Fernández, N., Alcaraz-Segura, D., Oyonarte, C., Pineiro, G., Altesor, A., Delibes, M., Paruelo, J.M., 2012. The ecosystem functioning dimension in conservation: insights from remote sensing. Biodivers. Conserv. 21, 3287–3305. https://doi.org/10.1007/s10531-012-0370-7.

Carrasco-Letelier, L., Beretta, A.N., 2017. Soil erosion by water estimated for 99 Uruguayan basins. Ciencia e Investigación Agraria: Revista Latinoamericana De Ciencias De La Agricultura 44 (2), 184–194.

Cazorla, B.P., Cabello, J., Peñas, J., Garcillán, P.P., Reyes, A., Alcaraz-Segura, D., 2021. Incorporating ecosystem functional diversity into geographic conservation priorities using remotely sensed ecosystem functional types. Ecosystems 24, 548–564. https://doi.org/10.1007/s10021-020-00533-4.

Costanza, R., Norton, B. G., & Haskell, B. D. (Eds.). (1992). Ecosystem health: new goals for environmental management. Island Press, pp. 269.

Daily, G.C., Matson, P.A., 2008. Ecosystem services: From theory to implementation. Proc. Natl. Acad. Sci. 105, 9455–9456. https://doi.org/10.1073/pnas.0804960105.

Dale, V., Beyeler, S., 2001. Challenges in the development and use of ecological indicators. Ecol. Ind. 1, 3–10. https://doi.org/10.1016/S1470-160X(01)00003-6.

Davis, A.S., Hill, J.D., Chase, C.A., Johanns, A.M., Liebman, M., 2012. Increasing Cropping System Diversity Balances Productivity. Profitability and Environmental Health. Plos One. 7 (10), e47149.

De Groot, R.S., Alkemade, R., Braat, L., Hein, L., Willemen, L., 2010. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecol. Complex. 7, 260–272. https://doi.org/10.1016/j.ecocom.2009.10.006.

Díaz, S., Demissew, S., Carabias, J., Joly, C., Lonsdale, M., Ash, N., Larigauderie, A., Adhikari, J. r., Arico, S., Báldi, A., Bartuska, A., Baste, I. A., Bilgin, A., Brondizio, E., Chan, K., Figueroa, V. E., Duraiappah, A., Fischer, M., Hill, R., Koetz, T., Leadley, P., Lyver, P., Mace, G. M., Martin-Lopez, B., Okumura, M., Pacheco, D., Pascual, U., Pérez, E. S., Reyers, B., Roth, E., Saito, O., Scholes, R. J., Sharma, N., Tallis, H., Thaman, R., Watson, R., Yahara, T., Hamid, Z. A., Akosim, C., Al-Hafedh, Y., Allahverdiyev, R., Amankwah, E., Asah, S. T., Asfaw, Z., Bartus, G., Brooks, L. A., Caillaux, J., Dalle, g., Darnaedi, D., Driver, A., Erpul, G., Escobar-Eyzaguirre, P., Failler, P., Mokhtar-Fouda, A. M., Fu, B., Gundimeda, H., Hashimoto, S., Homer, F., Lavorel, S., Lichtenstein, G., Mala, W. A., Mandivenyi, W., Matczak, P., Mbizvo, C., Mehrdadi, M., Metzger, J. P., Mikissa, J. B., Moller, H., Mooney, H. A., Mumby, P., Nagendra, H., Nesshover, C., Oteng-Yeboah, A. A., Pataki, G., Roué, M., Rubis, J., Schultz, M., Smith, P., Sumaila, R., Takeuchi, K., Thomas, S., Verma, M., Yeo-Chang, Y., Zlatanova, D. 2015.. The IPBES Conceptual Framework—connecting nature and people. Current Opinion in Environmental Sustainability. 14, 1-16. https://doi.org/10.1016/j.cosust.2014.11.002.

Dillon, E., Hennessy, T., Buckley, C., Donnellan, T., Hanrahan, K., Moran, B., Ryan, M., 2016. Measuring progress in agricultural sustainability to support policy-making. Int. J. Agric. Sustain. 14, 31–44. https://doi.org/10.1080/14735903.2015.1012413.

Donnelly, A., Jones, M., O'Mahony, T., Byrne, G., 2007. Selecting environmental indicator for use in strategic environmental assessment. Environ. Impact Assess. Rev. 27, 161–175. https://doi.org/10.1016/j.eiar.2006.10.006.

