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Relación entre la altura del pastizal, manejo del pastoreo y la avifauna en sistemas ganaderos de la región este de Uruguay

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Dedico este trabajo a Guillermina López Gallinal:

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Resumen

En los pastizales naturales del Río de la Plata, donde la producción ganadera puede coexistir con la biodiversidad, las prácticas efectivas de manejo de pastizales desempeñan un papel crítico. Este estudio combina dos aproximaciones para arrojar luz sobre la relación entre el manejo del ganado y las especies de aves. El primer estudio analiza las respuestas de las aves a diversas variables que son influenciadas por el manejo ganadero. Tomamos datos de campo y analizamos 69 especies de aves en 454 potreros de 46 predios de la región este de Uruguay. Encontramos que los pastizales artificiales tuvieron un impacto negativo en muchas especies de aves, mientras que la presencia de parches de cespitosas (pajonales) está asociada con efectos positivos, especialmente en las especies en peligro de extinción. Los especialistas de pastizal mostraron sensibilidad a la altura del pasto, con respuestas positivas a los parches de cespitosas, pero negativas a la cobertura arbórea. Ajustar las cargas ganaderas para controlar la altura del pasto emerge como una herramienta valiosa para fomentar a los especialistas en pastizales. El escenario ideal para la biodiversidad de aves implica un mosaico de pastizales nativos cortos y altos, intercalados con parches de cespitosas y árboles. Además, encontramos que las respuestas específicas de las especies se vieron influenciadas por los rasgos de las aves, como el tamaño y el comportamiento de forrajeo, lo que mejora la capacidad predictiva para el manejo de los pastizales. El segundo estudio se centró en evaluar la respuesta de las aves a cambios en el manejo en seis predios para aumentar la altura del pasto, lo que mejoraría la ingesta de materia seca por parte del ganado y potencialmente aumentaría la producción ganadera. Aunque la altura promedio del pasto aumentó de 6 cm a 12 cm, las abundancias y la riqueza de especies de aves no variaron. No obstante, dado que las abundancias de las especies evaluadas se mantuvieron constantes al aumentar la altura del pasto (y, por lo tanto, la materia seca disponible para el ganado), concluimos que es posible proteger un subconjunto de aves especialistas de pastizal mientras se aumenta potencialmente la producción ganadera.

Palabras clave: conservación, producción, pastizal natural, aves silvestres, intensificación ecológica

Relationship between grass height, grazing management and avifauna in livestock systems in the eastern region of Uruguay

Summary

In the natural grasslands of the Río de la Plata, where livestock production can coexist with biodiversity, effective pasture management practices play a critical role. This study combines two approaches to shed light on the relationship between livestock management and bird species. The first study analyzes birds' responses to various variables influenced by livestock management. Field data were collected and 69 bird species were analyzed across 454 paddocks on 46 properties in the eastern region of Uruguay. We found that artificial grasslands had a negative impact on many bird species, while the presence of tussock patches (grasslands) was associated with positive effects, especially in endangered species. Grassland specialists exhibited sensitivity to grass height, responding positively to tussock patches but negatively to tree coverage. Adjusting livestock stocking rates to control grass height emerges as a valuable tool to promote grassland specialists. The ideal scenario for bird biodiversity involves a mosaic of native grasslands, both short and tall, interspersed with tussock patches and trees. Additionally, we found that species-specific responses were influenced by bird traits, such as body size and foraging behavior, enhancing predictive capacity for pasture management. The second study focused on assessing bird responses to changes in management on six properties to increase grass height, thereby improving dry matter intake by livestock and potentially increasing livestock production. Although the average grass height increased from 6 cm to 12 cm, bird abundances and species richness did not vary. However, since the abundances of the assessed species remained constant with increasing grass height (and thus, available dry matter for livestock), we conclude that it is possible to protect a subset of grassland specialist birds while potentially increasing livestock production.

Keywords: conservation, livestock production, native grasslands, birds, ecological intensification

1. **Introducción**

Los pastizales se definen como territorios no inundables con al menos un 10 % de cobertura vegetal, dominada por formas de crecimiento de gramíneas y hierbas, y donde, en climas templados, los árboles forman un dosel de una sola capa con menos del 10 % de cobertura y una altura de 5 metros (Dixon et al., 2014). Los pastizales globalmente están presentes en todos los continentes (excepto la Antártida) y cubren entre un 31 % y 43 % de la superficie terrestre (Gibson, 2009; Dixon et al., 2014). Sin embargo, la mayor parte de los pastizales altamente productivos han sido convertidos a cultivos y praderas artificiales. Esto se debe, en parte, a que los pastizales templados son el bioma con el menor porcentaje de protección (Hoekstra et al., 2005).

Los pastizales naturales ofrecen numerosos y valiosos servicios ecosistémicos. Estos servicios ecosistémicos incluyen el sostén de biodiversidad vegetal animal y vegetal, el control de la erosión del suelo, el almacenamiento de carbono orgánico en el suelo, la regulación del ciclo de nutrientes y la provisión de agua. Además, proveen belleza escénica, cultura y medios de subsistencia para sus habitantes (Gibson, 2009; Weyland et al., 2017). Dentro de la biodiversidad animal, las aves se destacan por sus abundancias, diversidad y rol en el ecosistema (Barzan et al., 2021).

1.1. Las aves en los pastizales

Las aves de pastizales son aquellas que están adaptadas a o que pueden depender de una variedad de tipos de pastizales para todo su ciclo de vida o parte de él, como la reproducción (incluyendo alimentación y nidificación), la migración o la invernada (Vickery et al., 1999). El número de especies es variable según la región, pero, en general, son ensambles relativamente sencillos, con algunas especies dominantes (*e. g.*, Knopf, 1996). Las especies se reparten más o menos equitativamente entre paseriformes y no paseriformes (Azpiroz et al., 2012; Knopf, 1996; Vickery et al., 1999); por lo tanto, la diversidad ecológica, morfológica y comportamental es bastante amplia. Sin embargo, un aspecto bastante extendido en muchas aves de pastizal es la nidificación en el suelo. Generalmente ubican los nidos a una altura de hasta 30 cm del suelo, ocultos en pequeños cúmulos de vegetación graminoide que los protege de los depredadores y adversidades meteorológicas. A las especies que nidifican en el suelo y que dependen del hábitat de pastizal para todo su ciclo de vida se las denomina «obligatorias de pastizal», mientras que las especies que utilizan otros hábitats como bosques y humedales, además del pastizal, para parte de su ciclo de vida son consideradas «facultativas de pastizal» (Azpiroz et al., 2012; Knopf, 1996; Vickery et al., 1999). Los árboles y arbustos son un recurso muy utilizado por numerosas especies de aves de pastizal para nidificar y refugiarse (Dias et al., 2014).

A las especies de aves de los pastizales se las suele clasificar según su relación con la altura del pastizal (Fisher y Davis, 2010). Muchas especies están adaptadas a los pastos cortos, de menos de 10 cm de altura, mientras que otras requieren pastizales altos, de más de 60 cm de altura. Además, hay especies que utilizan pastos cortos para buscar su alimento y pastos altos para nidificar, descansar y refugiarse. Las preferencias de altura del pasto de las diferentes especies parecen estar relacionadas con sus adaptaciones para la alimentación y supervivencia. Muchas especies se alimentan de insectos y semillas que localizan en el suelo; la altura baja del pasto les permite moverse con facilidad y detectar visualmente sus presas o ítems alimenticios (Butler et al., 2005). Además, las especies de aves que dependen de la detección temprana de los depredadores para escapar a tiempo se favorecen de espacios abiertos sin obstrucciones visuales (Whittingham et al., 2004); en este sentido, el pasto corto puede ser ventajoso. Por otro lado, hay aves que tienen como estrategia antidepredación el camuflaje con la vegetación circundante. Estas especies necesitan pastos más altos para esconderse y pasar desapercibidas para los depredadores, así como para nidificar y encontrar su alimento (Azpiroz et al., 2012).

1.1.2. La pérdida de hábitat de aves de pastizal en el mundo

En el mundo, los pastizales naturales templados han sufrido drásticas diminuciones de su superficie debido a la conversión para agricultura y praderas artificiales (Song et al., 2018; Squires et al., 2018). Consecuentemente, la biodiversidad de este ecosistema se ha visto severamente afectada. En particular, se han constatado fuertes declinaciones de abundancias en aves de pastizal en Norteamérica y Europa (Rosenberg et al., 2019). De hecho, de todos los biomas evaluados en Norteamérica, las aves de pastizal mostraron la mayor magnitud de pérdida poblacional desde 1970: se perdieron más de 700 millones de individuos reproductores en 31 especies y 74 % de las especies de pastizales están disminuyendo (Rosenberg et al., 2019). Al igual que en otras regiones, la principal causa fue la transformación del uso del suelo desde pastizales naturales a agricultura (Douglas et al., 2023). Se estima que pérdidas muy importantes, pero aún no documentadas, han ocurrido en otras regiones de pastizales templados (*e. g.*, Azpiroz et al., 2012).

1.1.3. La ganadería en pastizales y las aves en el mundo

El pastoreo de ganado es una herramienta de gestión importante para la conservación de las aves en muchas praderas nativas en todo el mundo (Derner et al., 2009; Nugent et al., 2022). La acción del pastoreo de ganado influye sobre el hábitat de las aves de pastizal a través de la modificación de la estructura de la vegetación. Los pastoreos más tradicionales en pastizales naturales templados han generado estructuras verticales de baja altura y homogéneas, lo cual limita los recursos disponibles para muchas especies (Fuhlendorf y Engle, 2001). Por este motivo, múltiples especies de aves que requieren pastizales altos para nidificar, alimentarse o refugiarse se han visto negativamente afectadas (Barzan et al., 2021). Además, cuando la cantidad de ganado es excesiva, se generan pérdidas importantes de nidos y huevos por el pisoteo del ganado (Sharps et al., 2017). Las aves que utilizan exclusivamente pastos cortos no parecen tener conflictos con el pastoreo tradicional (*e. g.*, Aldabe et al., 2019). Sin embargo, para promover ensambles de aves diversos y favorecer especies en declinación, numerosos trabajos reportan la necesidad de promover un ordenamiento del pastoreo en los predios que promueva heterogeneidad en la altura del pasto y así compatibilizar el hábitat de un amplio espectro de especies con los requerimientos del ganado (Isacch y Cardoni, 2011; Davis et al., 2020; Fuhlendorf et al., 2017; Pírez y Aldabe, 2022).

1.2. La problemática en los pastizales del Río de la Plata

Los pastizales ubicados en el Cono Sur de América son uno de los pastizales templados más extensos del mundo. Se distinguen en dos subregiones principales: la pampa, una llanura extensa de pastizales con poca vegetación arbórea, y los Campos en Uruguay, el sur de Brasil y el sur de Paraguay, que son más escarpados y cuentan con mayor presencia de arbustos, árboles y humedales (Cartes y Yanosky 2020; Modernel et al., 2016; Oyarzabal et al., 2020; Soriano et al., 1992).

Estos pastizales están en peligro crítico y requieren atención urgente (Hoekstra et al., 2005). La conversión de extensas áreas de pastizales nativos en tierras de cultivo (que aumentó un 23 % entre 2000 y 2013; Baeza y Paruelo, 2020), la forestación (Baeza et al., 2022) y el aumento de la ganadería intensiva están afectando la composición y estructura de la vegetación (Modernel et al., 2016), lo cual pone en peligro a las aves (Da Silva et al., 2015; Gavier-Pizarro et al., 2012; Goijman et al., 2020) y otros servicios ecosistémicos (Jaurena et al., 2021).

Uno de los problemas centrales que degradan la integridad ecosistémica y el potencial productivo de los pastizales naturales es la elevada intensidad de pastoreo con la que se manejan (Jaurena et al., 2021; Soca et al., 2008), que puede generar pastizales cuya estructura es uniforme (*i. e.*, baja heterogeneidad ambiental) y de baja altura (Rosa et al., 2019). Esta estructura, que implica baja oferta de forraje, limita la productividad (Do Carmo et al., 2016) e impacta negativamente sobre la biodiversidad (Modernel et al., 2016). Además, la excesiva intensidad de pastoreo afecta la composición botánica del pastizal, a través de una sustitución de especies cespitosas e invernales por postradas y estivales, y, en consecuencia, se reduce aún más su potencial productivo (Altesor et al., 2005). Por lo tanto, la biodiversidad y la productividad se ven actualmente afectadas de manera negativa por la intensidad inadecuada del pastoreo.

1.2.2. La problemática de las aves de pastizal en sistemas pastoriles en el Río de la Plata

El pastoreo del ganado también influye en la composición y abundancia de aves en los pastizales naturales al modificar la altura y densidad estructural y composición de la vegetación (Figura 1; Azpiroz y Blake, 2016; Barzan et al., 2021; Isacch y Cardoni, 2011; Dias et al., 2017). El pastoreo reduce la altura del pasto, lo que afecta negativamente a las especies de aves que dependen de pastizales altos (Dias et al., 2014; Blumetto y Castagna, 2019). En consecuencia, los ensambles de aves de pastizal están dominados (en términos de abundancia relativa) por especies que utilizan pastos cortos (ca. <10 cm), mientras que las especies de aves que utilizan pastos de mayor altura (*e. g.*, mayores a 35 cm como *Donacospiza albifrons, Embernagra platensis, Sporophila spp, Emberizoides ypiranganus* y *Rynchotus rufescens,* entre otras) son mucho más escasas o están ausentes (Azpiroz y Blake, 2016; Cardoni et al., 2015; Dias et al., 2017). Consecuentemente, la diversidad de aves a distintas escalas espaciales está por debajo de su potencial. Sin embargo, las respuestas institucionales han sido insuficientes para contrarrestar la pérdida de diversidad de aves. La superficie de pastizales con objetivos de manejo orientados para la restauración del ensamble de aves y otra biodiversidad es francamente insuficiente (Bilenca y Miñarro, 2004). Dado que la mayor parte de los pastizales de la región está en manos de privados y son utilizados para la producción agropecuaria, la conservación y restauración de la avifauna de pastizal está en la voluntad de los productores rurales (Fontana et al., 2016), que definen su gestión del pastizal en función de intereses principalmente económicos.

En este contexto, para promover la restauración de la diversidad de aves de pastizal en los pastizales del Río de la Plata, es necesario trabajar en los sistemas de producción para mantener o, idealmente, mejorar el ingreso neto de los productores. De esta forma es más viable que los productores incorporen manejos que beneficien a las aves.

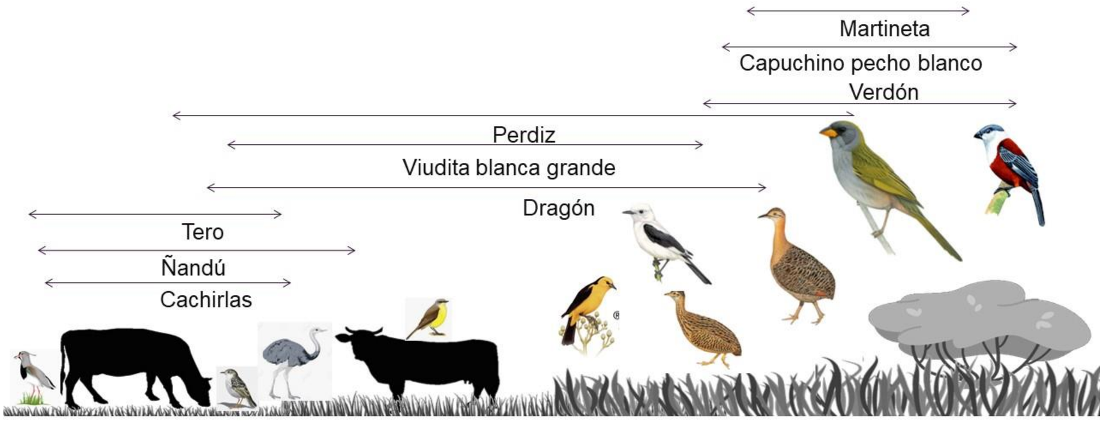

Figura 1. Las aves responden de manera particular según la especie a la altura del pasto, que está fuertemente influenciada por la presión de pastoreo de ganado (Fuente: elaboración propia).

En este sentido, resultados preliminares sugieren que esta situación en donde conservación y producción se favoreven, es posible en sistemas pastoriles familiares en el este de Uruguay, en donde cambios en la gestión espacio-temporal del pastoreo promovió heterogeneidad de altura del pasto dentro y entre potreros, e incrementó la heterogeneidad de alturas de pasto y mejoró la producción ganadera (Albicette et al., 2017; Ruggia et al., 2021), lo cual favorece la diversidad del ensamble de aves de pastizal (Dias et al., 2017).

1.2.3. Relación entre altura del pasto y aves como insumo para la gestión del pastoreo

Conocer el efecto de la altura del pasto sobre la comunidad de aves y sobre especies individuales es fundamental para el diseño de modelos de gestión del pastoreo que mantengan o aumenten la diversidad de aves, así como para la promoción de hábitat adecuado de especies de aves de particular importancia para la conservación (*e. g.*, especies amenazadas o prioritarias para su conservación).

La relación entre la altura de pasto y el ensamble de aves ha sido parcialmente estudiada en los pastizales del Río de la Plata mediante análisis de ordenación. Este abordaje metodológico presenta limitantes en la capacidad de predecir la composición de especies del ensamble, así como las respuestas de especies individuales. La capacidad para predecir las respuestas de ensambles y especies ha sido destacada como uno de los aspectos clave para avanzar en el área de la ecología aplicada a la conservación (Mouquet et al., 2015). Los hallazgos más relevantes indican que la altura del pasto, la heterogeneidad de alturas de pasto (*i. e.*, el parcheado) y la presencia de un estrato arbóreo influyen en la ocurrencia de una proporción relevante de especies de aves de pastizal en los ensambles estudiados, así como en la diversidad (Azpiroz y Blake, 2016; Cardoni et al., 2015; Dias et al., 2014). No obstante, es necesario abarcar nuevas regiones, especies, otras variables relacionadas con el manejo del pastoreo, y profundizar en el detalle de la relación altura del pasto y aves para especies individuales.

Se reconocen tres grupos de especies de aves según la altura del pasto que utilizan: especies que utilizan pastizales cortos, pastizales de altura intermedia y pastizales altos. Las especies que utilizan pastos de altura intermedia y alta tienen un rango que va desde aproximadamente 20 cm a más de 50 cm, mientras que las especies que utilizan pastizales cortos usan pastos cuya altura promedio tiene un rango de 2 a 20 cm (Agra et al., 2015; Dias et al., 2014). Cabe destacar que se desconoce el efecto de los cambios de altura de pasto dentro de cada grupo de aves (de pasto corto, intermedio y alto), con algunas excepciones que muestran que muy pocos centímetros del estrato herbáceo bajo pueden cambiar notablemente el uso por parte de una determinada especie (Aldabe et al., 2019).

