Comment on: "Back to the future? Conservative grassland management can preserve soil health in the changing landscapes of Uruguay" On the risks of good intentions and poor evidence.

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Abstract. In this article we make comments on some methodological issues and on the general approach of the paper "Back to the future? Conservative grassland management can preserve soil health in the changing landscapes of Uruguay" by Ina Säumel, Leonardo R. Ramírez, Sarah Tietjen, Marcos Barra, and Erick Zagal, Soil 9, 425–442, https://doi.org/10.5194/soil-9-425-2023. We identified several design and methodological problems in this article that may induce potential misinterpretations. Our concerns are related to, first, aspects of the study design and methodology that, in our opinion, introduce biases and critical errors. Secondly, we are concerned about the possible interpretations of a study in the design of policies and development of non-tariff barriers for South American countries.

1 Introduction

The article “Back to the future? Conservative grassland management can preserve soil health in the changing landscapes of Uruguay” written by Ina Säumel, Leonardo R. Ramírez, Sarah Tietjen, Marcos Barra, and Erick Zagal in Soils 9, 425–442, https://doi.org/10.5194/soil-9-425-2023, analysed a set of soil parameters that describe the chemical conditions of the first 10 cm of 101 sampling areas under different land uses and land covers in Uruguay. Upon thorough examination, several deficiencies and considerations were discerned within the article that warrant attention, as they have the potential to give rise to erroneous or misleading interpretations.

2 Our main criticisms

Given the sensitive nature of soil degradation, the potential ramifications of drawing conclusions based on insufficient evidence could lead to misguided interpretations and subsequent actions. Our concerns on the article are related to, first, aspects of the study design and methodology that, in our opinion, introduce biases and critical errors. Secondly, we are concerned about the possible interpretations of a study in the design of policies and development of non-tariff barriers for South American countries.

2.1. Design and methodological issues

2.1.1. <u>About the sampling scheme:</u> The authors indicate that they "randomly selected monitoring sites across the country". Having a randomized design is certainly an advantage. However, no details are given on how this randomization was carried out. This process requires stratification, the definition of a grid, a criterion to discard sites that cannot be accessed, etc. A detail of a possible country-wide stratification alternative is presented in Altesor et al. (2019), in which the authors used land cover maps and a 10x10 km grid where 20 cells were drawn. Within each cell, 5 squares of 1x1 km were randomly chosen and in that area two patches belonging to two different natural grassland communities were sampled by Lezama et al. (2019). Those areas corresponded to a MODIS grid pixel (231x231 m), allowing for a clear localization of the sample. Random sampling implies complex logistics of displacements in the field, especially in areas with low road density as the north of Uruguay. The absence of a description of the design and the coincidence of the location of the sampling areas with the distribution of roads (particularly National No. 5) does not allow to dispel doubts about possible biases in the collection of samples. The coordinates of each of the sampling sites are also not indicated in the study, although it is stated that they were used to locate the soil groups in the Soil Map of Uruguay at scale 1:1,000,000. Aside from the general design, the authors indicated that “We sampled topsoil three times at each land use at the edges of the plot” (Säumel et al., 2023, p. 427)”. Does this mean 3 samples/plot (so the 280 samples are full of pseudo-replications) or 3 different times? Did they use composite samples?

2.1.2. Representativeness of land use types: It is striking that the proportions of land use types sampled in Säumel et al. (2023) study differ strongly from those present in Uruguay, particularly if the sample sites were randomly chosen. According to the latest cartographies, both forestry and native forests are overrepresented. Both occupy 12.5 % (2,204,060 ha according to Baeza et al., 2022) and in the article, the samples of these land covers corresponded to 54% of the total. It is also difficult to make inferences about croplands in Uruguay with no samples in the SW region of the country (i.e., the main cropland area) (Baeza and Paruelo, 2020). This lack of coverage of the main croplands zone of the country is evident by the low number of samples under annual crop use (see Table 1 in the original manuscript). As an example, the authors only had 15 samples of cropland sites (5%), while during 2015/2016 (when the sampling was conducted), there agricultural covers more 31% of Uruguay (https://uruguay.mapbiomas.org/, Collection 1).