Elnashar, A., Zeng, H., Wu, B., Fenta, A.A., Nabil, M., Duerler, R., 2021. Soil erosion assessment in the Blue Nile Basin driven by a novel RUSLE-GEE framework. Sci. Total Environ. 793, 148466 https://doi.org/10.1016/j.scitotenv.2021.148466.

Fahrig, L., Baudry, J., Brotons, L., Burel, F.G., Crist, T.O., Fuller, R.J., Siriwardena, G.M., Martin, J.L., 2011. Functional landscape heterogeneity and animal biodiversity in agricultural landscapes. Ecol. Lett. 14, 101–112. https://doi.org/10.1111/j.1461-0248.2010.01559.x.

Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. Ecol. Econ. 68, 643–653. https://doi.org/10.1016/j.ecolecon.2008.09.014.

Foley, J. A., Ramankutty, N., Brauman, K. A., Cassidy, E. S., Gerber, J. S., Johnston, M., Mueller, N. D., O'Connell, C., Ray, D, K., West, P. C., Blazer, C., Bennet, E. M., Carpenter, S, R., Hill, J., Monfreda, C., Polanski, S., Rockström, J., Sheehan, J., Siebert, S,. Tilman, D., Zaks, D. P., 2011. Solutions for a cultivated planet. Nature. 478, 337-342. https://doi.org/10.1038/nature10452.

Funk, C., Peterson, P., Landsfeld, M., Pedreros, D., Verdin, J., Shukla, S., Husak, G., Rowland, J., Harrison, L., Hoell, A., Michaelsen, J., 2015. The climate hazards infrared precipitation with stations—a new environmental record for monitoring extremes. Sci. Data 2, 1–21. https://doi.org/10.1038/sdata.2015.66.

Gallego, F., Camba-Sans, G.C., Di Bella, C.M., Tiscornia, G., Paruelo, J.M., 2023b. Performance of real evapotranspiration products and water yield estimations in Uruguay. Remote Sens. Appl.: Soc. Environ. 32, 101043 https://doi.org/10.1016/j.rsase.2023.101043.

Gallego, F., Bagnato, C., Baeza, S., Camba-Sans, G., Paruelo, J. M., 2023a. Río de la Plata Grasslands: How Did Land-Cover and Ecosystem Functioning Change in the Twenty-First Century?. In South Brazilian grasslands: ecology and conservation of the Campos Sulinos (pp. 475-493). Cham: Springer International Publishing. https://doi.org/10.1007/978-3-031-42580-6_18.

García-Inza, G., Paruelo, J. M., Zopolo R., 2023. Aportes Científicos y Tecnológicos del INIA-Uruguay a las Trayectorias Agroecológicas. 1a ed - Ciudad Autónoma de Buenos Aires. Fundación CICCUS. Available at: http://www.ainfo.inia.uy/digital/bitstream/item/17223/1/Aportes-cientificos-y-tecnologicos-del-INIA-a-trayectorias-agroecologicas.pdf.

Gaudin, A.C., Tolhurst, T.N., Ker, A.P., Janovicek, K., Tortora, C., Martin, R.C., Deen, W., 2015. Increasing crop diversity mitigates weather variations and improves yield stability. PLoS One 10 (2), e0113261.

Gil, J.D.B., Reidsma, P., Giller, K., Todman, L., Whitmore, A., van Ittersum, M., 2019. Sustainable development goal 2: Improved targets and indicators for agriculture and food security. Ambio 48, 685–698. https://doi.org/10.1007/s13280-018-1101-4.

Gorelick, N., Hancher, M., Dixon, M., Ilyushchenko, S., Thau, D., Moore, R., 2017. Google Earth Engine: Planetary-scale geospatial analysis for everyone. Remote Sens. Environ. 202, 18–27. https://doi.org/10.1016/j.rse.2017.06.031.

Gurr, G. M., Lu, Z., Zheng, X., Xu, H., Zhu, P., Chen, G., Yao, X., Cheng, J., Zhu, Z., Catindig, J. L., Villareal, S., Chien, H. O., Cuong, L. Q., Channoo, C., Chengwattana, N., Lan, L. P., Hai, l. H., Chaiwong, J., Nicol, H. I., Perovic, D. J., Wratten, S. D., Heong, K. L., 2016. Multi-country evidence that crop diversification promotes ecological intensification of agriculture. Nature Plants. 2, 1-4. https://doi.org/10.1038/nplants.2016.14.