La altura del estrato herbáceo inferior es la que responde más rápidamente a los cambios en la intensidad de pastoreo. Se espera que aumentos de la altura del pasto en este estrato beneficien a las aves en términos de éxito reproductivo y para ocultamiento (Cardoni et al., 2015; Zalba y Cozzani, 2004). Un estudio reportó un incremento de la densidad y riqueza de aves cuando la altura del pasto aumentó desde el rango de 0-15 cm al de 15-30 cm (Zalba y Cozzani, 2004). Sin embargo, en general, se desconoce cómo responde el ensamble a los cambios de altura del pasto en el estrato herbáceo inferior. En este sentido, uno de los objetivos de esta tesis es modelar la relación entre altura del pasto y la abundancia en un grupo de especies en pastizales del este de Uruguay en predios con ganadería.

1.3. Problemáticas productivas del campo natural

La mayoría de las explotaciones ganaderas en la región de los pastizales del Río de la Plata que basan su sistema productivo en campo natural presentan grandes brechas respecto a su potencial productivo (Modernel et al., 2018). Consecuentemente, estos sistemas perdieron competitividad respecto a otros rubros como la agricultura, lo que favoreció la conversión de grandes superficies de campo natural. Entonces, el desafío es identificar estrategias de intensificación que incrementen la rentabilidad manteniendo los campos naturales y sus servicios ecosistémicos.

Los sistemas productivos con base en campo natural están caracterizados por pastoreos continuos (todo el año) con cargas más o menos constantes. Sin embargo, la productividad primaria de estos pastizales muestra grandes variaciones estacionales, con valores mínimos en el período invernal y máximos en los meses de primavera y verano (Berreta et al., 2000). Como resultado del exceso de carga animal en los momentos de escasez forrajera se genera sobre pastoreo recurrente, baja oferta de forraje y pastos cortos y estructuralmente homogéneos, con impactos negativos sobre el desempeño animal.

1.3.1. Estrategias basadas en tecnologías de proceso para aumentar la rentabilidad y conservar el campo natural y sus servicios ecosistémicos

Para contrarrestar las debilidades productivas del campo natural (*i. e.*, marcada estacionalidad), se han propuesto varias recomendaciones de manejo. El control de la oferta de forraje y altura del pasto constituye uno de los factores determinantes de la producción vegetal y animal en los ecosistemas pastoriles (Carvalho et al., 2019; Do Carmo et al., 2018). A través del control de la oferta de forraje se busca generar suficiente masa foliar para garantizar el desempeño animal, así como asegurar un residuo de hojas verdes para que la planta capte suficiente radiación para la fotosíntesis y, de esta manera, se favorezca el crecimiento del pasto (Rinaldi, 1997). En este sentido, incrementos de la oferta de forraje (y de la altura del pasto) favorecen la producción y tasa de consumo de forraje, y la eficiencia de uso de la energía consumida por los animales, dado que se optimiza la tasa de consumo y el tiempo diario destinado a la rumia y pastoreo (Do Carmo et al., 2016).

El manejo de la carga ganadera es una de las estrategias de mayor impacto para lograr ofertas de forraje adecuadas a lo largo del año. Para ajustar la carga a la disponibilidad de forraje, es necesario monitorear la producción de forraje y el peso vivo de los animales, y realizar las modificaciones correspondientes de asignación de peso vivo a cada potrero. Varios casos de estudio demostraron que esta estrategia incrementa varios indicadores productivos (Albicette et al., 2017; Claramunt et al., 2018; Do Carmo et al., 2018). Además, el manejo de la estructura de la vegetación (*e. g.*, altura) tiene una gran influencia sobre la eficiencia de los procesos de forrajeo (Carvalho et al., 2013). Por lo tanto, además del manejo de la carga, el manejo de la altura del pastizal es una herramienta para incrementar el aprovechamiento del pasto. Finalmente, las exclusiones temporales de pastoreo permiten generar reservas que pueden ser aprovechadas en períodos de bajo crecimiento estacional (Fedrigo et al., 2018).

Por otro lado, existen herramientas de bajo costo que se basan en el manejo nutricional del ganado que pueden aumentar la rentabilidad y reducir los costos. Hacer coincidir los momentos del año en donde el ganado tiene los mayores requerimientos energéticos con la época de mayor producción de pasto resulta en incrementos en varios indicadores productivos (Do Carmo et al., 2016). Esto se logra ajustando los tiempos de entore para que los nacimientos ocurran sobre la primavera (Spitzer et al., 1995). Otras intervenciones implican priorizar vacas primíparas, destinando a estos animales los recursos alimenticios de mayor calidad para favorecer su crecimiento y función reproductiva, y el destete precoz (Quintans et al., 2009). Todas estas intervenciones tienden a mover los sistemas hacia situaciones con mayor disponibilidad y altura del pasto durante todo el año, lo que favorece la rentabilidad y reduce los riesgos por sequía. No obstante, cómo responde la avifauna a estos incrementos de altura del pasto no es conocido.

1.4. ¿Es posible gestionar la altura del pasto para aumentar la producción ganadera y la diversidad de aves?

De forma general, un incremento en la oferta de forraje puede aumentar la altura y biomasa del pastizal y favorecer la presencia de parches de pastizal alto y, consecuentemente, la presencia de especies de aves de pastizal alto (Da Trindade et al., 2016; Rosa et al., 2019). Con niveles de oferta de forraje limitado (*e. g.*, 4 % PV/día) se forma un pastizal de perfil uniforme, de baja altura, con predominio de especies postradas. Con ofertas de forraje un poco mayores (*e. g.*, 8 % PV/día) se incrementa la altura del estrato inferior y aumenta la heterogeneidad estructural de la vegetación por aumento en la proporción de matas de pastizales altos (pajonales) y especies subarbustivas, dado que estas son poco seleccionadas por el ganado bajo condiciones normales. No obstante, ninguna de estas dos ofertas de forraje potencia el desempeño animal. Con ofertas de forraje moderadas a altas (*e. g.*, 12 % a 16 %) el pastizal es más abundante, con mayor cobertura de pajonales y especies subarbustivas (Pinto et al., 2019), con incrementos claros en el desempeño animal (Da Trindade et al., 2016). Las ofertas de forraje altas determinan una típica estructura doble estrato (*i. e.*, estrato corto y estrato alto), con superficies crecientes de pastizales altos (*e. g.*, pajonales) que pueden incrementar la heterogeneidad ambiental y la diversidad de aves (Dias et al., 2017). De esta manera, incrementando la oferta de forraje se podría lograr una situación que beneficie la producción y la conservación, en la cual se pueda mejorar, a través del aumento de la altura del pasto, la diversidad de aves e intensificar la producción ganadera. Sin embargo, a la fecha no se han realizado experiencias de manejo que evalúen la respuesta de las aves a cambios de manejo tendientes a incrementar la altura del pasto.

1.5 Objetivos

a) Determinar el efecto de la altura del pastizal y otras variables de vegetación influenciadas por el manejo ganadero sobre la ocurrencia de especies individuales de aves en predios ganaderos criadores de la región este de Uruguay (artículo 1).

b) Evaluar la relación entre los cambios en manejos del pastoreo tendientes a aumentar la altura del pastizal y el ensamble de aves de pastizal en predios comerciales de la región este de Uruguay (artículo 2).

2. Bird species responses to rangeland management in relation to their traits: Rio de la Plata Grasslands as a case study

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2.1. Resumen

Las áreas utilizadas para la producción de ganado y dominadas por pastizales nativos representan una oportunidad única para conciliar la conservación de la biodiversidad y la producción ganadera. Sin embargo, el conocimiento limitado sobre las respuestas de especies individuales a la gestión de pastizales restringe nuestra capacidad para diseñar prácticas de pastoreo que favorezcan a las especies en peligro de extinción y otras aves prioritarias. En este trabajo, aplicamos el Modelo Jerárquico de Comunidades de Especies (HMSC, por sus siglas en inglés) para estudiar las respuestas de especies individuales, así como la influencia de rasgos en dichas respuestas, a variables relacionadas con la gestión de pastizales utilizando aves de los Pastizales del Río de la Plata como estudio de caso. Basándonos en datos de presencia-ausencia recopilados en 454 potreros en 46 predios, inferimos la respuesta de 69 especies considerando la detección imperfecta. Este grado de detalle llena una brecha importante en la gestión de pastizales, ya que las respuestas de especies individuales pueden utilizarse para alcanzar objetivos de conservación específicos, además de maximizar la riqueza o abundancia. Encontramos que los pastizales artificiales tuvieron un impacto generalmente negativo en muchas especies de aves, mientras que la presencia de matas de plantas cespitosas tuvieron un efecto positivo, incluyendo a todas las especies en peligro de extinción. Las aves especialistas de pastizal fueron en general sensibles a la altura del pasto y tendieron a responder positivamente a las matas de cespitosas pero negativamente a la cobertura arbórea. Controlar la altura del pasto mediante ajustes en la carga ganadera puede ser una herramienta útil para favorecer a las especialistas en pastizales. Para favorecer una amplia gama de especies de aves en los predios, es deseable un mosaico de pastizales nativos bajos y altos con matas de cespitosas y árboles. También encontramos que las respuestas específicas de las especies fueron moduladas por sus rasgos: las aves de tamaño pequeño respondieron positivamente a las plantas cespitosas y la cobertura arbórea, mientras que las especies grandes respondieron negativamente al aumento de la altura del pasto. Los forrajeadores terrestres prefirieron el pasto corto, mientras que las aves que apenas utilizan este estrato no se vieron afectadas por la altura del pasto. Los resultados sobre la influencia de los rasgos en las respuestas de las aves son una novedad importante en relación con trabajos anteriores en pastizales y potencialmente aumentan nuestra capacidad de predicción y transferibilidad del modelo en regiones de pastizales.

Palabras clave: tamaño corporal, conservación, comportamiento de forrajeo, detección imperfecta, modelos de ocupación multi-específicos, especies objetivo

2.2. Summary

Areas used for livestock production and dominated by native grasses represent a unique opportunity to reconcile biodiversity conservation and livestock production. However, limited knowledge on individual species responses to rangeland management restricts our capacity to design grazing practices that favor endangered species and other priority birds. In this work, we applied Hierarchical Modeling of Species Communities (HMSC) to study individual species responses, as well as the influence of traits on such responses, to variables related to rangeland management using birds of the Rio de la Plata Grasslands as a case study. Based on presence-absence data collected in 454 paddocks across 46 ranches we inferred the response of 69 species considering imperfect detection. This degree of detail fills a major gap in rangeland management, as species-level responses can be used to achieve targeted conservation goals other than maximizing richness or abundance. We found that artificial pastures had an overall negative impact on many bird species, whereas the presence of tussocks had a positive effect, including all threatened species. Grassland specialists were in general sensitive to grass height and tended to respond positively to tussocks but negatively to tree cover. Controlling grass height via adjustments in stocking rate can be a useful tool to favor grassland specialists. To favor a wide range of bird species in ranches, a mosaic of short and tall native grasslands with patches of tussocks and trees is desirable. We also found that species-specific responses were modulated by their traits: small-sized birds responded positively to tussocks and tree cover while large species responded negatively to increasing grass height. Ground foragers preferred short grass while birds that scarcely use this stratum were not affected by grass height. Results on the influence of traits on bird responses are an important novelty in relation to previous work in rangelands and potentially increase our predicting capacity and model transferability across grassland regions.

Key words: body size, conservation, foraging behavior, imperfect detection, multi-species occupancy models, target species

2.3. Introduction

Fifty percent of ice-free land is devoted to agroecosystems, and the current levels of intensification in food production have led to a global crisis of environmental degradation and species loss (Newbold et al. 2015, Shukla et al. 2019). Thus, to face the biodiversity crisis and buffer the global impacts that characterize the Anthropocene, it is necessary to properly manage agroecosystems (Kremen and Merenlender 2018). Among these, rangelands (areas used for livestock grazing) are extremely important as they represent ~25% of the earth's land surface delivering 10% of the global meat supply (Alkemade et al. 2013). Rangelands dominated by native grasses represent a unique opportunity to reconcile biodiversity conservation and livestock production (Barzan et al. 2021, Gurney et al. 2021) as they frequently provide habitat for significant elements of biodiversity while favoring weight gain in cattle (Odadi et al. 2011). However, inadequately managed rangelands in recent decades have led to a drastic biodiversity decrease, and reducing this impact represents a global challenge (Varijakshapanicker et al. 2019).

From a functional ecology perspective, one of the desired properties of agroecosystems is that they sustain as many species and individuals as possible, under the assumption that this will provide stability in ecosystem functioning (Altieri 1999, Schwartz et al. 2000). Accordingly, efforts by ecologists have frequently been focused on increasing overall measures of biodiversity such as abundance or species richness (e.g.,Atkinson et al. 2005, Bailey and Brown 2011, Barzan et al. 2021, Hovick et al. 2015, Prieto-Benítez, Méndez 2011 Yarnell et al. 2007). Even though aggregated information at the community level provides a general picture about management effects on biodiversity, they may overlook shifts in community composition (Dornelas et al. 2014, Neilly et al. 2016). In addition, management schemes designed to enhance richness in agroecosystems usually benefit common species, whereas rare ones are frequently overlooked (Batáry et al. 2015). This is particularly relevant from a conservation perspective as endangered species are usually rare, contributing little to abundance or biodiversity of the whole assemblage. Moreover, in certain communities, most species tend to be rare while only a few are abundant (Schwartz et al. 2000). Therefore, quantifying individual species responses, including those infrequently observed, to management practices in agroecosystems is fundamental to achieve targeted conservation goals other than maximizing richness or abundance (Neilly et al. 2016).

In this context, multi-species occupancy models are a useful statistical tool as they estimate individual species responses to the environment (Devarajan et al. 2020, Ovaskainen et al. 2017,). In these models, species responses are part of a common multivariate distribution (Guillera-Arroita 2017). As a result, species can share information to attain more precise estimates for rare ones (Ovaskainen and Soininen 2011). Hierarchical Modeling of Species Communities (HMSC) is a type of multi-species model in which the effects of species traits and phylogeny on species responses to the environment are explicitly modeled (Ovaskainen et al. 2017). This approach reduces the risk of pulling rare species responses towards community means. In addition, inferring species responses based on their traits and not their taxonomic identity, allows to generalize across similar species, and hence, predict those unobserved or with few field records (Vesk et al. 2021). Furthermore, understanding trait-environment relationship is an important step towards model transferability across regions, something critical when designing evidence-based management practices (Schuwirth et al. 2019).

Here we use an HMSC occupancy model, based on the one recently developed by Morán-López et al. (2022), to assess individual bird species responses to rangeland management in the Rio de La Plata Grasslands (RPG). The RPG ecoregion is considered critically endangered and needs urgent attention (Hoekstra et al. 2005) due to the conversion of extensive areas of native grasslands into croplands and increased stocking rates that affect vegetation composition and structure (Modernel et al. 2016). Among taxa, birds are particularly sensitive to changes in the vegetation structure driven by rangeland management (Barzan et al. 2021). Rangelands in RPG harbor bird species with contrasting environmental requirements from short grass specialists to those preferring tussocks (Vickery et al. 1999, Azpiroz et al. 2012, Dias et al. 2017).

In this sense, management that promotes vegetation heterogeneity has been applied but was not always sufficient to achieve increases on threatened or rare species abundances (Isacch and Cardoni 2011, Pírez and Aldabe 2022). Therefore, to complement the heterogeneity approach in the RPG we need to focus management actions on habitat requirement of target species.

However, the available information on how individual bird species respond to vegetation features related to management in the RPG is very scarce. This is because most previous works have used statistical ordination tools or diversity measures to describe grassland bird community composition patterns across environmental gradients (Zalba and Cozzani 2004, Azpiroz and Blake 2009, 2016, Isacch and Cardoni 2011, Dias et al. 2014, 2017, Cardoni et al. 2015, Codesido and Bilenca 2021, Pírez and Aldabe 2022). These methods lack predictive power at the species level, which is key for guiding future actions (Mouquet et al. 2015). Here, our goals are to quantify and predict species-specific, as well as community level, responses to vegetation features affected by rangeland managements and identify the main drivers of grassland bird occupancies. As landscape composition and configuration define the pool of potential species, and management at the local level will act as a filter allowing the establishment and persistence of those species whose ecological requirements are locally met (Geiger et al. 2010, Concepción et al. 2012), we considered the effect of different landscape units on species responses. Thus, our work represents a first step towards designing targeted conservation measures such as protecting threatened species or encouraging particular bird assemblages such as grassland specialists. Furthermore, we relate both detection probability and environmental responses to species traits and their phylogeny. By doing so, we illustrate the potential of HMSC to integrate ecological concepts with their application to the design of evidence-based conservation actions.

2.4. Materials and methods

2.4.1. Study area

The Rio de la Plata Grasslands, in the Southern Cone of South America, represent one of the most extensive temperate grasslands in the world (Figure 1). Pampas and Campos are the two main sub-regions that are distinguished according to physiognomic, geomorphologic, and edaphic features. The pampas can be considered in general terms as an extensive and continuous plain of grasslands with a low occurrence of tree formations, while the Campos in Uruguay and southern Brazil are steeper and with a higher occurrence of tree formations and wetlands (Soriano et al. 1992, Modernel et al. 2016). This study was carried out in the Campos ecoregion, in southeastern Uruguay, on 46 livestock ranches with grasslands. Most ranches were located within a 30 km radius of Aiguá town (lat -34.205785 long -54.766290; Figure 1), in an approximate extension of 200.000 hectares. The landscape is composed of two distinct units: areas of mountainous relief of moderate elevation and slopes (sierras), and valley areas (sierras and valley, hereafter; Figure S5). The average elevation is 135 m with a range of 57 m to 320 m; the average slope is 3.5% with a range of 0% to 23.3%. On average, ranch area was 250 ha (from 40 to 1290 ha). Ranches are divided into paddocks of 26 hectares average (from 5.7 ha to 59 ha) for cattle grazing management. Among paddocks, natural grasslands (i.e., without tillage or artificial grasses) show spatial variation in grass height due to different soil types and grazing intensities. In the study area, tussock grass (*Panicum prionitis* and *Paspalum quadrifarium*), as well as wet-mesic grasslands create small patches (from 0.1 to 0.5 hectares) of tall grasses (ca. 1 m height). Some paddocks are sown with artificial pastures to increase forage yields and on average represent 13% (range: 0 - 76%) of ranch area (artificial pastures, hereafter). In the study region, there is a well-represented woody vegetation, which is patchily distributed throughout the grassland matrix and covers between 0 and 60% of the ranches area (5% on average).