2.1.3. Design of the study: Any study that intends to establish differences associated with land use types from spatial sampling must minimize sources of variation excluding the factor to be compared (e.g. soil depth, texture, slope, rockiness, water availability, etc.). Two widely used approaches in observational studies are paired sites or block sampling (e.g. Perelman et al., 2019). The article by Säumel et al. (2023), compared all different land uses against each other, implicitly assuming that the observed differences were only due to land use types, without controlling other factors that also co-varied in space with land uses (e.g. all riverine native forest are located in lowlands). In addition, normally in a paired design aimed to compare land use effects on ecosystems, it is necessary to document that the paired sites sampled are located in an equivalent topographic position, soil types, etc. There is no evidence that this was done in this study. How did the authors control these types of effects? Does the differences in soil characteristics between tree plantations and native forest (or grasslands) resulted from the effect of the land cover or were the consequence of planting trees on soils defined a priori for this use? The design of the study precludes an answer to these questions. This is not a trivial point because those soils defined as "Afforestation priority" in Uruguay have, originally, low fertility and pH. Actually, the authors recognized the importance of soil heterogeneity: "In addition, the lateral heterogeneity of Pampean soils over short distances makes separating geochemical and anthropic signatures difficult (Roca, 2015)" (Säumel et al., 2023, p. 434) a key point that we consider they did not properly contemplate.

In addition, the land use trajectories proposed in Säumel et al. (2023), are oversimplified into four categories. The authors ignored well known land use sequences in the region such as annual crops—grassland returns, rotations with annual crops and perennial pastures, and the cropping history prior to 1986 (the agricultural peak of the 1950s).

2.1.4. Soil type characterization: The only approach used to characterize the site is the soil map of Uruguay at a scale of 1:1,000,000. This does not allow to perceive critical edaphic and topographic differences. The soil group in such Soil Map is defined by the dominant soil type at scale 1:1,000,000. It is well known that the fact that two sites belong to the same soil group does not mean that they have the same soil type (large differences in texture, and other soil properties are common between soils in the same soil group). The assignment of a soil group without any field evidence is, at least, striking given the coarse resolution of the used map. It is surprising that the authors did not evaluate texture to characterize soils at least for two reasons: (i) it is key to give evidence about the comparability between pairs, (ii) it is a property that correlates/explains all other soil properties measured in Säumel et al. (2023), such as soil organic carbon (SOC) and cation exchange capacity (CEC). In addition, in the article there is a strong emphasis on CONEAT units. This is a conceptual error because CONEAT units are not a soil type per se: "CONEAT groups are not strictly basic soil mapping units, but constitute homogeneous areas, defined by their productive capacity in terms of beef, sheep and wool (Art. 65 of Law 13695; https://www.gub.uy/ministerio-ganaderia-agricultura-pesca/politicas-y-gestion/coneat). Again, inside a single CONEAT unit there are normally large variations in soil types and properties.