Haberl, H., Erb, K. H., Plutzar, C., Fischer-Kowalski, M., Krausmann, F., 2007. Human appropriation of net primary production (HANPP) as indicator for pressures on biodiversity. In: Sustainability indicators: A scientific assessment (Hák, T., Moldan, B., Dahl, A. L., eds). 67, 271-288.

Haberl, H., Wackernagel, M., Krausmann, F., Erb, K.H., Monfreda, C., 2004. Ecological footprints and human appropriation of net primary production: a comparison. Land Use Policy 21, 279–288. https://doi.org/10.1016/j.landusepol.2003.10.008.

Haines-Young, R., Potschin, M., 2010. The links between biodiversity, ecosystem services and human well-being. Ecosystem Ecology: a new synthesis, 1, 110-139. In: Raffaelli, D., Frid, C. (Eds.), Ecosystem Ecology: A New Synthesis. BES Ecological Reviews Series. CUP Cambridge (Ch.7).

Hayati, D., Ranjbar, Z., Karami, E., 2011. Measuring agricultural sustainability. Biodiversity, Biofuels, Agroforestry and Conservation Agriculture. vol. 5. p 73-100. .

Hengl, T., Gupta, S., 2019. Soil Water Content (Volumetric %) for 33 kPa and 1500 kPa Suctions Predicted at 6 Standard Depths (0, 10, 30, 60, 100 and 200 Cm) at 250 m Resolution. Version v0, 1. Recovery from: https://zenodo.org/record/2784001 (Accessed 15 October 2022).

Hengl, T., Wheeler, I., (2018). Soil organic carbon content in x 5 g/kg at 6 standard depths (0, 10, 30, 60, 100 and 200 cm) at 250 m resolution (Version v02) [Data set]. Zenodo. 10.5281/zenodo.1475457 (Accessed 15 October 2022).

Hengl, T., (2018). Soil texture classes (USDA system) for 6 soil depths (0, 10, 30, 60, 100 and 200 cm) at 250 m (Version v02) [Data set]. Zenodo. 10.5281/zenodo.1475451 (Accessed 15 October 2022).

Jørgensen, S. E., Xu, F. L., Salas, F., Marques, J. C., 2005. Application of indicators for the assessment of ecosystem health. In: Handbook of ecological indicators for assessment of ecosystem health. 2, 5-65. Sven Jørgensen, Liu Xu, Robert Costanza (eds). CRC Press, pp 469.

Kehoe, L., dos Reis, T. N., Meyfroidt, P., Bager, S., Seppelt, R., Kuemmerle, T., Berenguer, E., Clark, M., Davis, K. F., Ermgassen, E., Farrell, K N., Friis, C., Haberl, H., Kastner, T., Murtough, K. L., Persson, U P., Romero-Muñoz, A., O'Connell, C., Sch€afer, V. V., Virah-Sawmy, M., de Waroux, Y. P., Kiesecker, J., 2020. Inclusion, transparency, and enforcement: How the EU-Mercosur trade agreement fails the sustainability test. One Earth. 3, 268-272. https://doi.org/10.1016/j.oneear.2020.08.013.

Kremen, C., Iles, A., Bacon, C., 2012. Diversified farming systems: an agroecological, systems-based alternative to modern industrial agriculture. Ecol. Soc. 17 (4) https://doi.org/10.5751/ES-05103-170444.

Laterra, P., Orúe, M.E., Booman, G.C., 2012. Spatial complexity and ecosystem services in rural landscapes. Agr Ecosyst Environ 154, 56–67. https://doi.org/10.1016/j.agee.2011.05.013.

Latruffe, L., Diazabakana, A., Bockstaller, C., Desjeux, Y., Finn, J., Kelly, E., Ryan, M., Uthes, S., 2016. Measurement of sustainability in agriculture: a review of indicators. Studies in Agricultural Economics 118, 123–130.

Lezama, F., Pereira, M., Altesor, A., Paruelo, J.M., 2019. Grasslands of Uruguay: classification based on vegetation plots. Phytocoenologia. https://doi.org/10.1127/phyto/2019/0215.

Liang, S., 2000. Numerical experiments on the spatial scaling of land surface albedo and leaf area index. Remote Sens. Rev. 19, 225–242. https://doi.org/10.1080/02757250009532420.

Lynch, J., Skirvin, D., Wilson, P., Ramsden, S., 2018. Integrating the economic and environmental performance of agricultural systems: A demonstration using Farm Business Survey data and Farmscoper. Sci. Total Environ. 628, 938–946. https://doi.org/10.1016/j.scitotenv.2018.01.256.