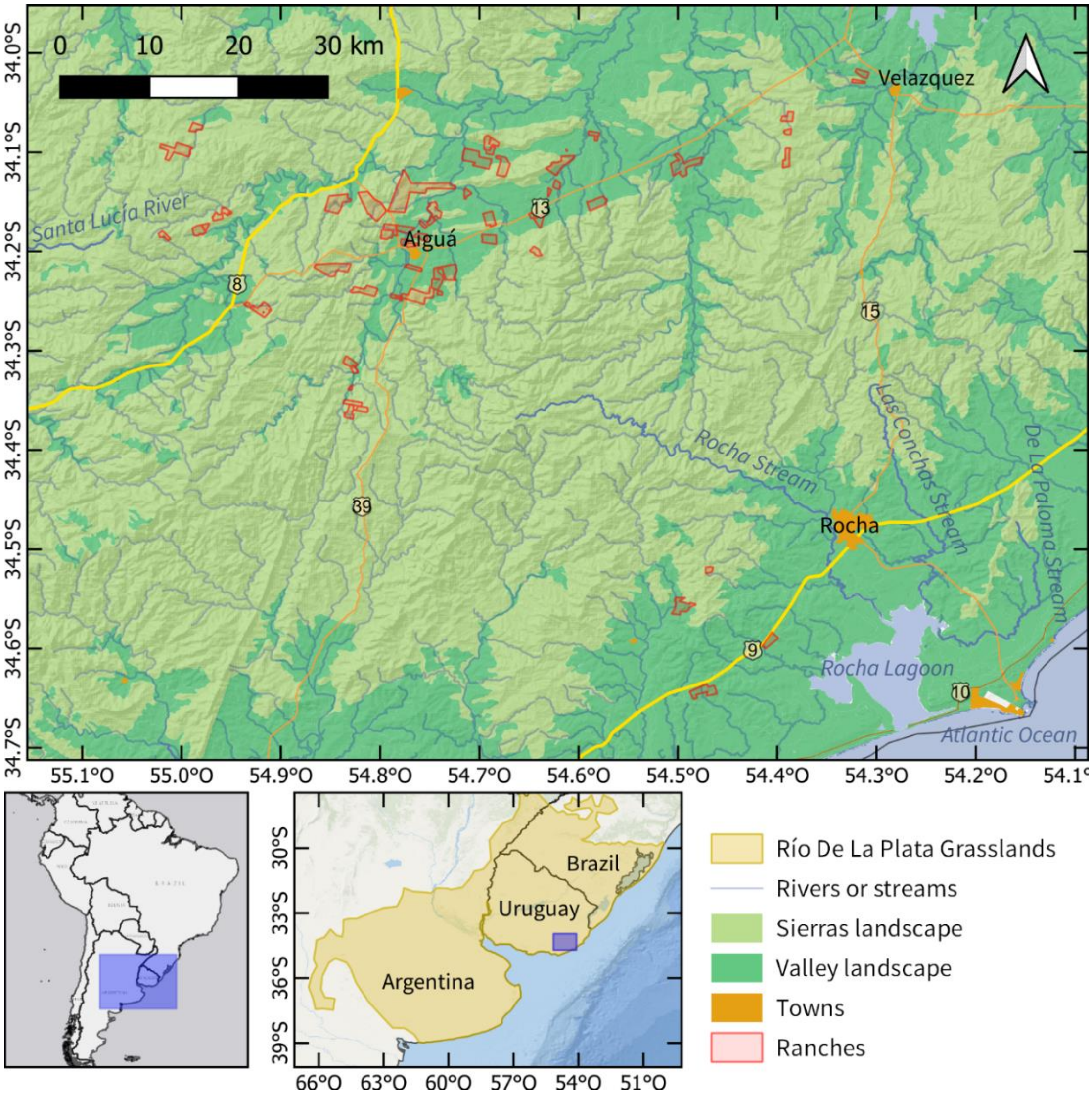


Figure 1 Study area showing the Rio de la Plata Region and the specific area where the 46 ranches were located. Ranches are highlighted in red and are distributed in valleys and highlands. Coordinate system: Decimal degree (EPSG:4326)

2.4.1.1. Study design and field methods

We selected 46 ranches located in the Sierras and valleys (22 and 25, respectively) and we sampled 6 paddocks within each ranch. To cover a gradient as wide as possible of grassland vegetation heights, we selected paddocks being grazed (or recently grazed) and others in resting periods (no grazing at the time of the field sampling). The 46 ranches were sampled in the austral spring and summer between 2015 and 2018 (4 sampling seasons in total).

Within each paddock, we randomly established one transect of 300 m length. The average spacing between transects was 679 m (range 313 to 2578). Observers walked along each transect recording and identifying each individual bird. We recorded all detected birds within ca. 150 m from each side of the transect line. We recorded only those birds that were using the grassland habitat within the transect (e.g., feeding, resting, or perching) but not those birds that just overfly the surveyed area. Bird surveys were conducted in the first 4 hours of the morning or in the last 4 hours of the afternoon. To estimate detection probability (MacKenzie et al. 2002), we repeatedly surveyed birds in six transects in each of six different ranches; we did this within a short period of time (between 2 and 4 times over a 3-day period). To homogenize heterogeneity in detection (Miller et al. 2015), time of bird surveys was randomized across ranches and paddocks. Considering the 4 years and 46 ranches, total sampling effort was 1019 transects distributed in 454 paddocks. Total number of species recorded was 69 (Appendix S1: Table S2)

2.4.1.2. Rangeland management

We assessed the effect of local rangeland management in the context of landscape effects. To do so, we considered if a ranch was in the Sierras or Valley landscapes. Then, we measured local variables related to management. In each transect from each paddock and year we measured vegetation variables that are directly related to rangeland management. In this sense, grass height is highly affected by cattle grazing intensity, while tussocks are usually cut, burned, and grazed. Trees are also cut or overgrazed, and artificial grasslands are sown. Therefore, the status of these vegetation variables reflects rangeland management. Measured variables related to rangeland management were grass height (cm), the presence of tussock and wet-mesic grasslands (tussock hereafter), percent of tree cover, and whether pastures were natural or artificial. Grass height was measured at points located every 50-m along transects, registering a total of 6 measurements that were later averaged for data modeling (see below). We used Robel's method, which consists of quantifying the visual obstruction caused by grass vegetation on a 1.8 m height pole divided into 10 cm segments (Robel et al. 1970). The observer recorded the lowest completely visible segment on the pole from a 4 m distance and 1 m eye-height. We quantified the cover of tussock within a polygon buffer of 150 m from each transect using satellite images, using Google Maxar Technologies images through Google Earth online software. Image resolution was 1000 x 700 ppi. Buffer width was 150 m at each side of the transect. The percentage of tree cover was visually assessed for each transect at the same time as bird counts. Finally, in each paddock, we visually assessed if the pastures were natural or artificial. Artificial pastures sown within the last two years are easily distinguished from natural ones, as the first is constituted by only one grass species (usually Lolium multiflorum). Geographic coordinates were recorded in the middle of each transect to model spatial autocorrelation among paddocks and ranches (see data analysis below).

2.4.1.3. Data analysis

To model species-specific responses, we used the Hierarchical Modelling of Species Communities approach (HMSC, Ovaskainen et al. 2017). In this approach, species environmental responses (coefficients) are sampled from a multivariate normal distribution with expected values that are a function of species traits and a variance-covariance matrix that is modulated by species phylogeny. Being a hierarchical model, species can borrow information from others with similar traits or that are close phylogenetic relatives. Until recently (Tobler et al. 2019), the main pitfall of earlier multi-species distribution models was that they did not account for imperfect detection in their formulation (Beissinger et al. 2016). In our model, imperfect detection is explicitly acknowledged (see Morán-López et al. 2022 for details). Furthermore, to gain precision and accuracy, species detectability is also modeled following an HMSC approach (i.e., species-specific detectability depends on their traits and phylogeny).

The basal presence/absence of bird species was a function of whether a ranch was in a valley or in a sierras area. This was then modified by the effects of local environmental covariates (i.e., variables related to rangeland management) within transects. These transect-level covariates were grass height, the presence or absence of tussocks, tree coverage, and whether the pasture is natural or artificial. We included a random effect for the sampling year, and a spatial autocorrelation term that was a function of the geographic distance between ranches. Expected parameter values varied by bird species as a function of their (log) body mass, degree of insectivory, degree of foraging on the ground stratum, and whether they are gregarious or not. Similarly, expected detection probability per species were a function of (log) body size, antipredator behavior (camouflage or early predator detection and escaping), gregarism, and how often species typically vocalize. We also estimated the variability around expected values and how much the phylogenetic correlation among species informed species responses. A detailed description of the model, including equations is provided in Appendix S1: Section 1. Behavior data was obtained from Birds of the World (Billerman et al. 2022) and the authors' personal experience (Appendix S1: Table S1). Data on body size, degree of insectivory, and degree of foraging on the ground stratum were obtained from the Elton traits database (Wilman et al. 2014). Species phylogenetic correlations were derived from a thousand phylogenetic trees obtained from Birdtree (Jetz et al. 2014). Species of outstanding body size such as *Rhea americana* were not included in the model because of number instability and model non convergence.

The model was implemented in Stan (Carpenter et al. 2017). All continuous covariates and traits were centered and standardized (mean = 0, sd = 1). For model parametrization we used weakly informative priors (details in Appendix 1: Section 1). We run 4 HMC chains for 10000 iterations after a warmup of 2000 iterations and checked for convergence (Rhat < 1.1) and effective sizes of posterior distributions. To evaluate if species responded to variables related to rangeland management and landscape type, we inspected the proportion (f) of the posterior distribution whose sign coincided with that of its mean. Here, we considered an f >= 0.95 to reflect a clear effect of a covariate on species occupancy. Lastly, to assess the adequacy of our models, we performed posterior predictive checks (Gelman and Hill 2007). To do so, for each species and year we simulated observed landscape occupancy (proportion of times a species was detected in a transect). For each iteration (4000 in total), we averaged occupancy across years. We then compared the mean and credible intervals of our predictions (0.025-0.975 quantiles) with observed values.

To illustrate how to estimate expected responses to management, we used the model to predict changes in the probability of occurrence of grassland specialist and threatened species in response to shifts in four variables related to rangeland management. For this, we obtained posterior distributions for the probability of presence of grassland specialist and threatened bird species at a transect in a ranch located in the valleys and under average environmental conditions. We then predicted new posterior distributions for the probability of presence when increasing continuous rangeland management variables (grass height, and tree cover) by doubling their corresponding mean. We also considered the case of including tussocks in the paddock and changing an artificial pasture to a natural one. We did this by choosing one covariate at a time and keeping the others at their mean or reference value.

2.5. Results

We recorded 69 bird species that were making direct use of grasslands in our sampling units (transects). A substantial proportion of these species (77%) were Passerines. 75% of the total species were uncommon and detected in less than 20% of the sample units, while the remaining 25% of the species were detected in between 20% and 59% of the transects (Appendix S1: Figure S1). We found 2 globally threatened grassland bird species: Xanthopsar flavus (detected in 4.13% of the transects), and Xolmis dominicanus (detected in 3.54% of the transects). We also found 2 near threatened species: Spartonoica maluroides (detected in only one transect), and Limnnoctites rectirostris (detected in 1% of the transects). We found 15 grassland specialists (based on Vickery et al. 1999) which in general showed low occupancy (10 species detected in less than 2% of the transects) (Appendix S1: Table S2).

Using Stan, we obtained samples for all model posteriors. All chains converged (the largest R-hat was 1.01) and we obtained sufficient bulk and tail effective samples from all posteriors (bulk mean 3525 and range from 257 to 4529, tail mean 3463 and range from 284 to 4379). Posterior predictive checks showed a good match between simulated and observed mean species occupancy, suggesting that the model adequately fitted the data (Appendix S1: Figure S1). We found a very weak effect of phylogeny on the species response to the environment and in detection probability as their mean posterior for parameters were 0.172 (90% credible interval 0.0014 to 0.45), and 0.158 (0.009 to 0.580) respectively. Estimated detection probability varied considerably among species with only 17% having values larger than 0.5, and some species had large credible intervals (Appendix S1: Figure S2). In general, detection probability was low, with an average of 0.343 and ranged from 0.064 to 0.758 (interquartile range 0.202 to 0.482, Appendix S1: Table S3). We found that detection probability was affected by vocalization behavior: species with frequent vocalization habits showed a higher probability of detection than species with lower vocalization activity. We did not find an effect of body size, antipredator behavior or gregarism on detection probability (Appendix S1: Figure S3).

Considering regression coefficients that had at least 95% of their posterior with the same sign as the posterior mean, we identified the following community level responses. There was a higher probability of occupancy in valley areas than in the sierras (22% of species had a clear negative coefficient for the sierras landscape type; Data available in Morales 2023). Regarding local management, the implantation of artificial pastures negatively affected the occurrence of 29% of bird species. In contrast, the presence of tussock had an overall positive effect in 37.7% of species. Grass height affected 37.7% of the total species (21.7% responded negatively and 15.9% positively). In addition, 36.2% of species responded to tree cover but no clear trend was found for the whole community as a similar proportion of species responded positively and negatively to tree cover (Figure 2).

Figure 2

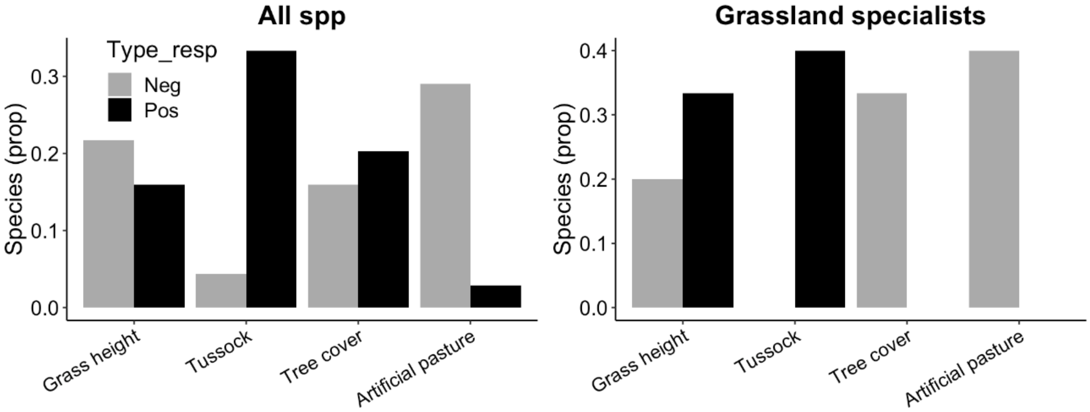


Figure 2
Summary of bird species responses to environmental covariates related to grassland management. Only those species that had a clear response are considered (that is, at least 95% of the posterior with the same sign as the posterior mean).

Our model was able to estimate the individual response of 69 species to variables related to rangeland management (Figures 3 and 4). We found considerable variation among bird species responses to variables related to rangeland management. For example, species could respond either positively or negatively to a variable while others showed a reverse response, or no clear effect was detected. Among grassland specialists, five species responded positively to increasing grass height while four preferred short grass (green triangle in Figure 3a). If we consider credible intervals of 80%, then all except two recorded grassland specialists were sensitive to grass height (Figure 3a). Also, different species in the whole species sample responded to the same variable in the same way (e.g., positive) but with different magnitude. For example, when including tussocks in a paddock, the increase in the probability of presence of

Embernagra platensis was almost twice of that of Pseudoleistes virescens (Figure 3b). Ten non-grassland specialists responded positively to increasing tree cover (Figure 4a), while all grassland specialists sensitive to tree cover responded negatively (Figure 2). Considering the whole species sample, credible intervals were larger for those species with few records (Figures 3 and 4).

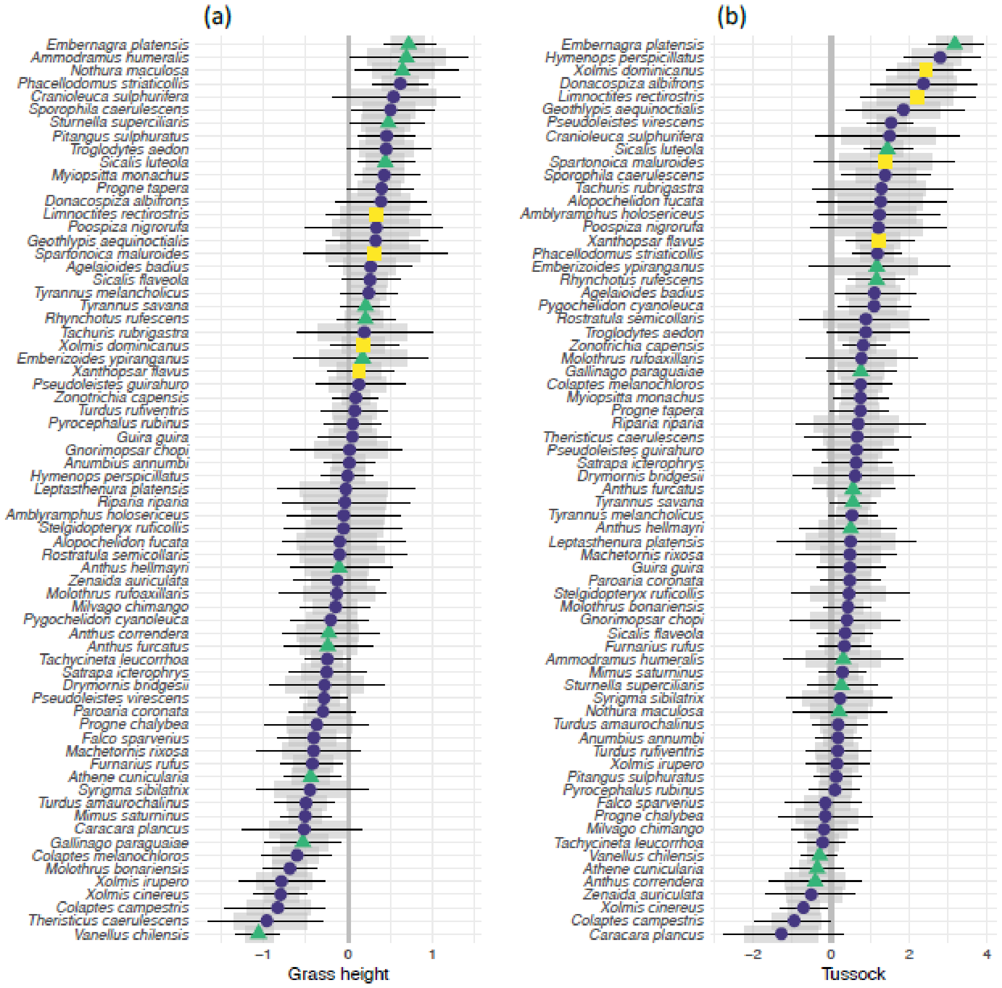


Figure 3 Species responses to grass height (a) and area of tussocks (b). Symbols show posterior means and horizontal black lines the 95% credible intervals with grey bands showing 80% credible intervals. The x axis shows the coefficient value. If this value is larger than zero, the probability of the species being present increases with increasing values of that covariate, while negative values mean a decrease in the probability of presence. Green triangles are grassland specialists while yellow squares are threatened species.