2.1.5. <u>SOC data</u>: One of the major shortcomings of the paper is the lack of details on the way soil organic carbon (SOC) is reported. First, characterizing SOC changes only from the first 10 cm is, at least, incomplete and risky. Even more, if the particulate and mineral associated fractions of the organic C are not differentiated. Land cover, management, or changes in the relative abundance of plant functional types, may change the vertical distribution of SOC. In fact, within the same land use (native grasslands), paired grazed-ungrazed areas significantly differ in the upper layer distribution of SOC and belowground C inputs (Piñeiro et al., 2009; López-Mársico et al., 2015). Such effects are evident way below 10 cm. Second, the authors reported SOC as a percentage or concentration without indicating if data are on a gravimetric or a volumetric basis. Reporting SOC without considering bulk density precludes any reasonable comparison on an equivalent soil mass (Gifford and Roderick, 2003). No data on bulk density were reported, which is well-known to be affected by land use types evaluated in this work, such as afforestation (Hernandez et al., 2016) or crop production and crop-pasture rotations (Rubio et al., 2021). This is particularly critical if only data for the first cm of the soils are reported. SOC stocks would differ dramatically between soils with different levels of compaction and, hence, differing on bulk density. Also, soils under native forests and tree plantations have an upper layer with mixed soil and plant residues ("litter layer"). Were litter layers excluded/included in the sampled soils? Furthermore, the comparisons of C stocks between riparian forests, tree plantations and grasslands made by Säumel et al. (2023), need to be standardized for two the key factors in determining SOC, C inputs (Net Primary Production) and soil texture (Parton et al., 1994; Schimel et al., 1994; Krull et al., 2001). Riverine forests have a completely different water regime than grasslands or tree plantations and consequently differences in net primary production. Alluvial soils are expected to have profound differences in soil texture compared to upland areas and this will impact on the SOC saturation level of the soil (Chung et al., 2008; Stewart et al., 2007; Mayzelle et al., 2014; Pravia et al., 2017). The saturation level is largely associated with texture, particularly with the fine soil particles fraction (Hassink, 1997; Feng et al., 2013). However, the study ignores the well-known effects of texture on SOC.

2.1.6. <u>Grasslands (GL) categories:</u> Säumel et al. (2023), "... subdivided GL plots according to the intensity of use: (i) undisturbed GLs (without grazing), (ii) partially grazed GLs (with sporadic grazing and low animal charge), and (iii) highly grazed GLs (with high animal charge)" (Note: We assume that "animal charge" means stocking rate). Some of the authors of this reply have been working on grassland ecology in Argentina, Brazil and Uruguay for more than 35 years. We were particularly interested in identifying different grazing situations. Actually, we have compiled a set of ungrazed situations based on an extensive search (Lezama et al., 2014). The sites available were very few. Except for very particular situations, we found it extremely difficult to define the level of grazing intensity in commercial ranches because such information is seldom recorded (but see Lezama and Paruelo, 2022). Aside from how they were defined, it is not clear how the different "categories" of grassland entered into the analysis. However, in the results the authors said that no differences were detected "among different GL subtypes". Several local studies on paired grazed and ungrazed native grasslands have previously showed important changes in SOC stocks that varied according to soil types (Piñeiro et al., 2009, 2010). More recent studies showed that belowground C inputs are heavily impacted by the grazing condition (grazed-ungrazed) (López-Mársico et al., 2023). Säumel et al. (2023), ignored the well documented differences among native grazed or ungrazed grasslands in species composition and vegetation structure. Furthermore, grassland communities of Uruguay have been thoroughly described (Lezama et al., 2019) and mapped (Baeza et al., 2019) in detail, showing that the phytosociological units defined for the country are quite stable under different levels of grazing intensity and degradation (Altesor et al., 2019).

2.1.7. Some additional issues: Some other issues related to the analysis and conclusions preclude clear comparisons with previous studies and/or generalizations. For example:

a. "Total P concentration was determined calorimetrically after microwave-assisted digestion with a Unicam spectrometer at a wavelength of 660 nm." (Säumel et al., 2023, p. 429) --- total soil P is not a fertility parameter, as it has a low correlation with P availability.

b. "The pH of our topsoil samples are mainly in the category of very strongly to extremely acidic and is lowest in TPs (Fig. 6), below the means reported so far (Jobbagy and Jackson, 2003; Céspedes-Payret et al., 2012)." (Säumel et al., 2023, p. 433) --- The authors measured pH in $CaCl_2$ ["Acidity was measured by adding calcium chloride (0.01 M) to the samples at a 2.5:1 proportion, and after shaking and 2 h rest, read with a pH meter (HI2550 meter, Hanna Instruments, USA)." (Säumel et al., 2023, p. 427)], while Céspedes-Payret et al. (2012) and Jobbagy and Jackson (2003) measured it in water, so the results are not comparable. The pH measured in water extractions is more common or standard lab analysis in Uruguay (Hernandez et al., 2016; Beretta-Blanco et al., 2019; Grahmann et al., 2020) and results in higher values than pH measured with $CaCl_2$.

c. We found the conclusions related to the role of riverine forest soils as a sink for trace metals extremely speculative. This kind of analysis must be performed at the catchment level. No evidence is provided on the location of the data reported. Do they correspond to the same basin? Are they physically connected?