MA, 2022. Informe final de la Huella de la Ganadería en Uruguay, Ministerio de Ambiente. Available at: .

McNaughton, S.J., Oesterheld, M., Frank, D.A., Williams, K.J., 1989. Ecosystem-level patterns of primary productivity and herbivory in terrestrial habitats. Nature 341, 142–144. https://doi.org/10.1038/341142a0.

Mello, A.L., Lezama, F., Baeza, S., 2023. Patrones y controles regionales de la fragmentación de pastizales naturales en Uruguay. Ecosistemas. 32, 2534. https://doi.org/10.7818/ECOS.2534.

Millennium Ecosystem Assessment, 2005. Ecosystems and human well-being: Biodiversity Synthesis. World Resource Institute, Washington, DC,USA. Available at: https://www.millenniumassessment.org/documents/document.354.aspx.pdf.

Modernel, P., Dogliotti, S., Alvarez, S., Corbeels, M., Picasso, V., Tittonell, P., Rossing, W. A., 2018. Identification of beef production farms in the Pampas and Campos area that stand out in economic and environmental performance. Ecol. Ind. 89, 755–770. https://doi.org/10.1016/j.ecolind.2018.01.038.

Moran, M.S., Jackson, R.D., 1991. Assessing the spatial distribution of evapotranspiration using remotely sensed inputs. J. Environ. Qual. 20, 725–737. https://doi.org/10.2134/jeq1991.00472425002000040003x.

Nelson, E. J., Daily, G. C., 2010. Modeling ecosystem services in terrestrial systems. *F1000 biology reports*, *2*. Available at: https://www.ncbi.nlm.nih.gov/pmc/articles/PMC2990460/.

Nelson, E., Mendoza, G., Regetz, J., Polasky, S., Tallis, H., Cameron, D., Chan, K.M., Daily, G.C., Goldstein, J., Kareiva, P.M., Lonsdorf, E., Naidoo, R., Ricketts, T.H., Shaw, M., 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. Front. Ecol. Environ. 7, 4–11. https://doi.org/10.1890/080023.

Oesterheld, M., Loreti, J., Semmartin, M., Paruelo, J.M., 1999. Grazing, fire, and climate effects on primary productivity of grasslands and savannas. In: Walker, L. (Ed.), Ecosystems of disturbed ground. Elsevier, Oxford, UK, pp. 287–306.

Pacini, C., Wossink, A., Giesen, G., Vazzana, C., Huirne, R., 2003. Evaluation of sustainability of organic, integrated and conventional farming systems: a farm and field-scale analysis. Agr. Ecosyst. Environ. 95, 273–288. https://doi.org/10.1016/S0167-8809(02)00091-9.

MapBiomas Pampa 2023. Project MapBiomas Trinational Pampa – Collection 3 of the Annual Land Cover and Land Use Maps in Trinational Pampa. Available at:: https://pampa.mapbiomas.org/.

Panario, D., Gutiérrez, O., Sánchez Bettucci, L., Peel, E., Oyhantçabal, P., Rabassa, J., 2014. Ancient landscapes of Uruguay. Gondwana Landscapes in southern South America: Argentina, Uruguay and southern Brazil, 161-199.In: Rabassa, J., Ollier, C. (eds) Gondwana Landscapes in southern South America. Springer Earth System Sciences. Springer, Dordrecht. https://doi.org/10.1007/978-94-007-7702-6_8.

Paruelo, J.M., 2008. La caracterización funcional de ecosistemas mediante sensores remotos. Available at: Ecosistemas. 17 (3) https://www.revistaecosistemas.net/index.php/ecosistemas/article/view/83.

Paruelo, J.M., Jobbágy, E.G., Sala, O.E., 2001. Current distribution of ecosystem functional types in temperate South America. Ecosystems 4, 683–698. https://doi.org/10.1007/s10021-001-0037-9.

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

Paruelo, J.M., Texeira, M., Staiano, L., Mastrángelo, M., Amdan, L., Gallego, F., 2016. An integrative index of Ecosystem Services provision based on remotely sensed data. Ecol. Ind. 71, 145–154. https://doi.org/10.1016/j.ecolind.2016.06.054.