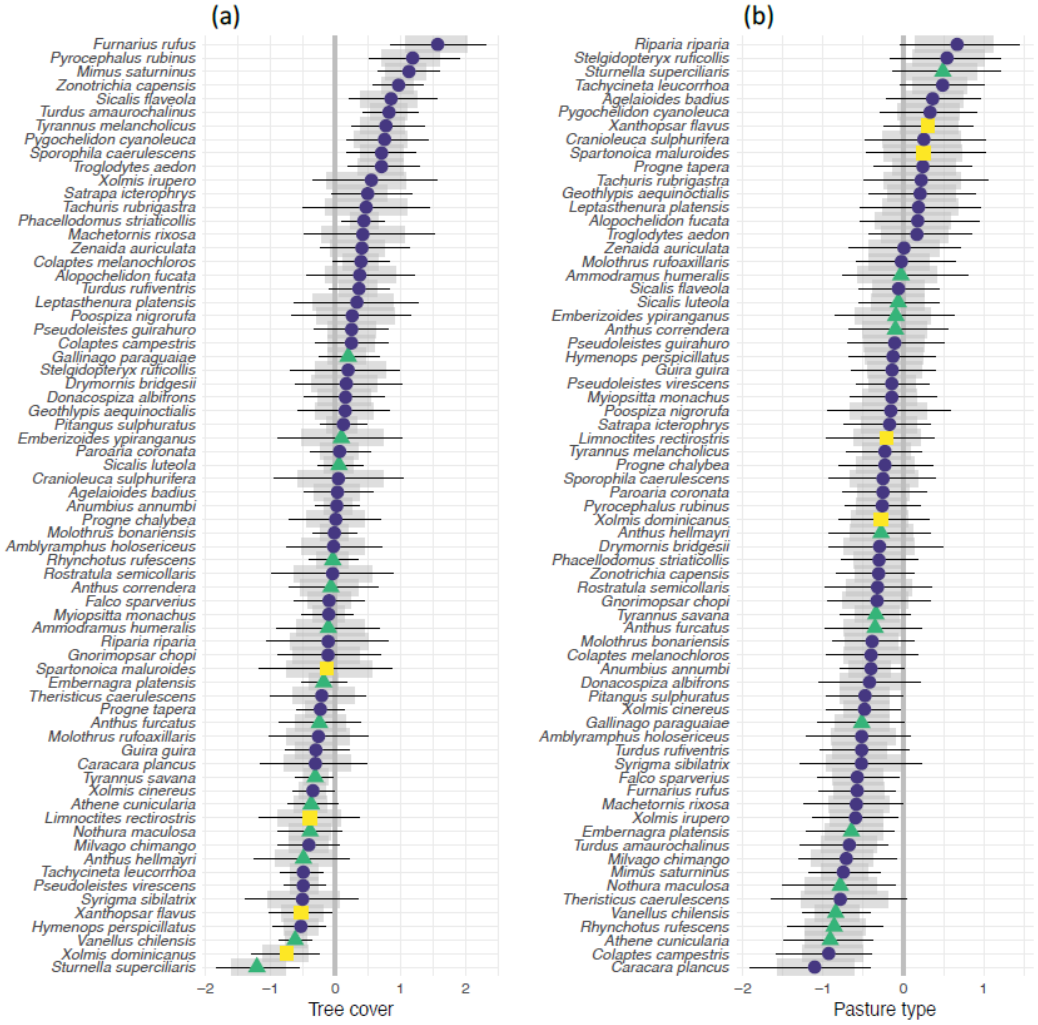


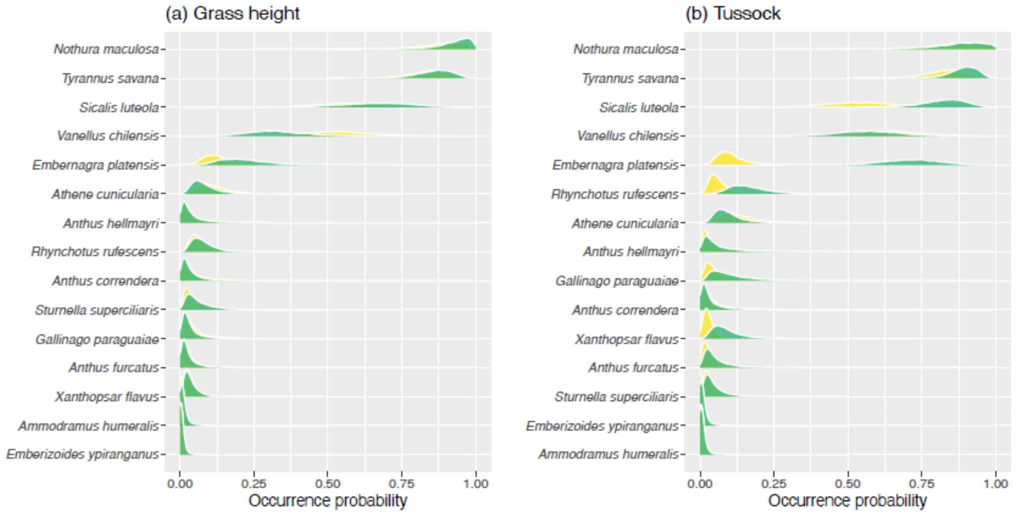
Figure 4 Species responses to tree cover (a) and pasture type (b) positive coefficients represent increased probability of presence in artificial pastures. Dots show posterior means and horizontal black lines the 95% credible intervals with grey bands showing 80% credible intervals. The x axis shows the coefficient value. If this value is larger than zero, the probability of the species being present increases with increasing values of that covariate, while negative values mean a decrease in the probability of presence. Green triangles are grassland specialists while yellow squares are threatened species.

2.5.1. Predicting individual responses to management variables

Grassland specialists *Athene cunicularia, Gallinago paraguaiae* and *Vanellus chilensis* decreased their probability of being present at a paddock when grass height doubled from its mean value of 11.76 cm (Figure 5a); whereas other grassland specialists such as *Nothura maculosa, Embernagra platensis* and *Sicalis luteola* increased their probability of presence. The last two species plus *Rynchotus rufescens* increased their probability of occurrence when tussocks were included in a paddock (Figure 5b). Four grassland specialists (*Xanthopsar flavus, Tyrannus savanna, Vanellus chilensis* and *Sturnella superciliaris*) decreased their probability of occurrence when tree cover was doubled from its mean of 4.97% (Figure 5c). Notably, when changing an artificial pasture to a natural one, 8 out of 15 grassland specialists increased their probability of occurrence (Figure 5d).

The two threatened species (*Xanthopsar flavus* and *Xolmis dominicanus*) had a higher probability of occurrence in rangelands located in the valleys compared to the Sierras (Morales 2023). Keeping all other variables at average values, the probability of occurrence of *Xanthopsar flavus* and of *Xolmis dominicanus* more than doubled when comparing a site on the sierras to one on the valleys (from 0.025 to 0.056 and from 0.0056 to 0.0155 respectively).

When tussocks are present in a paddock, the probability of occurrence of all threatened species had a relevant increase (Appendix S1: Figure S4). Keeping all other environmental variables at their mean, and in a valley, *Limnoctites rectirostris* probability of occurrence increased from 0.0072 to 0.0683, when the tussocks are included in a paddock. *Xanthopsar flavus* and *Xolmis dominicanus* responded negatively to increasing tree cover with their probability of occurrence decreasing from 0.015 to 0.013 and from 0.006 to 0.004 respectively when tree cover was doubled from its mean of 4.97%.



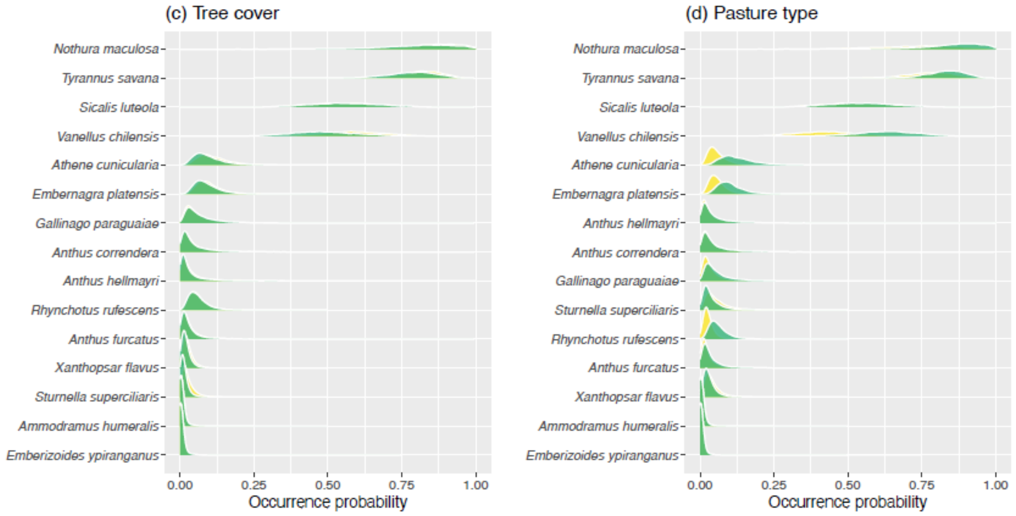


Figure 5 Grassland specialist birds predicted responses to (A) Doubling in grass height from its mean of 11.76 cm, (B) the inclusion of tussock in the paddock. (C) Doubling in tree cover from its mean of 4.97%, and (D) changing an artificial pasture to a natural one. The yellow shade are posterior probabilities of occurrence under average conditions in a valley, and the green shade the predicted posterior after changing levels of variables related to rangeland management.

2.5.2. Species trait responses to management

We found that morphological and behavioral traits influenced species responses. Larger species, as well as those that predominantly use the ground, tended to prefer the valley landscape type instead of the Sierras (Appendix S1: Table S4). As body size increased, the species tended to respond negatively to increasing grass height, tree cover, and to the presence of tussocks (Figure 6 a,b,c). Small birds tended to respond positively to tussocks and tree cover (Figure 6b and 6c). Species that predominantly use the ground tended to prefer short grass heights and low tree cover (Appendix S1: Table S4), while those scarcely using this stratum seemed to be insensitive to grassland height (Figure 6e). Artificial pastures negatively influenced bigger-sized species that do not usually form large groups (i.e., not gregarious; Figure 6d and 6f).

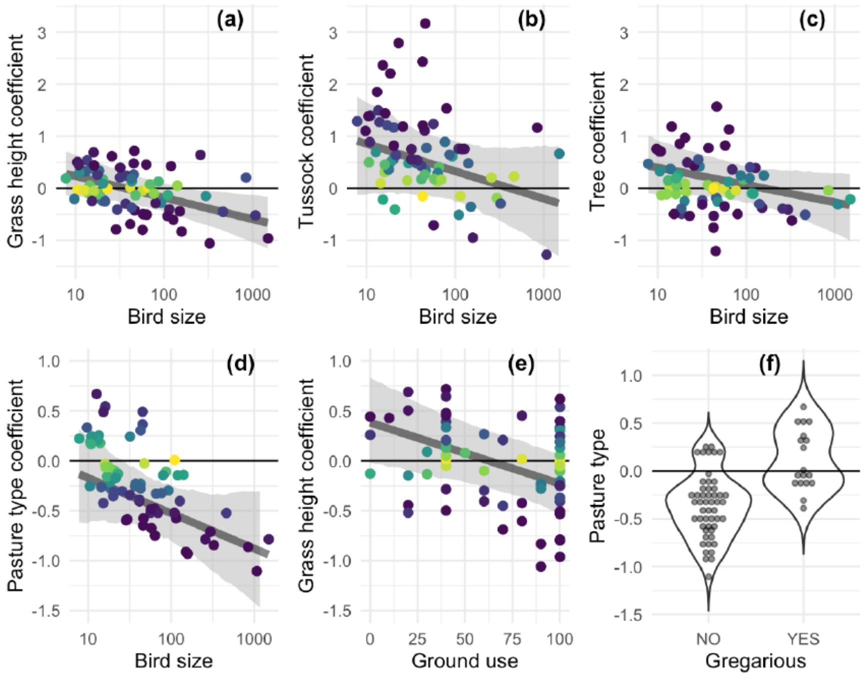


Figure 6 Relationships between bird species traits and variables related to rangeland management. Dots show the value of coefficients for different species. Only those traits that had a clear effect on species responses to the environment are shown. Darker color of dots show coefficient with clearer statistical effect as indicated with the fraction of the posterior that is of the same sign as the posterior mean. (A) Body size and response to grass height. (B) Body size and response to the presence of tussocks. (C) Body size and response to tree cover. (D) Body size and the effect of artificial pastures. (E) Ground use (percentage= 0 means low ground use while 100 means intense use of this stratum) and response to grass height. (F) Gregariousness and responses to artificial pastures. Body size is in log scale.

2.6. Discussion

Here, applying Hierarchical Modelling of Species Communities (HMSC,Ovaskainen et al. 2017) to presence-absence surveys with imperfect detection, we studied the responses of 69 bird species to rangeland management in the Rio de la Plata Grasslands. Our species-level approach fills a major gap in rangeland management (Neilly et al. 2016), as they can be used to achieve targeted conservation goals (e.g., favor threatened species) other than maximizing richness or abundance (Concepción et al. 2020). For example, we found that globally Endangered Xanthopsar flavus and the Vulnerable Xolmis dominicanus, as well as most grassland specialists, responded positively to tussocks and negatively to tree cover. Grassland specialists were sensitive to grass height. At the community level, there was a positive effect of tussocks on individual species, but a clear negative effect of artificial pastures. Furthermore, we were able to relate species traits to their response to variables related to rangeland management: small-size birds tended to respond positively to tussocks and trees while ground-foraging birds tended to prefer short grass. These findings are an important novelty with respect to previous works in rangelands and a key aspect for model transferability across grassland regions as we expect that other bird species with similar traits might respond similarly.

As species tended to respond differentially to variables related to rangeland management, it is relevant to keep those variables at different levels to favor a wide range of species. Most grassland species were sensitive to grass height; some responded positively while others responded negatively. Similarly, several broadly distributed and generalist species such as Furnarius rufus responded positively to tree cover while grassland specialist tended to respond negatively to trees. Therefore, to favor a wide range of bird species it is desirable to have a mix of short and tall grasslands with patches of tussocks and low tree cover. This is in line with other works that recommended promoting grassland heterogeneity to increase bird richness (Fuhlendorf et al. 2006, Hovick et al. 2015, Dias et al. 2017). Here, we extend these recommendations by specifically indicating key structures, such as tussocks and trees, to promote such heterogeneity. An important novelty is that we enrich previous approaches by building a model that can be used to predict the composition of the expected species assemblage as a response to the above-mentioned management.

Even though rangeland science has highlighted the broad importance of heterogeneity for birds and biodiversity in general (Fuhlendorf et al. 2017), management oriented to increase habitat heterogeneity in rangelands was not always sufficient to achieve desired bird conservation outcomes (e.g., Davis et al. 2020). This could be for example because some species require large areas of continuous habitat (Doncaster et al. 1996), or because certain features such as very tall grasses are not included within the heterogeneity of habitats (Pírez and Aldabe 2022). Therefore, in line with other works, we highlight that conservation actions oriented to specific targets might be very relevant to complement the heterogeneity approach (Neilly et al. 2016, Concepción et al. 2020, Davis et al. 2020). In this sense, the degree of detail of our results (species level) could be used to suggest a wide variety of targeted management actions. This degree of detail is important since different species may be preferentially present in certain landscape units (valley vs sierras), be more sensitive to different variables related to rangeland management (e.g., grass height vs tree cover) or show contrasting responses (negative vs positive). Therefore, depending on which species (or group of species) we want to promote, different management actions should be prioritized. For example, the grassland specialists Gallinago paraguaiae and Athene cunicularia responded negatively to grass height but the Xanthopsar flavus and Xolmis dominicanus preferred tall grass height in valley landscapes. Therefore, if conservation of the latter two species is a priority, actions should focus on increasing grass height in ranches located in valleys (e.g., through adjusting stocking rate; Da Trindade et al. 2016).

Potential target species such as globally threatened were all favored by the presence of tussocks. This is coincident with previous work that reported tussocks as important habitat for other threatened species such as Sporophila palustris (Di Giacomo et al. 2010, Vizentin-Bugoni et al. 2013) It is well-established that in the RPG passerines, including threatened species, frequently use patches of tussock to feed, nest, and refuge (Comparatore et al. 1996, Isacch and Martínez 2001, Isacch et al. 2003, Zalba and

Cozzani 2004, Cozzani and Zalba 2009). However, even though continuous grazing with moderate grazing intensities results in adequate production performances in paddocks with up to 30% of tussock cover (Da Trindade et al. 2016), many ranchers opt to cut or burn tussocks as they perceive them as low energy forage for cattle production. Consequently, this resource has been seriously impacted by cattle ranchers and crops (Herrera et al. 2009). Therefore, technical assistance to ranchers is required to inform them on how to protect tussocks while attaining adequate livestock production performances (Ruggia et al. 2021).

We found that artificial pastures negatively impacted many bird species, and in particular grassland specialists such as Gallinago paraguaiae, Rynchotus rufescens and Embernagra platensis. Our results are consistent with, and widely extend, other works in the Rio de la Plata grasslands (Azpiroz and Blake 2009, Agra et al. 2015, Fontana et al. 2016, Codesido and Bilenca 2021). This pronounced negative effect might be related to the fact that artificial pastures, when recently cultivated, are homogeneous from the point of view of the structure of the vegetation as all seeds are planted at the same time and usually pertains to the same species (Agra et al. 2015). However, Isacch et al. (2005) found that the composition of birds in artificial grasslands did not change in relation to native grasslands. This controversy may be because not all artificial grasslands are structurally the same. As the years pass since their implantation, artificial grasslands tend to incorporate plant species that provide structural heterogeneity and recover species richness and composition (da Silva and Fontana 2019). Thus, we recommend promoting vegetation heterogeneity in artificial pastures through cattle grazing management. This heterogeneity has been shown to increase production results in cow-milk production in artificial grasslands (Menegazzi et al. 2021). Even more important, we encourage ranchers to be more efficient at managing native grasslands instead of sowing artificial pastures (see Da Trindade et al. 2016, Claramunt et al. 2018, Do Carmo et al. 2018).

Last, we describe how species responses were influenced by morphological and behavioral traits. Body size was the trait that affected more species responses. On the one hand, small sized birds usually responded positively to tussocks and tree cover.

This coincides with other studies that found a higher number of small species (perching birds) in tussock grasslands compared to those dominated by short grasses (Comparatore et al. 1996, Cardoni et al. 2015, Dias et al. 2017). Possibly the biological explanation is linked, at least in part, to the use of tussocks and trees as perch. Perches are highly used by passerines and constitute a key feature of the foraging strategies of these species (e.g., flycatchers use perches to capture insects on the air). Therefore, management that burn or cut tussocks will reduce perch availability and thus perching birds. On the other hand, coincident with Howland et al. (2016) for birds in Australian temperate grasslands, large species responded negatively to increasing grass height. In contrast, Soderstrom et al. (2001), working in semi-natural dry pastures of farmland landscapes in Sweden found that larger insectivorous bird species tended to prefer intermediate levels of grass height. Regarding foraging behavior, we found that species that forage in the lower stratum of the grassland preferred short grass and low tree cover. Coincident with this finding, ground insectivores/omnivores birds tend to be more abundant in savannas ecosystems with short grasses (Soderstrom et al. 2001, Howland et al. 2016, Neilly and Schwarzkopf 2019). Birds that feed on the ground can find food more easily (e.g., arthropods and seeds) when the grass is short and does not represent visual obstruction (Butler and Gillings 2004). The positive response of ground feeding species to short grass has been observed in other species such as grassland shorebirds (Aldabe et al. 2019) and in other grassy regions such as Australia and Sweden (Soderstrom et al. 2001, Howland et al. 2016, Neilly and Schwarzkopf 2019). Our results support the hypothesis that bird foraging strata preferences is a good predictor of species susceptibility to grazing (Martin and Possingham 2005), as short grass is an indicator of high grazing intensity (Da Trindade et al. 2016). So far, this hypothesis has been mostly restricted to woodland rangelands and here we provide evidence to extend this hypothesis to grass-dominated ecosystems. Lastly, we found that larger species tended to negatively respond to artificial pastures which might be related to higher food requirements of big species that are not met in homogeneous and artificial habitats (e.g., Pompermaier et al. 2020).