2.2. Misleading interpretations and its consequences

2.2.1. Two recommendations that the authors made in the discussion go against grassland conservation:

a) the conversion of grasslands into silvopastoral systems

b) the expansion of native forests and the use of native species in tree plantations

There is profuse evidence that planting trees in open ecosystems, such as Uruguayan grasslands, are not a solution for its restoration nor conservation (Veldman et al., 2015; 2019; among others), although this evidence go against popular beliefs, particularly originated in countries originally covered by native forests.

2.2.2. <u>The article included some generalizations that may lead to some serious misinterpretations:</u>

a): “Our topsoil data indicate that carbon sequestration occurs mainly in the topsoils of native riverine forests that cover less than 5 % of Uruguayan territory.” (Säumel et al., 2023, p. 435) --- We think that Säumel et al. (2023), cannot state that SOC sequestration occurs mainly in the topsoil because: (i) they did not measure SOC stocks or bulk density, (ii) they cannot relate a non-paired, observational study to cause-effect processes (no checking of same soil type besides CONEAT, which includes several soil types), (iii) they did not sample below 10 cm. Moreover, SOC accumulation in riverine areas may result from erosion (natural or anthropic) of SOC formed in upland soils, and therefore correspond to a spatial reallocation of SOC.

b) “Organic carbon content and the exchangeable cations are strongly reduced in the topsoils of GLs, TPs and AC compared to NFs (Figs. 4b, d–h and 5b, d–h).” (Säumel et al., 2023, p. 433). As we stated before, the experimental design does not allow to evaluate reductions or changes in soil cations, because it’s an observational study without any explicit control of the other forming factors (in particular parental material and topography that widely differs among Uruguayan soils) that would allow to use a space-for-time substitution approach necessary to relate observed differences to land use changes.

2.2.3. <u>The discussion links the agricultural production system with widespread soil degradation in Uruguay:</u>

The discussion of the article from Säumel et al. (2023) starts linking the agricultural sector of Uruguay with “Socioeconomic and conventional management practices that drive soil degradation” and the generation of “inputs trap” and “credit or poverty trap”. Even though the characteristics, practices, and structure of the agricultural sector of Uruguay are open to criticism and debate, the article presents no data or evidence to start a discussion about this issue. Aside from the intention of the authors, such comment at the beginning of the discussion may be interpreted as the characterization of the agricultural sector of a country of the Global South by the developed Global North part of the world. The general impression of an independent reader is that soil degradation is widespread in Uruguay, which is not the case, since Uruguay is the country in the region that has the highest area under natural grasslands (Baeza et al., 2022). Moreover, this type of “scientific evidence” on the bad environmental performance of South American countries, spread by scientists of European countries (see i.e. Kerhoe et al., 2020) helps to build non-tariff barriers for primary products and provides excuses to set conditions in international trade agreements. Nevertheless, we strongly agree that Uruguay and other South American countries have major environmental problems. Most of the authors of this reply have been and are involved in documenting, modifying, proposing solutions and/or generating policies in our region, including Uruguay, Argentina, and Brazil (Staiano et al., 2021; Baeza et al., 2022; Overbeck et al., 2022; Paruelo et al., 2022; Gallego et al., 2023; Baldassini et al., 2023; among others).

We are also involved in identifying the underlying causes of the environmental problems in the Global South. National debts, lack of commitment of developed countries with environmental agreements, Nature commodification, land grabbing, the role of multinational financial markets in the agricultural sector are some of the factors promoting land use and land cover changes and degradation, and setting limits to country-level policies. Considering all of the above, we want to stress the risks of simplifying a complex problem that involves a myriad of actors and factors, based on what we believe is not solid scientific evidence.

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