Pettorelli, N., Wegmann, M., Skidmore, A., Mücher, S., Dawson, T.P., Fernandez, M., Lucas, R., Schaepman, M.E., Wang, T., O'Connor, B., Jongman, R.H.G., Kempeneers, P., Sonnenschein, R., Leidner, A.K., Böhm, M., He, K.S., Nagendra, H., Dubois, G., Fatoyinbo, T., Hansen, M.C., Paganini, M., de Klerk, H.M., Asner, G.P., Kerr, J.T., Estes, A.B., Schmeller, D.S., Heiden, U., Rocchini, D., Pereira, H.M., Turak, E., Fernandez, N., Lausch, A., Cho, M.A., Alcaraz-Segura, D., McGeoch, M.A., Turner, W., Mueller, A., St-Louis, V., Penner, J., Vihervaara, P., Belward, A., Reyers, B., Geller, G.N., 2016. Framing the concept of satellite remote sensing essential biodiversity variables: challenges and future directions. Remote Sens. Ecol. Conserv. 2, 122–131. https://doi.org/10.1002/rse2.15.

Pettorelli, N., Nagendra, H., Rocchini, D., Rowcliffe, M., Williams, R., Ahumada, J., de Angelo, C., Atzberger, C., Boyd, D., Buchanan, G., Chauvenet, A., Disney, M., Duncan, C., Fatoyinbo, T., Fernandez, N., Haklay, M., He, K., Horning, N., Kelly, N., de Klerk, H., Liu, X., Merchant, N., Paruelo, J.M., Roy, H., Roy, S., Ryan, S., Sollmann, R., Swenson, J., Wegmann, M., 2017. Remote Sensing in Ecology and Conservation: three years on. Remote Sens. Ecol. Conserv. 3, 53–56. https://doi.org/10.1002/rse2.53.

Potter, C.S., Randerson, J.T., Field, C.B., Matson, P.A., Vitousek, P.M., Mooney, H.A., Klooster, S.A., 1993. Terrestrial ecosystem production: A process model based on global satellite and surface data. Global Biogeochem. Cycles 7, 811–841. https://doi.org/10.1029/93GB02725.

Prata, A.J., Caselles, V., Coll, C., Sobrino, J.A., Ottle, C., 1995. Thermal remote sensing of land surface temperature from satellites: Current status and future prospects. Remote Sens. Rev. 12, 175–224. https://doi.org/10.1080/02757259509532285.

Rasmussen, L.V., Bierbaum, R., Oldekop, J.A., Agrawal, A., 2017. Bridging the practitioner-researcher divide: Indicators to track environmental, economic, and sociocultural sustainability of agricultural commodity production. Glob. Environ. Chang. 42, 33–46. https://doi.org/10.1016/j.gloenvcha.2016.12.001.

Renard, K.G., Freimund, J.R., 1994. Using monthly precipitation data to estimate the R-factor in the revised USLE. J. Hydrol. 157, 287–306. https://doi.org/10.1016/0022-1694(94)90110-4.

Renard, K. G., 1997. Predicting soil erosion by water: a guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE). US Department of Agriculture, Agricultural Research Service. Washington, USA.

Richmond, A., Kaufmann, R.K., Myneni, R.B., 2007. Valuing ecosystem services: A shadow price for net primary production. Ecol. Econ. 64, 454–462. https://doi.org/10.1016/j.ecolecon.2007.03.009.

Robling, H., Hatab, H.A., Säll, S., Hansson, H., 2023. Measuring sustainability at farm level – A critical view on data and indicators. Environ. Sustain. Indic. 18 https://doi.org/10.1016/j.indic.2023.100258.

Ruimy, A., Saugier, B., Dedieu, G., 1994. Methodology for the estimation of terrestrial net primary production from remotely sensed data. J. Geophys. Res. Atmos. 99, 5263–5283. https://doi.org/10.1029/93JD03221.

Rusch, G.M., Oesterheld, M., 1997. Relationship between productivity, and species and functional group diversity in grazed and non-grazed Pampas grassland. Oikos 78, 519–526.

Salemi, L.F., Groppo, J.D., Trevisan, R., de Moraes, J.M., de Paula Lima, W., Martinelli, L. A., 2012. Riparian vegetation and water yield: a synthesis. J. Hydrol. 454, 195–202. https://doi.org/10.1016/j.jhydrol.2012.05.061.

Sierra, M. , Sotelo, D. , Negro, C. , Soria, A. , Lapetina, J. 2023. INIA CERTEC.Agro: evaluación de tecnologías con el aporte de usuarios calificados. Revista INIA 75. December 2023. Available at: http://www.inia.uy/Publicaciones/Paginas/publicacionAINFO-64406.aspx.