Previous works conducted in Europe and Australia show that ground foragers, species nesting above the ground level (i.e., elevated nests), and birds that rely on early detection of predators were more likely to use short grass (Soderstrom 2001, Martin and Possingham 2005, Whittingham et al. 2006, Howland et al. 2016). In contrast, small aerial insectivores, and ground nesting species (that rely on concealment to avoid predators) were more likely to use grasslands with taller grass in Montana, North America (Fondell and Ball 2004). Grassland specialists in our study showed responses to various variables related to management variables, with grass height being particularly influential. This group of birds displays a diverse range of traits, and their responses are likely attributable to different mechanisms. For instance, species such as Athene cunicularia, Gallinago paraguaiae, and Vanellus chilensis, which rely almost exclusively on ground insects as their food source, necessitate short grass that facilitates prey detection. Moreover, their defense mechanism against predators depends on early detection which is also benefited from shorter grass. Conversely, species such as Nothura maculosa, Rynchotus rufescens, Embernagra platensis, and Sicalis luteola, which responded positively to increased grass height, primarily feed on seeds. Tall grasses foster a greater seed abundance (Sagario et al 2020), which may explain their response. Additionally, their antipredator strategy revolves around remaining inconspicuous and taller grass provides these species with better concealment. Furthermore, due to their habit of nesting on or near the ground, they tend to select taller grasses to hide their nests from potential threats. Lastly, numerous grassland specialists were unlikely to be found in areas characterized by dense tree cover. Grasslands containing scattered tree patches might fragment the grassland habitat, thereby exerting a negative impact on species reliant on ground-dwelling insects as their food source (Marcolin et al 2021), exemplified in our study by grassland specialists Xanthopsar flavus, Tyrannus savanna, Vanellus chilensis, and Sturnella superciliaris.

Our work advances the knowledge of the ecology and management of grassland birds in the RPG, as we inferred the individual response of 30% of the species listed as grassland birds of the SE of South America (according to Azpiroz et al. 2012). We highlight that even though 75% of the recorded species were uncommon, our HMSC model was able to infer individual responses of all recorded species. This is very relevant to conservation action, as many threatened species are rare and traditional statistical methods are less powerful to infer individual responses to management. Therefore, most of the studies restrict their analysis to the most common species (e.g., Neilly and Schwarzkopf 2019). Furthermore, considering how morphological and behavioral traits influenced species response we were able to 1) guide actions targeted towards groups of species with certain traits (e.g., small-sized passerines benefit from the presence of tussocks), 2) contribute to our understanding of how species traits interact with the environment (Brown et al. 2014), and 3) provide new insights into the ecological mechanisms driving the consequences of different management practices. With a better understanding of such ecological mechanisms, we can improve our predictive capabilities as well as our capacity to design actions to achieve targeted conservation goals.

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2.7. References

Agra, M., D. Bilenca, and M. Codesido. 2015. Responses of birds to planting of Lotus tenuis pasture in the Flooding Pampas, Argentina. Emu 115.

Aldabe, J., R. B. Lanctot, D. Blanco, P. Rocca, and P. Inchausti. 2019. Managing Grasslands to Maximize Migratory Shorebird Use and Livestock Production. Rangeland Ecology and Management 72.

Alkemade, R., R. S. Reid, M. Van Den Berg, J. De Leeuw, and M. Jeuken. 2013. Assessing the impacts of livestock production on biodiversity in rangeland ecosystems. Proceedings of the National Academy of Sciences of the United States of America 110:20900–20905.

Altieri, M. A. 1999. The ecological role of biodiversity in agroecosystems. Agriculture, Ecosystems and Environment 74:19–31.

Atkinson, P. W., R. J. Fuller, J. A. Vickery, G. J. Conway, J. R. B. Tallowin, R. E. N. Smith, K. A. Haysom, T. C. Ings, E. J. Asteraki, and V. K. Brown. 2005. Influence of agricultural management, sward structure and food resources on grassland field use by birds in lowland England. Journal of Applied Ecology 42:932–942.

Azpiroz, A. B., and J. G. Blake. 2009. Avian Assemblages in Altered and Natural Grasslands in the Northern Campos of Uruguay. The Condor 111:21–35.

Azpiroz, A. B., and J. G. Blake. 2016. Associations of grassland birds with vegetation structure in the Northern Campos of Uruguay. The Condor 118:12–23.

Azpiroz, A. B., J. P. Isacch, R. A. Dias, A. S. Di Giacomo, C. S. Fontana, and C. M. Palarea. 2012. Ecology and conservation of grassland birds in southeastern South America: A review. Journal of Field Ornithology 83:217–246.

Bailey, D. W., and J. R. Brown. 2011. Rotational grazing systems and livestock grazing behavior in shrub-dominated semi-arid and arid rangelands. Rangeland Ecology and Management 64:1–9.

Barzan, F. R., L. M. Bellis, and S. Dardanelli. 2021. Livestock grazing constrains bird abundance and species richness: A global meta-analysis. Basic and Applied Ecology 56:289–298.

Batáry, P., L. V Dicks, D. Kleijn, and W. J. Sutherland. 2015. The role of agri-environment schemes in conservation and environmental management. Conservation Biology 29:1006–1016.

Beissinger, S. R., K. J. Iknayan, G. Guillera-Arroita, E. F. Zipkin, R. M. Dorazio, J. A. Royle, and M. Kéry. 2016. Incorporating Imperfect Detection into Joint Models of Communities: A response to Warton et al. Trends in Ecology and Evolution 31:736–737.

Billerman, S. M., B. K. Keeney, P. G. Rodewald, and T. S. Schulenberg. 2022. Birds of the World. Page (S. Billerman, B. Keeney, P. Rodewald, and T. Schulenberg, Eds.). Cornell Laboratory of Ornithology, Ithaca NY.

Butler, S.J., Gillings, S. 2004. Quantifying the effects of habitat structure on prey detectability and accessibility to farmaland birds. Ibis 146:123–130.

Cardoni, D. A., J. P. Isacch, and O. Iribarne. 2015. Avian responses to varying intensity of cattle production in Spartina densiflora saltmarshes of South-Eastern South America. meu 115.

Do Carmo, M., G. Cardozo, M. Jaurena, and P. Soca. 2018. Demonstrating control of forage allowance for beef cattle grazing Campos grassland in Uruguay to improve system productivity. Tropical Grasslands 7:35–47.

Carpenter, B., A. Gelman, M. D. Hoffman, D. Lee, B. Goodrich, M. Betancourt, M. A. Brubaker, J. Guo, P. Li, and A. Riddell. 2017. Stan: A probabilistic programming language. Journal of Statistical Software 76.

Claramunt, M., A. Fernández-Foren, and P. Soca. 2018. Effect of herbage allowance on productive and reproductive responses of primiparous beef cows grazing on Campos grassland. Animal Production Science 58:1615–1624.

Codesido, M., and D. Bilenca. 2021. Avian assemblages associated with different grasslands managements in cattle production systems in the pampas of Argentina. Perspectives in Ecology and Conservation.

Comparatore, V. M., M. M. Martinez, A. I. Vassallo, M. Barg, and J. P. Isacch. 1996. Abundancia y Relaciones con el Hábitat de Aves Y Mamíferos En Pastizales de Paspalum quadrifarium (paja colorada) manejados con fuego (Provincia de Buenos Aires, Argentina). Interciencia 21:228–237.

Concepción, E. D., I. Aneva, M. Jay, S. Lukanov, K. Marsden, G. Moreno, R. Oppermann, A. Pardo, S. Piskol, V. Rolo, A. Schraml, and M. Díaz. 2020. Optimizing biodiversity gain of European agriculture through regional targeting and adaptive management of conservation tools. Biological Conservation 241:108384.

Concepción, E. D., M. Díaz, D. Kleijn, A. Báldi, P. Batáry, Y. Clough, D. Gabriel, F. Herzog, A. Holzschuh, E. Knop, E. J. P. Marshall, T. Tscharntke, and J. Verhulst. 2012. Interactive effects of landscape context constrain the effectiveness of local agri-environmental management. Journal of Applied Ecology 49:695–705.

Cozzani, N., and S. Zalba. 2009. Estructura de la vegetación y selección de hábitats reproductivos en aves del pastizal pampeano. Ecologia Austral 19:35–44.

Davis, K. P., D. J. Augustine, A. P. Monroe, J. D. Derner, and C. L. Aldridge. 2020. Adaptive rangeland management benefits grassland birds utilizing opposing vegetation structure in the shortgrass steppe. Ecological Applications 30:1–14.

Devarajan, K., T. L. Morelli, and S. Tenan. 2020. Multi-species occupancy models: review, roadmap, and recommendations. Ecography 43:1612–1624.

Dias, R. A., V. A. G. Bastazini, and A. T. Gianuca. 2014. Bird-habitat associations in coastal rangelands of southern Brazil. Iheringia. Série Zoologia 104:200–208.

Dias, R. A., A. T. Gianuca, J. Vizentin-Bugoni, M. S. S. Gonçalves, G. A. Bencke, and V. A. G. Bastazini. 2017. Livestock disturbance in Brazilian grasslands influences avian species diversity via turnover. Biodiversity and Conservation 26:2473–2490.

Doncaster, C. P., T. Micol, and S. P. Jensen. 1996. Determining Minimum Habitat Requirements in Theory and Practice. Page Source: Oikos.

Dornelas, M., N. J. Gotelli, B. McGill, H. Shimadzu, F. Moyes, C. Sievers, and A. E. Magurran. 2014. Assemblage time series reveal biodiversity change but not systematic loss. Science 344:296–299.

Fondell, T. F., and I. J. Ball. 2004. Density and success of bird nests relative to grazing on western Montana grasslands. Biological Conservation 117:203–213.

Fontana, C. S., G. Dotta, C. K. Marques, M. Repenning, C. E. Agne, and R. J. dos Santos. 2016. Conservation of grassland birds in South Brazil: a land management perspective. Natureza e Conservacao 14:83–87.

Fuhlendorf, S. D., R. W. S. Fynn, D. A. McGranahan, and D. Twidwell. 2017. Heterogeneity as the Basis for Rangeland Management. Pages 169–196 in D. D. Briske, editor. Rangeland Systems. Springer International Publishing, Cham.

Fuhlendorf, S. D., W. C. Harrell, D. M. Engle, R. G. Hamilton, C. A. Davis, and D. M. Leslie Jr. 2006. Should heterogeneity be the basis fo conservation? Grassland bird response to fire and grazing. Ecological Applications 16:1706–1716.

Geiger, F., G. R. de Snoo, F. Berendse, I. Guerrero, M. B. Morales, J. J. Oñate, S. Eggers, T. Pärt, R. Bommarco, J. Bengtsson, L. W. Clement, W. W. Weisser, A. Olszewski, P. Ceryngier, V. Hawro, P. Inchausti, C. Fischer, A. Flohre, C. Thies, and T. Tscharntke. 2010. Landscape composition influences farm management effects on farmland birds in winter: A pan-European approach. Agriculture, Ecosystems and Environment 139:571–577.

Gelman, A., and J. Hill. 2007. Data Analysis Using Regression and Multilevel/Hierarchical Models. Cambridge University Press, Cambridge.

Di Giacomo, A. S., P. D. Vickery, H. Casaas, O. A. Spitznagel, C. Ostrosky, S. Krapovickas, and A. J. Bosso. 2010. Landscape associations of globally threatened grassland birds in the aguapey river important bird are ommunitietes, Argentina. Bird Conservation International 20:62–73.

Guillera-Arroita, G. 2017. Modelling of species distributions, range dynamics and communities under imperfect detection: advances, challenges and opportunities. Ecography 40:281–295.

Gurney, G. G., E. S. Darling, G. N. Ahmadia, V. N. Agostini, N. C. Ban, J. Blythe, J. Claudet, G. Epstein, A. Himes-cornell, H. D. Jonas, D. Armitage, S. J. Campbell, C. Cox, W. R. Friedman, D. Gill, P. Lestari, S. Mangubhai, E. Mcleod, N. A. Muthiga, R. Ranaivoson, A. Wenger, I. Yulianto, and S. D. Jupiter. 2021. Biodiversity needs every tool in the box : use OECMs. Nature 595:646–649.

Hoekstra, J. M., T. M. Boucher, T. H. Ricketts, and C. Roberts. 2005. Confronting a biome crisis: Global disparities of habitat loss and protection. Ecology Letters 8:23–29.

Herrera, L. P., P. Laterra, N. O. Maceira, K. D. Zelaya, and G. A. Martínez. 2009. Fragmentation status of tall-tussock grassland relicts in the Flooding Pampa, Argentina. Rangeland Ecology and Management 62:73–82.

Hovick, T. J., R. D. Elmore, S. D. Fuhlendorf, D. M. Engle, and R. G. Hamilton. 2015. Spatial heterogeneity increases diversity and stability in grassland bird communities. Ecological Applications 25:662–672.

Howland, B. W. A., D. Stojanovic, I. J. Gordon, J. Radford, A. D. Manning, and D. B. Lindenmayer. 2016. Birds of a feather flock together: Using trait-groups to understand the effect of macropod grazing on birds in grassy habitats. Biological Conservation 194:89–99.

Isacch, A. J. P., M. S. Bo, N. O. Maceira, M. R. Demaría, S. Peluc, J. P. Isacch, M. S. Bo, N. O. Maceira, and M. R. Demarı. 2003. Composition and seasonal changes of the bird community in the west pampa grasslands of Argentina. Journal of Field Ornithology 74:59–65.

Isacch, J. P., and D. A. Cardoni. 2011. Different Grazing Strategies Are Necessary to Conserve Endangered Grassland Birds in Short and Tall Salty Grasslands of the Flooding Pampas. The Condor 113:724–734.

Isacch, J. P., N. O. Maceira, M. S. Bo, M. R. Demaría, and S. Peluc. 2005. Bird-habitat relationship in semi-arid natural grasslands and exotic pastures in the west pampas of Argentina. Journal of Arid Environments 62:267–283.

Isacch, J. P., and M. M. Martínez. 2001. Estacionalidad Y Relaciones Con La Estructura Del Habitat De La Comunidad De Aves De Pastizales De Con Fuego En La Provincia De Buenos Aires. Ornitologia Neotropical 12:345–354.

Jetz, W., G. H. Thomas, J. B. Joy, D. W. Redding, K. Hartmann, and A. O. Mooers. 2014. Global Distribution and Conservation of Evolutionary Distinctness in Birds. Current Biology 24:919–930.

Kremen, C., and A. M. Merenlender. 2018. Landscapes that work for biodiversity and people. Science 362.

MacKenzie, D. I., J. D. Nichols, G. B. Lachman, S. Droege, A. A. Royle, and C. A. Langtimm. 2002. Estimating site occupancy rates when detection probabilities are less than one. Ecology 83:2248–2255.

Marcolin, F., T. Lakatos, R. Gallé, and P. Batáry. 2021. Fragment connectivity shapes bird communities through functional trait filtering in two types of grasslands. Global Ecology and Conservation 28.

Martin, T. G., and H. P. Possingham. 2005. Predicting the impact of livestock grazing on birds using foraging height data. Journal of Applied Ecology 42:400–408.

McElreath, R. 2020. Statistical Rethinking; A Bayesian Course with Examples in R and Stan; Second Edition. Second. CRC Press, Boca Ratón.

Menegazzi, G., P. Y. Giles, M. Oborsky, O. Fast, D. A. Mattiauda, T. C. M. Genro, and P. Chilibroste. 2021. Effect of Post-grazing Sward Height on Ingestive Behavior, Dry Matter Intake, and Milk Production of Holstein Dairy Cows. Frontiers in Animal Science 2.

Miller, D. A. W., L. L. Bailey, E. H. C. Grant, B. T. Mcclintock, L. A. Weir, and T. R. Simons. 2015. Performance of species occurrence estimators when basic assumptions are not met: A test using field data where true occupancy status is known. Methods in Ecology and Evolution 6:557–565.

Modernel, P., W. A. H. Rossing, M. Corbeels, S. Dogliotti, V. Picasso, and P. Tittonell. 2016. Land use change and ecosystem service provision in Pampas and Campos grasslands of southern South America. Environmental Research Letters 11.

Morales, J.M. 2023. Data and code from Bird Species Responses to Rangeland Management in Relation to Their Traits: Río de la Plata Grasslands as a Case Study. Ecological Applications 2023 (grassbirdsuruguay). Zenodo. https://doi.org/10.5281/zenodo.8099794Morán-López, T., S. Ruiz-Suarez, J. Aldabe, and J. M. Morales. 2022. Improving inferences and predictions of species environmental responses with occupancy data. Methods in Ecology and Evolution.

Mouquet, N., Y. Lagadeuc, V. Devictor, L. Doyen, A. Duputié, D. Eveillard, D. Faure, E. Garnier, O. Gimenez, P. Huneman, F. Jabot, P. Jarne, D. Joly, R. Julliard, S. Kéfi, G. J. Kergoat, S. Lavorel, L. Le Gall, L. Meslin, S. Morand, X. Morin, H. Morlon, G. Pinay, R. Pradel, F. M. Schurr, W. Thuiller, and M.

Loreau. 2015, October 1. Predictive ecology in a changing world. Blackwell Publishing Ltd.

Neilly, H., and L. Schwarzkopf. 2019. The impact of cattle grazing regimes on tropical savanna bird assemblages. Austral Ecology 44:187–198.

Neilly, H., J. Vanderwal, and L. Schwarzkopf. 2016. Balancing Biodiversity and Food Production: A Better Understanding of Wildlife Response to Grazing Will Inform Off-Reserve Conservation on Rangelands. Rangeland Ecology and Management 69:430–436.

Newbold, T., L. N. Hudson, S. L. L. Hill, S. Contu, I. Lysenko, R. A. Senior, L. Börger, D. J. Bennett, A. Choimes, B. Collen, J. Day, A. De Palma, S. Díaz, S. Echeverria-Londoño, M. J. Edgar, A. Feldman, M. Garon, M. L. K. Harrison, T. Alhusseini, D. J. Ingram, Y. Itescu, J. Kattge, V. Kemp, L. Kirkpatrick, M. Kleyer, D. L. P. Correia, C. D. Martin, S. Meiri, M. Novosolov, Y. Pan, H. R. P. Phillips, D. W. Purves, A. Robinson, J. Simpson, S. L. Tuck, E. Weiher, H. J. White, R. M. Ewers, G. M. MacE, J. P. W. Scharlemann, and A. Purvis. 2015. Global effects of land use on local terrestrial biodiversity. Nature 520:45–50.

Odadi, W. O., M. M. Karachi, S. A. Abdulrazak, and T. P. Young. 2011. African wild ungulates compete with or facilitate cattle depending on season.

Ovaskainen, O., and J. Soininen. 2011. Making more out of sparse data: hierarchical modeling of species communities. Page Ecology.

Ovaskainen, O., G. Tikhonov, A. Norberg, F. G. Blanchet, L. Duan, D. Dunson, T. Roslin, and N. Abrego. 2017. How to make more out of community data ? A conceptual framework and its implementation as models and software. Ecology Letters:561–576.

Pírez, F., and J. Aldabe. 2022. Comparison of the bird community in livestock farms with continuous and rotational grazing in eastern Uruguay. Ornithology Research.

Pompermaier, V. T., T. B. Kisaka, J. F. Ribeiro, and G. B. Nardoto. 2020. Impact of exotic pastures on epigeic arthropod diversity and contribution of native and exotic plant sources to their diet in the central Brazilian savanna. Pedobiologia 78.

Prieto-Benítez, S., and M. Méndez. 2011, February. Effects of land management on the abundance and richness of spiders (Araneae): A meta-analysis.

Robel, R. J., J. N. Briggs, A. D. Dayton, and L. C. Hulbert. 1970. Relationships Between Visual Obstruction Measurements and Weight of Grassland Vegetation1. Journal of Rangeland Management 23:295–297.

Ruggia, A., S. Dogliotti, V. Aguerre, M. M. Albicette, A. Albin, O. Blumetto, G. Cardozo, C. Leoni, G. Quintans, S. Scarlato, P. Tittonell, and W. A. H. Rossing. 2021. The application of ecologically intensive principles to the systemic redesign of livestock farms on native grasslands: A case of co-innovation in Rocha, Uruguay. Agricultural Systems 191:103148.

Sagario, M. C., V. R. Cueto, A. Zarco, R. Pol, and L. Marone. 2020. Predicting how seed-eating passerines respond to cattle grazing in a semi-arid grassland using seed preferences and diet. Agriculture, Ecosystems and Environment 289:106736.

Schuwirth, N., F. Borgwardt, S. Domisch, M. Friedrichs, M. Kattwinkel, D. Kneis, M. Kuemmerlen, S. D. Langhans, J. Martínez-López, and P. Vermeiren. 2019, November 1. How to make ecological models useful for environmental management. Elsevier B.V.

Schwartz, M. W., C. A. Brigham, J. D. Hoeksema, K. G. Lyons, M. H. Mills, and P. J. Van Mantgem. 2000. Linking Biodiversity to Ecosystem Function: Implications for Conservation Ecology.

Shukla, P. R., J. Skea, E. Calvo Buendia, V. Masson-Delmotte, H. O. Pörtner, D. C. Roberts, P. Zhai, R. Slade, S. Connors, R. van Diemen, M. Ferrat, E. Haughey, S. Luz, S. Neogi, M. Pathak, J. Petzold, J. Portugal Pereira, P. Vyas, E.

Huntley, K. Kissick, M. Belkacemi, and J. Malley. 2019. Climate Change and Land. An IPCC Special Report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems.

da Silva, T. W., and C. S. Fontana. 2019. Success of active restoration in grasslands: a case study of birds in southern Brazil. Restoration Ecology.

Soderstrom, B., T. Part, and E. Linnarsson. 2001. Grazing Effects on Between-Year Variation of Farmland Bi ommunitiestes. Ecological Applications 11:1141–1150.

Soriano, A., R. León, O. Sala, R. Lavado, V. Deregibus, M. Cauhepé, O. Scaglia, C. Velázquez, and J. Lemcoff. 1992. Rio de la Plata Grasslands. Pages 367–407 in R. Coupland, editor. Ecosystems of the world 8A. Natural grasslands: Introduction and Western Hemisphere. Elsevier, Amsterdam.

Tobler, M. W., M. Kéry, F. K. C. Hui, G. Guillera-Arroita, P. Knaus, and T. Sattler. 2019. Joint species distribution models with species correlations and imperfect detection. Ecology 100:1–14.

Da Trindade, J. K. D., F. P. Neves, C. E. Pinto, C. Bremm, J. C. Mezzalira, L. B. Nadin, T. C. M. Genro, H. L. Gonda, and P. C. F. Carvalho. 2016. Daily Forage Intake by Cattle on Natural Grassland: Response to Forage Allowance and Sward Structure. Rangeland Ecology and Management 69:59–67.

Varijakshapanicker, P., S. McKune, L. Miller, S. Hendrickx, M. Balehegn, G. E. Dahl, and A. T. Adesogan. 2019. Sustainable livestock systems to improve human health, nutrition, and economic status. Animal Frontiers 9:40–49.

Vesk, P. A., W. K. Morris, W. C. Neal, K. Mokany, and L. J. Pollock. 2021. Transferability of trait-based species distribution models. Ecography 44:134–147.

Vickery, P. D., P. L. Tubaro, J. M. C. Da Silva, B. G. Peterjohn, J. R. Herkert, and R. B. Cavalcanti. 1999. Conservation of grassland birds in the western hemisphere. Studies in Avian Biology:2–26.

Vizentin-Bugoni, J., J. I. Areta, A. G. Di Giacomo, A. S. Di Giacomo, F. Jacobs, M. A. Afonso Coimbra, and R. A. Dias. 2013. Breeding biology and conservation of the Marsh Seedeater Sporophila palustris. Bird Conservation International 23:147–158.

Whittingham, M. J., C. L. Devereux, A. D. Evans, and R. B. Bradbury. 2006. Altering perceived predation risk and food availability: Management prescriptions to benefit farmland birds on stubble fields. Journal of Applied Ecology 43:640–650.

Wilman, H., J. Belmaker, J. Simpson, C. de la Rosa, M. M. Rivadeneira, and W. Jetz. 2014. EltonTraits 1.0: Species-level foraging attributes of the world’s birds and mammals. Ecology 95:2027–2027.

Yarnell, R. W., D. M. Scott, C. T. Chimimba, and D. J. Metcalfe. 2007. Untangling the roles of fire, grazing and rainfall on small mammal communities in grassland ecosystems. Oecologia 154:387–402.

Zalba, S. M., and N. C. Cozzani. 2004. The impact of feral horses on grassland bird communities in Argentina. Animal Conservation 7:35–44.

2.8. Supplementary material

Section S1. Model details

Species-specific occupancy was modeled as follows:

$$y_{i,f,j,t} \sim \text{Binomial}\left(\phi_{i,f,j,t},\, M_{i,t}\right)$$

$$\phi_{i,f,j,t} = z_{i,f,j,t}\, p_j$$

$$z_{i,f,j,t} \sim \text{Bernoulli}\left(\psi_{i,f,j,t}\right)$$

$$\text{logit}\left(\psi_{i,f,j,t}\right) = \beta_{0,j,f} + \sum_{k=1}^{K} \beta_{j,k}\, X_{i,k} + \tau_t + \epsilon_f$$

$$\tau_{\mathrm{t}} \sim N(0,\, \sigma_t)$$

$$\beta_{0,j,f} \sim N\left(\delta_{0,j} + \delta_{1,j} L_f,\, \sigma_f\right)$$

$$\epsilon_f = \text{MVN}(\mathbf{0}, \Sigma)$$

$$\Sigma_{\mathrm{r,c}} = \eta^2 \exp\left(-\varphi^2 D_{r,c}^2\right) + d_{r,c}\sigma_s^2$$

In the *t*-th year, the number of times the *j*-th species in the *i*-th transect from the *f*-th ranch was detected ($y_{i,f,j,t}$), was assumed to follow a Binomial distribution with probability of being observed ($\phi_{i,f,j,t}$) and the number of visits to that transect in that year ($M_{i,t}$). The probability that the *j*-th species was observed in the *i*-th transect at year *t* and ranch *f* ($\phi_{i,f,j,t}$) depended on the presence/absence of the species ($z_{i,f,j,t}$) and its detectability (p_j). The presence/absence of the species was sampled from a Bernoulli distribution with occupancy probability $\left(\psi_{i,f,j,t}\right)$ which was a function of landscape variables at the ranch level ($\beta_{0,j,f}$), the effects of *K* local environmental covariates (*i.e*,. variables affected by rangeland management variables) within transects ($\sum_{k=1}^{K} \beta_{j,k} X_{i,k}$), a random effect for the sampling year (τ_t), and a spatial autocorrelation term (ϵ_f) that was sampled from a multivariate normal with zero mean and variance-covariance (Σ) that was a function of the geographic distance between ranches. Here, the maximum covariance (η^2) between ranches *r* and *c* decreases exponentially with the squared distance among them $\exp\left(-\varphi^2 D_{r,c}^2\right)$, and the term

$d_{r,c}\sigma_s^2$ adds possible extra variability (σ_s^2) within a ranch. The (logit) expected probability of occupancy at a particular ranch when all local covariates where equal to zero, depended on whether it was in a valley or in a sierras area (L_f, 0 for valley and 1 sierras). Transect-level (local) covariates (i.e. variables affected by rangeland management) were grass height (cm), the presence or absence of tussocks, tree coverage (%), and whether the pasture is natural or artificial (0 or 1).

Species-specific responses to the environment (rangeland management and landscape) and species-specific detectability were estimated following an HMSC approach. In what follows we explain in detail the HMSC approach for environmental responses. For each species we need to estimate four betas and two deltas (see above). These parameters can be arranged in a matrix (Θ) with as many rows as species and as many columns as parameters. The vectorial form of this matrix is sampled from a multivariate distribution where the expected value for the *k*-th parameter of species *j* ($\mu_{j,k}$) is a linear function of species traits ζ.

$$\mu_{j,k} = \gamma_{0k} + \sum_{l=1}^{L} \gamma_{lk}\zeta_{jl}$$

$$\text{vec}(\Theta) \sim \text{MVN}(\text{vec}(\boldsymbol{\mu}), V \otimes [\rho C + (1-\rho) I])$$

The variance/covariance matrix (VCM) of the multinormal distribution controls the variability around expected values and how the parameters from different species are correlated. This VCM is obtained by combining, via the Kronecker product (⊗), the six-by-six VCM of the parameters (*V*) with the 69 by 69 phylogenetic correlation matrix (*C*) corrected by the parameter ρ. When ρ is close to one, phylogenetically related species will tend to have similar responses to the environment. When ρ is zero, the product is made over the 69 x 69 identity matrix *I* so that there is no effect of phylogeny on how similar the parameters from different species are. In our model, species traits related to their responses to rangelands management variables and landscape type were body size (log-transformed), degree of insectivory (%), degree of foraging on the ground stratum (100% indicates a species mostly uses the ground to forage, while 0% means the opposite) and whether they are gregarious or not (1 or 0).

Following a similar approach, we estimated species-specific detectability. We sampled the logit of the vector of species detection probabilities from a multivariate normal distribution with expected values given by a linear function of body size (log transformed), camouflage (yes or no), gregarism (yes or no), and how often species typically vocalize (frequent or not). Behavior data was obtained from Birds of the World (Billerman et al. 2022) and the authors' personal experience (Appendix S1: Table S1). Data on body size, degree of insectivory, and degree of foraging on the ground stratum were obtained from the Elton traits database (Wilman et al. 2014). Species phylogenetic correlations were derived from a phylogenetic tree obtained from Birdtree (Jetz et al. 2014).

Priors

The priors for trait effects on species environmental responses and detectability were normal distributions with zero mean and standard deviation of one. The prior for the variance-covariance matrix of environmental effects was constructed as the quadratic form of their variance which was sampled from a student t distribution with 3 degrees of freedom, mean 0 and a scale of 10, and correlation matrix sampled from a LKJ distribution with shape of two (McElreath 2020). For variance terms (species detectability, year random effects, and variability at the ranch level), we used as priors student t distributions with 3 degrees of freedom, mean zero and scale of 10. For phylogenetic effects in species-specific environmental responses and detectability we assumed a mixed prior that assigned 0.5 probability for 0 values (independence among species) and the remaining 0.5 probability to values from a Beta distribution (shape = 2, rate = 2). Finally, for the spatial autocorrelation across transects, η^2 and φ^2 had exponentially distributed priors with mean 2 and 0.5 respectively.

Table S1. Species recorded and traits value for each one. These values traits were used to fit the HMSC model. Sources are cited in the main text.

	Species traits					
Species	**Body size (g)**	**Insect (%)**	**Ground_ stratum (%)**	**Antipredation behavior**	**Gregarism**	**Vocalization (frequent or not)**
Agelaioides_badius	45.25	60	50	Detection	Yes	High
Alopochelidon_fucata	13.96	100	0	Detection	No	Low
Amblyramphus_holosericeus	56.76	40	100	Detection	No	Low
Ammodramus_humeralis	15.99	20	60	Camouflage	No	High
Anthus_correndera	20.9	90	80	Camouflage	No	Low
Anthus_furcatus	20.06	90	80	Camouflage	No	Low
Anthus_hellmayri	18.61	100	90	Camouflage	No	Low
Anumbius_annumbi	41.5	80	50	Camouflage	No	High
Athene_cunicularia	150.61	20	80	Camouflage	No	Low
Caracara_plancus	1078.62	20	100	Detection	No	Low
Colaptes_campestris	158	90	100	Detection	No	High
Colaptes_melanochloros	127.27	80	0	Detection	No	High
Cranioleuca_sulphurifera	13.5	100	0	Camouflage	No	Low
Donacospiza_albifrons	15.25	40	80	Camouflage	No	Low
Drymornis_bridgesii	92.62	90	100	Detection	No	High
Emberizoides_ypiranganus	20.25	40	20	Camouflage	No	Low
Embernagra_platensis	46	40	20	Camouflage	No	High
Falco_sparverius	114.61	60	50	Detection	No	Low
Furnarius_rufus	46.42	80	70	Detection	No	High
Gallinago_paraguaiae	109.44	100	80	Camouflage	No	High
Geothlypis_aequinoctialis	13.1	100	0	Camouflage	No	High
Gnorimopsar_chopi	65.9	40	60	Detection	No	High
Guira_guira	141	40	70	Detection	Yes	High
Hymenops_perspicillatus	22.9	100	100	Detection	No	Low
Leptasthenura_platensis	10.6	100	0	Detection	No	Low
Limnoctites_rectirostris	18.5	100	100	Camouflage	No	Low
Machetornis_rixosa	29.6	100	100	Detection	No	Low
Milvago_chimango	296	20	100	Detection	No	Low
Mimus_saturninus	63.7	100	100	Detection	No	High
Molothrus_bonariensis	41.49	70	100	Detection	Yes	Low
Molothrus_rufoaxillaris	47.5	30	90	Detection	Yes	Low
Myiopsitta_monachus	120	10	20	Detection	Yes	High
Nothura_maculosa	257.48	40	100	Camouflage	No	High
Paroaria_coronata	36.75	60	20	Detection	No	Low
Phacellodomus_striaticollis	25.7	100	70	Camouflage	No	High
Pitangus_sulphuratus	62.85	40	50	Detection	No	High
Poospiza_nigrorufa	17	40	80	Camouflage	No	Low
Progne_chalybea	42.9	100	20	Detection	No	Low
Progne_tapera	32	100	0	Detection	Yes	Low
Pseudoleistes_guirahuro	86.42	40	100	Detection	Yes	Low
Pseudoleistes_virescens	79.93	40	100	Detection	Yes	High
Pygochelidon_cyanoleuca	9.7	100	0	Detection	Yes	Low
Pyrocephalus_rubinus	14.4	100	0	Detection	No	Low

	Species traits					
Species	**Body size (g)**	**Insect (%)**	**Ground_ stratum (%)**	**Antipredation behavior**	**Gregarism**	**Vocalization (frequent or not)**
Rhynchotus_rufescens	843.47	40	80	Camouflage	No	High
Riparia_riparia	12.68	100	0	Detection	Yes	Low
Rostratula_semicollaris	76.6	60	50	Camouflage	No	Low
Satrapa_icterophrys	21.5	100	0	Detection	No	Low
Sicalis_flaveola	17.7	0	40	Camouflage	Yes	High
Sicalis_luteola	16.25	0	30	Camouflage	Yes	High
Spartonoica_maluroides	11	100	0	Camouflage	No	Low
Sporophila_caerulescens	10.5	20	50	Detection	No	Low
Stelgidopteryx_ruficollis	16.1	100	0	Detection	Yes	Low
Sturnella_superciliaris	45.32	40	90	Detection	Yes	Low
Syrigma_sibilatrix	463	70	0	Detection	No	High
Tachuris_rubrigastra	7.8	100	0	Camouflage	No	Low
Tachycineta_leucorrhoa	15.3	100	0	Detection	Yes	Low
Theristicus_caerulescens	1500	100	20	Detection	No	High
Troglodytes_aedon	10.85	80	0	Camouflage	No	High
Turdus_amaurochalinus	57.9	40	20	Detection	No	Low
Turdus_rufiventris	69.44	50	100	Detection	No	Low
Tyrannus_melancholicus	37.4	100	0	Detection	No	Low
Tyrannus_savana	31.9	70	50	Detection	No	Low
Vanellus_chilensis	327	90	80	Detection	No	High
Xanthopsar_flavus	43	100	70	Detection	Yes	Low
Xolmis_cinereus	57.1	100	100	Detection	No	Low
Xolmis_dominicanus	42.8	100	100	Detection	No	Low
Xolmis_irupero	28.7	100	100	Detection	No	Low
Zenaida_auriculata	110.2	0	20	Detection	Yes	Low
Zonotrichia_capensis	20.31	30	100	Detection	Yes	High

Table S2

Taxonomic list of recorded bird species, indicating conservation status (LC=least concern, NT=near threatened, VU= vulnerable, EN=endangered) based on BirdLife International (2023), Grassland specialits (1=yes, 0= no) and observed occupancy

Order	**Family**	**Species**	**Conservation status**	**Grassland specialist**	**Occupancy**
Tinamiformes	Tinamidae	*Rhynchotus_rufescens*	LC	1	0.0609
Tinamiformes	Tinamidae	*Nothura_maculosa*	LC	1	0.2534
Columbiformes	Columbidae	*Zenaida_auriculata*	LC	0	0.2456
Cuculiformes	Cuculidae	*Guira_guira*	LC	0	0.0491
Charadriiformes	Charadriidae	*Vanellus_chilensis*	LC	1	0.3546
Charadriiformes	Scolopacidae	*Gallinago_paraguaiae*	LC	1	0.0324
Charadriiformes	Rostratulidae	*Rostratula_semicollaris*	LC	0	0.002
Pelecaniformes	Ardeidae	*Syrigma_sibilatrix*	LC	0	0.0069