Sone, J.S., Gesualdo, G.C., Zamboni, P.A., Vieira, N.O., Mattos, T.S., Carvalho, G.A., Oliveira, P.T.S., 2019. Water provisioning improvement through payment for ecosystem services. Sci. Total Environ. 655, 1197–1206. https://doi.org/10.1016/j.scitotenv.2018.11.319.

Staiano, L., Camba-Sans, G.H., Baldassini, P., Gallego, F., Texeira, M.A., Paruelo, J.M., 2021. Putting the Ecosystem Services idea at work: Applications on impact assessment and territorial planning. Environ. Dev. 38, 100570 https://doi.org/10.1016/j.envdev.2020.100570.

Staiano, L., Gallego, F., Altesor, A., Paruelo, J.M., 2022. Where and why to conserve grasslands socio-ecosystems? A spatially explicit participative approach. Front. Environ. Sci. 10, 820449 https://doi.org/10.3389/fenvs.2022.820449.

Stephens, P.A., Pettorelli, N., Barlow, J., Whittingham, M.J., Cadotte, M.W., 2015. Management by proxy? The use of indices in applied ecology. J. Appl. Ecol. 52, 1–6. http://www.jstor.org/stable/43868379.

Storkey, J., Maclaren, C., Bullock, J.M., Norton, L.R., Redhead, J.W., Pywell, R.F., 2024. Quantifying farm sustainability through the lens of ecological theory. Biol. Rev. https://doi.org/10.1111/brv.13088.

Tabatabai, M. A., 1996. Soil organic matter testing: An overview. In: Soil organic matter: analysis and interpretation Magdoff, F. R., Tabatabai, M. A., Hanlon Jr., E. A.. 46, 1-9. https://doi.org/10.2136/sssaspecpub46.c1.

Tscharntke, T., Clough, Y., Wanger, T.C., Jackson, L., Motzke, I., Perfecto, I., Vandermeer, J., Whitbread, A., 2012. Global food security, biodiversity conservation and the future of agricultural intensification. Biol. Conserv. 151, 53–59. https://doi.org/10.1016/j.biocon.2012.01.068.

United Nations. 2023. The Sustainable Development Goals Report 2023. United Nations Publications, United States of America, ISBN: 978-92-1-101460-0. Available at: https://unstats.un.org/sdgs/report/2023/The-Sustainable-Development-Goals-Report-2023.pdf.

Uruguay XXI. Sector agrícola en Uruguay. Promoción de inversiones, exportaciones e imagen país. Available at: https://www.uruguayxxi.gub.uy/uploads/informacion/20c2018b1a2e68514020b55bcd11b62c6874640e.pdf.

Van Bemmelen, J.M., 1891. Ueber die Bestimmungen des Wassers, des Humus, des Schwefels, der in den Colloidalen Silikaten gebunden Kieselsaeuren, des man-gans, u.s.w. im Ackerboden. Landwirtschaftliche Versuch Station 37, 279–290.

Vasen, F., Sierra, M., Paruelo, J.M., Negro, C., Nolla, F., Lapetina, J., Salvagno, M., 2021. Evaluation of technical production in agricultural sciences: a new certification scheme in Uruguay. Agrociencia Uruguay. 25 (2). https://doi.org/10.31285/agro.25.491.

Wischmeier, W. H., Smith, D. D., 1978. Predicting rainfall erosion losses: a guide to conservation planning (No. 537). Department of Agriculture, Science and Education Administration. Washington, DC, USA.

Volante, J.N., Alcaraz-Segura, D., Mosciaro, M.J., Viglizzo, E.F., 2012. Ecosystem functional changes associated with land clearing in NW Argentina. Agriculture, Ecosystems & Environment 154, 12–22.

Yamazaki, D., Ikeshima, D., Tawatari, R., Yamaguchi, T., O'Loughlin, F., Neal, J.C., Sampson, C.C., Kanae, S., Bates, P.D., 2017. A high-accuracy map of global terrain elevations. Geophys. Res. Lett. 44, 5844–5853. https://doi.org/10.1002/2017GL072874.

Verón, S.R., Blanco, L.J., Texeira, M.A., Irisarri, J.G.N., Paruelo, J.M. 2018. Desertification and ecosystem services supply: The case of the Arid Chaco of South America. Journal of Arid Environment. In press. https://doi.org/10.1016/j.jaridenv.2017.11.001.