Order	**Family**	**Species**	**Conservation status**	**Grassland specialist**	**Occupancy**
Pelecaniformes	Threskiornithidae	*Theristicus_caerulescens*	LC	0	0.0177
Strigiformes	Strigidae	*Athene_cunicularia*	LC	1	0.0668
Piciformes	Picidae	*Colaptes_campestris*	LC	0	0.1523
Piciformes	Picidae	*Colaptes_melanochloros*	LC	0	0.0648
Falconiformes	Falconidae	*Caracara_plancus*	LC	0	0.0354
Falconiformes	Falconidae	*Falco_sparverius*	LC	0	0.0747
Falconiformes	Falconidae	*Milvago_chimango*	LC	0	0.0619
Psittaciformes	Psittacidae	*Myiopsitta_monachus*	LC	0	0.1847
Passeriformes	Furnariidae	*Anumbius_annumbi*	LC	0	0.2849
Passeriformes	Furnariidae	*Cranioleuca_sulphurifera*	LC	0	0.001
Passeriformes	Furnariidae	*Drymornis_bridgesii*	LC	0	0.0029
Passeriformes	Furnariidae	*Furnarius_rufus*	LC	0	0.4843
Passeriformes	Furnariidae	*Leptasthenura_platensis*	LC	0	0.001
Passeriformes	Furnariidae	*Limnoctites_rectirostris*	NT	0	0.0108
Passeriformes	Furnariidae	*Phacellodomus_striaticollis*	LC	0	0.1179
Passeriformes	Furnariidae	*Spartonoica_maluroides*	NT	0	0.001
Passeriformes	Tyrannidae	*Hymenops_perspicillatus*	LC	0	0.0688
Passeriformes	Tyrannidae	*Machetornis_rixosa*	LC	0	0.0972
Passeriformes	Tyrannidae	*Pitangus_sulphuratus*	LC	0	0.335
Passeriformes	Tyrannidae	*Pyrocephalus_rubinus*	LC	0	0.3468
Passeriformes	Tyrannidae	*Satrapa_icterophrys*	LC	0	0.0373
Passeriformes	Tyrannidae	*Tachuris_rubrigastra*	LC	0	0.001
Passeriformes	Tyrannidae	*Tyrannus_melancholicus*	LC	0	0.2328
Passeriformes	Tyrannidae	*Tyrannus_savana*	LC	1	0.5904
Passeriformes	Tyrannidae	*Xolmis_cinereus*	LC	0	0.1876
Passeriformes	Tyrannidae	*Xolmis_dominicanus*	VU	0	0.0354
Passeriformes	Tyrannidae	*Xolmis_irupero*	LC	0	0.331
Passeriformes	Hirundinidae	*Alopochelidon_fucata*	LC	0	0.003
Passeriformes	Hirundinidae	*Progne_chalybea*	LC	0	0.0118
Passeriformes	Hirundinidae	*Progne_tapera*	LC	0	0.4627
Passeriformes	Hirundinidae	*Pygochelidon_cyanoleuca*	LC	0	0.0766
Passeriformes	Hirundinidae	*Riparia_riparia*	LC	0	0.002
Passeriformes	Hirundinidae	*Stelgidopteryx_ruficollis*	LC	0	0.004
Passeriformes	Hirundinidae	*Tachycineta_leucorrhoa*	LC	0	0.3094
Passeriformes	Troglodytidae	*Troglodytes_aedon*	LC	0	0.0196
Passeriformes	Turdidae	*Turdus_amaurochalinus*	LC	0	0.0796

Order	**Family**	**Species**	**Conservation status**	**Grassland specialist**	**Occupancy**
Passeriformes	Turdidae	*Turdus_rufiventris*	LC	0	0.0373
Passeriformes	Mimidae	*Mimus_saturninus*	LC	0	0.3841
Passeriformes	Motacillidae	*Anthus_correndera*	LC	1	0.0108
Passeriformes	Motacillidae	*Anthus_furcatus*	LC	1	0.0147
Passeriformes	Motacillidae	*Anthus_hellmayri*	LC	1	0.0128
Passeriformes	Passerellidae	*Ammodramus_humeralis*	LC	1	0.0039
Passeriformes	Passerellidae	*Zonotrichia_capensis*	LC	0	0.2456
Passeriformes	Icteridae	*Agelaioides_badius*	LC	0	0.0226
Passeriformes	Icteridae	*Amblyramphus_holosericeus*	LC	0	0.0039
Passeriformes	Icteridae	*Gnorimopsar_chopi*	LC	0	0.0039
Passeriformes	Icteridae	*Molothrus_bonariensis*	LC	0	0.3762
Passeriformes	Icteridae	*Molothrus_rufoaxillaris*	LC	0	0.007
Passeriformes	Icteridae	*Pseudoleistes_guirahuro*	LC	0	0.0147
Passeriformes	Icteridae	*Pseudoleistes_virescens*	LC	0	0.22
Passeriformes	Icteridae	*Sturnella_superciliaris*	LC	1	0.0334
Passeriformes	Icteridae	*Xanthopsar_flavus*	EN	1	0.0413
Passeriformes	Parulidae	*Geothlypis_aequinoctialis*	LC	0	0.0069
Passeriformes	Thraupidae	*Donacospiza_albifrons*	LC	0	0.0206
Passeriformes	Thraupidae	*Emberizoides_ypiranganus*	LC	1	0.001
Passeriformes	Thraupidae	*Embernagra_platensis*	LC	1	0.2554
Passeriformes	Thraupidae	*Paroaria_coronata*	LC	0	0.0992
Passeriformes	Thraupidae	*Poospiza_nigrorufa*	LC	0	0.001
Passeriformes	Thraupidae	*Sicalis_flaveola*	LC	0	0.167
Passeriformes	Thraupidae	*Sicalis_luteola*	LC	1	0.3733
Passeriformes	Thraupidae	*Sporophila_caerulescens*	LC	0	0.0196

Table S3 Detection probabilities for all bird species. Posterior mean, lower and upper limits for the 95% credible intervals (lo025 and hi975) are shown. Rhat is a convergence statistic (convergence is assumed for Rhat close to 1). Bulk and tail effective sample sizes for posteriors are also reported.

spp	*probs*	*lo025*	*hi975*	*Rhat*	*ess bulk*	*ess tail*
Agelaioides badius	0.275	0.075	0.541	1.002	2198	2403
Alopochelidon fucata	0.255	0.005	0.717	1.001	1191	1672
Amblyramphus holosericeus	0.218	0.005	0.641	1.003	1386	1406
Ammodramus humeralis	0.372	0.017	0.823	1.004	1176	1202
Anthus correndera	0.21	0.008	0.65	1.003	735	1010
Anthus furcatus	0.239	0.028	0.557	1.004	1159	1187
Anthus hellmayri	0.152	0.012	0.514	1.003	912	873
Anumbius annumbi	0.498	0.407	0.595	1.001	2657	2979
Athene cunicularia	0.44	0.262	0.604	1	3486	3556
Caracara plancus	0.064	0.027	0.148	1.004	1459	2258
Colaptes campestris	0.226	0.16	0.304	1.004	1801	1978
Colaptes melanochloros	0.279	0.145	0.457	1	2489	3123
Cranioleuca sulphurifera	0.201	0.004	0.65	1.003	1305	2047
Donacospiza albifrons	0.167	0.046	0.362	1	1821	2820
Drymornis bridgesii	0.276	0.005	0.744	1.005	1331	1158
Emberizoides ypiranganus	0.174	0.004	0.609	1.001	1269	1881
Embernagra platensis	0.602	0.524	0.685	1.001	3716	3486
Falco sparverius	0.187	0.084	0.323	1.001	1586	1626
Furnarius rufus	0.625	0.574	0.682	1	3065	3382
Gallinago paraguaiae	0.369	0.091	0.789	1.001	1905	2144
Geothlypis aequinoctialis	0.417	0.027	0.835	1.002	1072	1363
Gnorimopsar chopi	0.341	0.015	0.793	1.002	1700	1989
Guira guira	0.237	0.054	0.486	1.008	1019	513
Hymenops perspicillatus	0.713	0.502	0.912	1.001	3622	3945
Leptasthenura platensis	0.232	0.001	0.701	1.003	1370	1219
Limnoctites rectirostris	0.148	0.023	0.359	1.002	1681	2452
Machetornis rixosa	0.132	0.087	0.192	1.003	1792	2733
Milvago chimango	0.205	0.09	0.384	1.003	1892	2453
Mimus saturninus	0.591	0.528	0.656	1	3470	3614
Molothrus bonariensis	0.553	0.486	0.627	1.001	3415	3698
Molothrus rufoaxillaris	0.096	0.005	0.337	1.004	822	1468
Myiopsitta monachus	0.339	0.255	0.435	1	3237	3364
Nothura maculosa	0.303	0.246	0.367	1.001	2310	1818
Paroaria coronata	0.241	0.138	0.365	1.002	2064	2930

spp	probs	lo025	hi975	Rhat	ess bulk	ess tail
Phacellodomus striaticollis	0.482	0.36	0.617	1.001	3179	1816
Pitangus sulphuratus	0.489	0.412	0.564	1	2677	3259
Poospiza nigrorufa	0.166	0.003	0.579	1.003	989	1785
Progne chalybea	0.124	0.012	0.392	1.002	1046	1009
Progne tapera	0.57	0.514	0.625	1	3418	3367
Pseudoleistes guirahuro	0.255	0.027	0.628	1.004	1673	2130
Pseudoleistes virescens	0.508	0.418	0.606	1	3606	3933
Pygochelidon cyanoleuca	0.188	0.108	0.307	1.001	2006	3176
Pyrocephalus rubinus	0.508	0.435	0.583	1.001	2847	3414
Rhynchotus rufescens	0.513	0.361	0.679	1.001	3473	3861
Riparia riparia	0.206	0.004	0.652	1.003	1437	1463
Rostratula semicollaris	0.143	0.004	0.55	1.001	1667	1745
Satrapa icterophrys	0.237	0.053	0.536	1.001	1383	1798
Sicalis flaveola	0.346	0.242	0.459	1.001	1947	2675
Sicalis luteola	0.534	0.474	0.597	1	3800	3864
Spartonoica maluroides	0.202	0.004	0.652	1.004	1215	1324
Sporophila caerulescens	0.345	0.054	0.795	1.003	1417	1985
Stelgidopteryx ruficollis	0.125	0.004	0.465	1.005	955	1189
Sturnella superciliaris	0.414	0.133	0.784	1.002	2653	3079
Syrigma sibilatrix	0.184	0.006	0.587	1.001	1683	1511
Tachuris rubrigastra	0.214	0.004	0.701	1.004	1037	1220
Tachycineta leucorrhoa	0.531	0.455	0.602	1	3666	3612
Theristicus caerulescens	0.098	0.022	0.268	1.002	1490	2242
Troglodytes aedon	0.475	0.093	0.883	1.002	1403	2202
Turdus amaurochalinus	0.48	0.274	0.72	1.001	2718	3389
Turdus rufiventris	0.276	0.082	0.561	1.001	1954	2512
Tyrannus melancholicus	0.421	0.324	0.514	1.001	2477	3232
Tyrannus savana	0.736	0.683	0.784	1.001	3550	3970
Vanellus chilensis	0.758	0.671	0.831	1	3769	3513
Xanthopsar flavus	0.458	0.211	0.705	1	3339	3772
Xolmis cinereus	0.436	0.337	0.545	1	3206	3723
Xolmis dominicanus	0.665	0.44	0.864	1	3400	3612
Xolmis irupero	0.407	0.347	0.474	1	2274	3451
Zenaida auriculata	0.137	0.027	0.367	1.004	1356	1769
Zonotrichia capensis	0.679	0.563	0.788	1.001	3322	3775

Table S4. Effect of traits on regression coefficients. Posterior mean, lower and upper limits for the 90% credible intervals (q5 and q95) and the fraction of the posterior with the same sign as the posterior mean (f) are shown. Rhat is a convergence statistic (convergence is assumed for Rhat close to 1). Bulk and tail effective sample sizes for posteriors are also reported.

Trait	*coefficient*	*mean*	*q5*	*q95*	*Rhat*	*f*	*ess_bulk*	*ess_tail*
intercept	grass height	-0.034	-0.260	0.203	1.00	0.62	3703	3247
size	grass height	-0.210	-0.352	-0.073	1.00	0.99	3467	3883
ground	grass height	-0.202	-0.325	-0.080	1.00	1.00	3742	4023
insectivory	grass height	-0.058	-0.179	0.066	1.00	0.78	3140	2582
gregarious	grass height	-0.024	-0.286	0.242	1.00	0.56	3804	3777
intercept	tussock	0.513	-0.038	0.958	1.00	0.94	2763	2541
size	tussock	-0.266	-0.563	0.037	1.00	0.92	3022	2673
ground	tussock	0.020	-0.237	0.289	1.00	0.55	3496	3919
insectivory	tussock	-0.002	-0.260	0.270	1.00	0.51	2609	1925
gregarious	tussock	0.018	-0.526	0.556	1.00	0.52	3431	3922
intercept	tree cover	0.183	-0.093	0.501	1.00	0.87	3856	3655
size	tree cover	-0.169	-0.347	0.003	1.00	0.95	3557	3722
ground	tree cover	-0.073	-0.228	0.081	1.00	0.79	3614	3748
insectivory	tree cover	-0.089	-0.241	0.069	1.00	0.82	2882	2888
gregarious	tree cover	-0.208	-0.537	0.137	1.00	0.84	3752	3252
intercept	pasture type	-0.401	-0.622	-0.175	1.00	1.00	3512	3666
size	pasture type	-0.179	-0.361	-0.005	1.00	0.95	2903	3581
ground	pasture type	0.072	-0.082	0.227	1.00	0.79	2982	3303
insectivory	pasture type	-0.110	-0.264	0.041	1.00	0.89	3387	3578
gregarious	pasture type	0.475	0.158	0.799	1.00	0.99	3135	3667
intercept	valley	-1.801	-2.861	-0.495	1.00	0.99	1310	2009
size	valley	0.851	0.190	1.492	1.00	0.98	1821	2557
ground	valley	0.042	-0.537	0.626	1.00	0.55	2783	3557
insectivory	valley	0.314	-0.245	0.868	1.00	0.82	1931	1916
gregarious	valley	0.841	-0.228	1.938	1.00	0.89	2362	3406
intercept	sierras	-0.131	-0.465	0.258	1.00	0.74	2538	3270
size	sierras	-0.356	-0.580	-0.146	1.00	1.00	3168	3399
ground	sierras	-0.219	-0.410	-0.036	1.00	0.98	3720	3840
insectivory	sierras	0.061	-0.127	0.245	1.00	0.71	3664	3364
gregarious	sierras	-0.310	-0.705	0.081	1.00	0.90	3544	3770

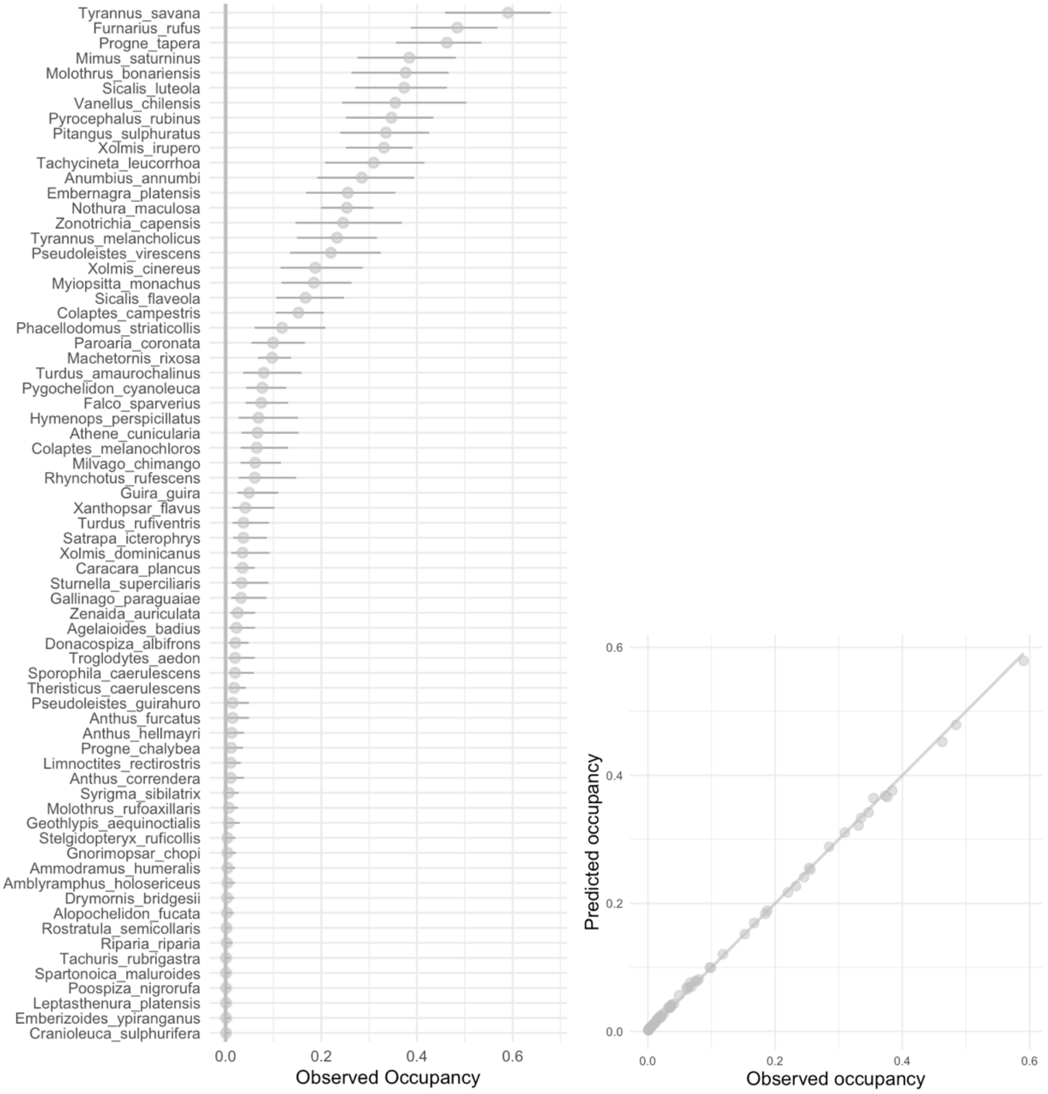


Figure S1. Posterior predictive check of species observed occupancy (averaged across years). Left panel: grey lines show 95% credible intervals across 4000 posterior simulations, gray dot observed values (proportion of times a species was detected in a transect). Right panel: relationship between predicted and observed occupancy, the line shows 1:1 value.

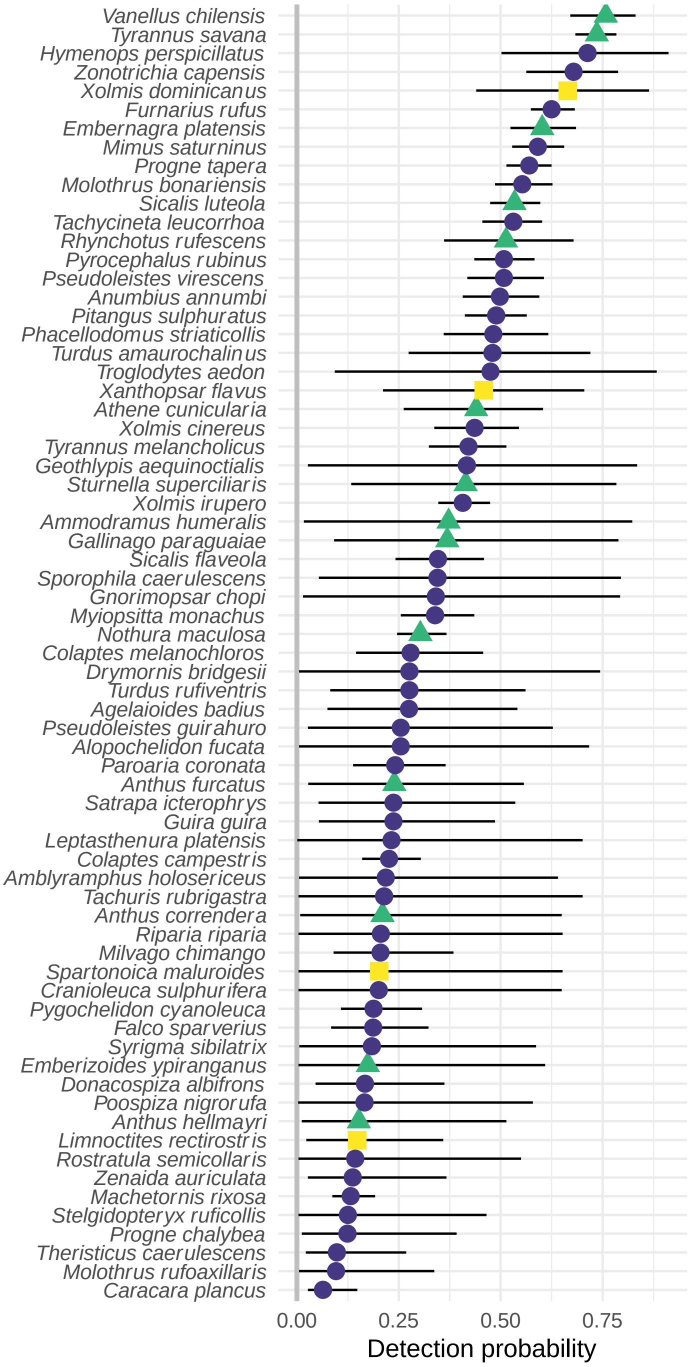


Figure S2. Detection probability for all bird species. Dots show posterior means and horizontal black lines the 95% credible intervals with grey bands showing 80% credible intervals. Green triangles are grassland specialists while yellow squares are threatened species.

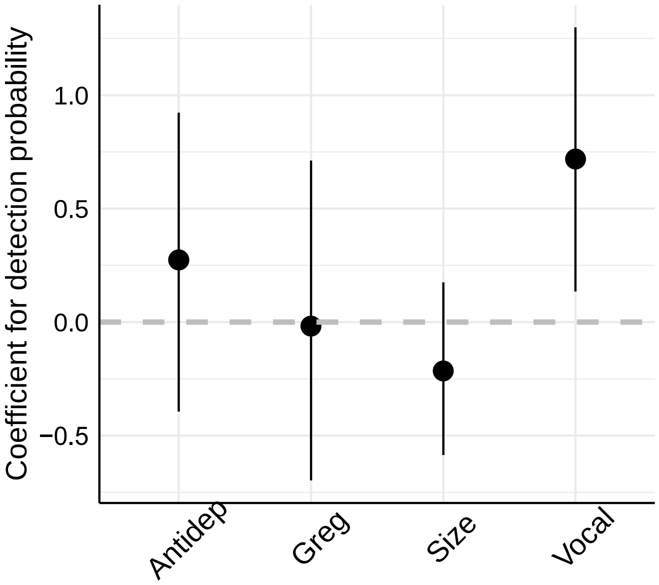
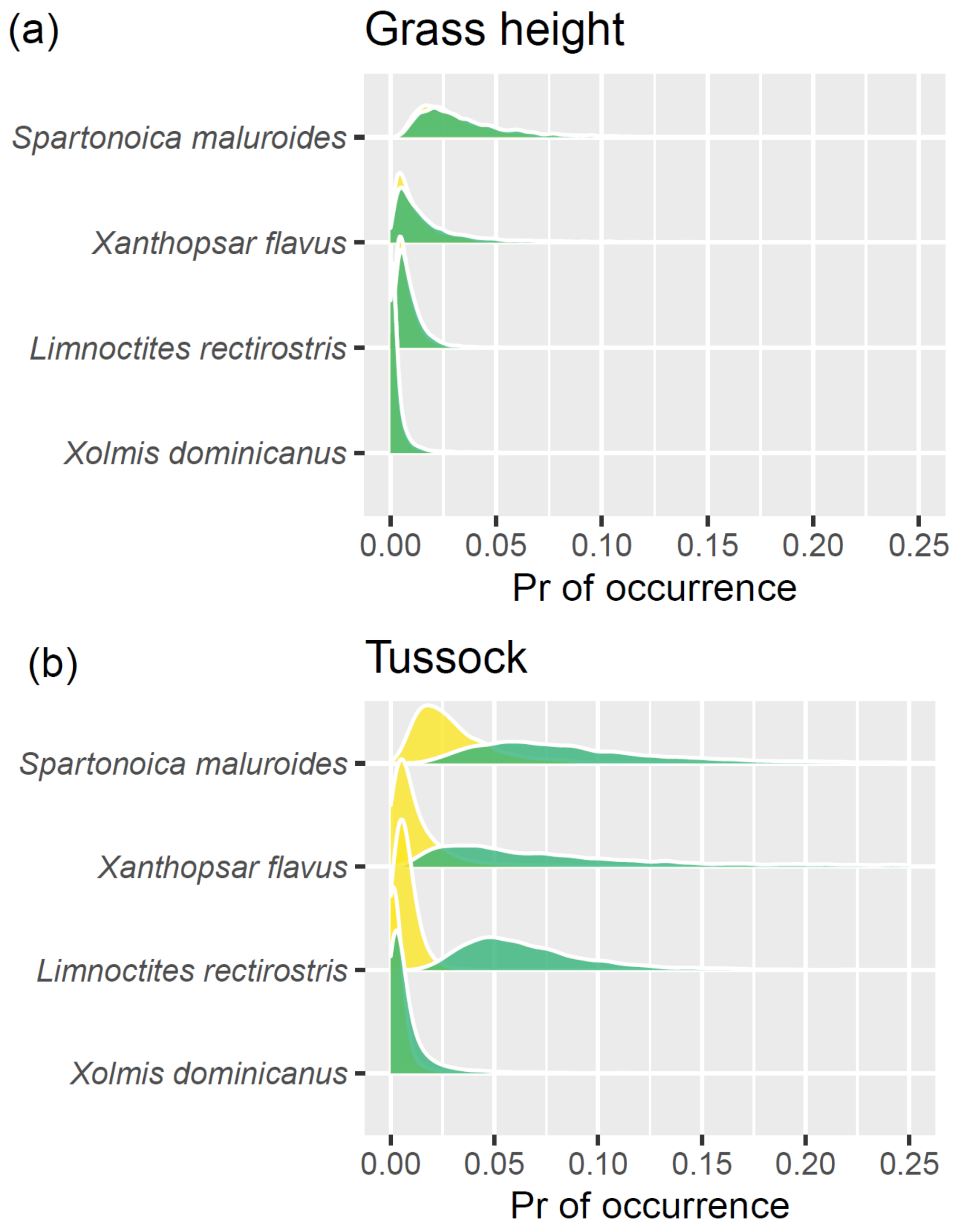


Figure S3. Effect of bird traits on the probability of detection. Antidep denotes those species that showed different antidepredatory behaviors. Greg is for gregarious species, Size denote bird body size, and Vocal is for species that vocalize frequently. Only vocalization had a clear effect on detection probability.


(a)
Grass height
Spartonoica maluroides
Xanthopsar flavus
Limnoctites rectirostris
Xolmis dominicanus
0.00
0.05
0.10
0.15
0.20
0.25
Pr of occurrence
(b)
Tussock
Spartonoica maluroides
Xanthopsar flavus
Limnoctites rectirostris
Xolmis dominicanus
0.00
0.05
0.10
0.15
0.20
0.25
Pr of occurrence

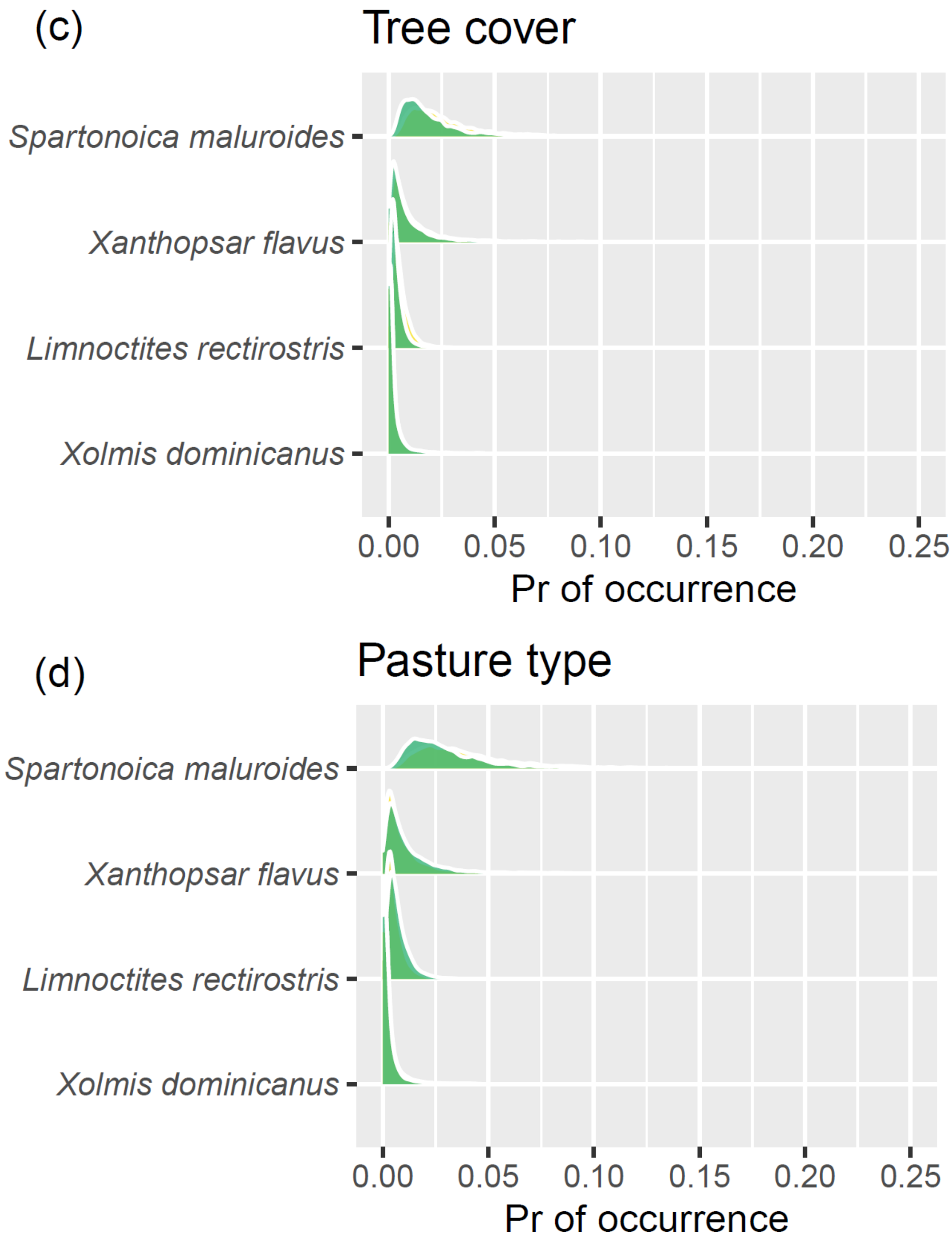


Figure S4. Threatened bird species predicted responses to (a) doubling in grass height from its mean of 11.76 cm, (b) the inclusion of tussock in the paddock, (c) Doubling tree cover from its mean of 4.97%, and (d) changing an artificial pasture to a natural one. The yellow shade are posterior probabilities of occurrence under average conditions in a valley, and the green shade the predicted posterior after the change in the environment.

Figure S5. Hill and valley landscape in Aigua.

3. Managing Grass Height for Birds and Livestock: Insights from the Río de la Plata Grasslands

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3.1. Resumen

El éxito en la conservación de la biodiversidad en los pastizales depende de la gestión efectiva de los sistemas para lograr resultados económicos positivos al mismo tiempo que se preserva la biodiversidad. En los pastizales nativos de la región del Río de la Plata, la producción de ganado a menudo no alcanza su potencial debido a la baja disponibilidad de forraje. Estrategias de manejo del ganado que promuevan una mayor altura del pasto pueden aumentar la masa de forraje sin reemplazar la vegetación nativa, mejorando la ingesta de materia seca del ganado y potencialmente aumentando la producción. Sin embargo, el impacto de tales cambios en las poblaciones de aves de pastizales sigue siendo incierto. En este estudio, implementamos cambios en el manejo para aumentar la altura del pasto en seis predios y evaluamos las respuestas de aves especialistas de pastizales. Inicialmente, la altura promedio del pasto fue de 6 cm, pero después de los cambios en el manejo, aumentó a un promedio de 12 cm. También aumentó la heterogeneidad del pasto, lo cual es deseable para apoyar una mayor riqueza de aves de pastizales. Encontramos que ni las abundancias de especies individuales de aves de pastizales ni la riqueza variaron después de los cambios en el manejo. Además, las especies amenazadas de aves de pastizales que dependen estrictamente de pastos más altos no colonizaron los potreros después de las alteraciones. Por lo tanto, concluimos que mantener la abundancia de siete especialistas de pastizales y potencialmente aumentar la producción de ganado es factible, ya que la altura del pasto alcanzada coincidió con niveles conocidos para maximizar la tasa de ingesta de materia seca del ganado. Sin embargo, la altura promedio del pasto alcanzada no fue lo suficientemente alta como para mantener a las especialistas en pastos altos en peligro de extinción. Es posible que marcos temporales más largos permitan que los pajonales y otras cespitosas logren mayores alturas y proporcionen condiciones de hábitat adecuadas. La mayoría de las investigaciones anteriores sobre la biodiversidad en pastizales se han centrado en evaluar las respuestas de la fauna en áreas sin pastoreo o fuertemente pastoreadas. Por lo tanto, nuestros hallazgos contribuyen a una mejor comprensión de las respuestas de la fauna dentro de este espectro y se alinean con un enfoque de producción-conservación mutuamente beneficioso, que puede ayudar a preservar la biodiversidad en escalas geográficas amplias.

Palabras clave: Conservación de biodiversidad, ecología, poblaciones de aves de pastizal, producción ganadera, manejo de sistemas ganaderos

3.2. Summary

Success in conserving biodiversity in rangelands depends on effectively managing the systems to achieve positive economic outcomes while preserving biodiversity. In the native grasslands of the Río de la Plata region, livestock production often falls below its potential due to low forage availability. Cattle management strategies that promote taller grass height can increase forage mass without replacing native vegetation, enhancing cattle dry matter intake and potentially boosting production. Nonetheless, the impact of such changes on grassland bird populations remains unclear. In this study, we implemented management changes to increase grass height across six ranches and evaluated the responses of grassland bird specialists. Initially, the average grass height was 6 cm, but after the management changes, it increased to an average of 12 cm. This also increased grass heterogeneity which is desirable for supporting a higher richness of grassland birds. We found that both the abundances of individual grassland bird species and richness did not vary after the management changes. Moreover, threatened grassland bird species that rely strictly on taller grasses did not colonize the paddocks following the alterations. Therefore, we conclude that maintaining the abundance of seven grassland bird specialists and potentially increasing livestock production is feasible, as the attained grass height coincided with levels known to maximize cattle dry matter intake rate. However, achieved average grass height was not sufficiently tall to support endangered tall grass specialists. It is possible that longer time frames allow tussocks to grow taller and provide such habitat conditions. Most previous research on rangelands biodiversity has focused on assessing faunal responses in either ungrazed or heavily grazed areas. Therefore, our findings contribute to a better understanding of faunal responses within this spectrum and align with a win-win production-conservation approach, which can aid in safeguarding biodiversity across large geographic scales.

Keywords: biodiversity conservation, ecology, grassland bird populations, livestock production, rangeland management

3.3. Introduction

To make a significant impact on global biodiversity conservation, it is crucial to prioritize measures within extensive production systems(Gurney et al. 2021). This is necessary because although protected areas play a vital role in biodiversity conservation, their coverage is limited to a small fraction of the Earth's surface (Ellis 2019). Moreover, establishing a matrix of suitable habitats is important to facilitate species movement between protected areas, and extensive productive landscapes have the potential to provide such habitats (Franklin and Lindenmayer 2009). Additionally, it should be noted that certain extractive activities, including extensive domestic cattle grazing, are allowed in many protected areas (Oldekop et al. 2016) emphasizing the need for biodiversity-focused management practices. Among these productive systems, rangelands in native grasslands are widely recognized as ecosystems with significant potential for both biodiversity conservation and livestock production (Alkemade et al. 2013, Gennet et al. 2017).

Studies have shown that ranchers demonstrate greater interest in biodiversity conservation when conservation actions also contribute to improving cattle production outcomes (Cortés-Capano et al. 2021). Hence, the success of biodiversity conservation in rangelands hinges on the ability to manage these systems in a way that simultaneously achieves positive economic results and maintains biodiversity. However, previous research has been ineffective in addressing both production and conservation challenges concurrently. One reason for this is the predominant focus of biodiversity studies solely on biodiversity preservation, without considering the outcomes for rangeland production. As a result, there is limited understanding of rangeland management practices that can effectively conserve biodiversity while also enhancing livestock production outcomes (Neilly et al. 2016).

The Rio de la Plata Grasslands, located in the Southern Cone of South America, are recognized as one of the largest temperate grasslands globally and are of utmost importance for global conservation efforts (Hoekstra et al. 2005). Research conducted in this region has demonstrated that grass height plays a critical role in influencing both wildlife and livestock productivity. Therefore, maintaining appropriate grass height is crucial for simultaneously conserving wildlife and achieving positive economic outcomes. Among the diverse fauna inhabiting these grasslands, birds are particularly significant due to their abundance and diversity. Grassland bird species exhibit varying environmental selection, ranging from those specialized in short grass to those favored by tall grass (Azpiroz et al. 2012). As a result, grassland birds are highly sensitive to the impact of livestock grazing on grass height (Azpiroz and Blake 2016, Dias et al. 2017, Barzan et al. 2021). Furthermore, in order to meet the habitat requirements of a wide range of grassland bird species, management practices that promote grass height heterogeneity have been strongly recommended (Fuhlendorf et al. 2006, 2017, Isacch and Cardoni 2011, Agra et al. 2015, Azpiroz and Blake 2016, Dias et al. 2017, Pírez and Aldabe 2022).

Grass height has been shown to have an impact on livestock production as well (Da Trindade et al. 2016). When grass height is excessively short, it restricts the intake of forage by livestock. Conversely, higher grass heights have been found to increase herbage production levels and enhance cattle's efficiency in utilizing the available forage (Claramunt et al. 2018, Do Carmo et al. 2018b). However, it's important to note that excessively tall grass heights can lead to lower intake rates of essential nutrients. Therefore, the key to successfully promoting both bird conservation and livestock production in rangelands lies in managing an optimal grass height that simultaneously supports both objectives. However, our understanding of how to effectively manage grass height to positively impact both cattle production and grassland birds is currently limited.

In the Rio de la Plata Grasslands, most grazing systems in native grasslands traditionally maintain low grass heights, which limits their productive capacity (Soriano et al. 1992, Modernel et al. 2019). Previous research has demonstrated that by implementing cattle management practices, it is possible to increase grass height, resulting in higher herbage mass that benefits livestock production and other ecosystem services (Ruggia et al. 2021). However, the response of bird species to such changes in grass height remains little explored. Therefore, our study aimed to evaluate whether increasing grass height through cattle management can effectively conserve bird species that rely exclusively on grassland habitats (i.e., grassland specialists). To achieve this, we selected ranches with traditional grazing management practices characterized by low average grass heights and implemented management changes, including strategies to increase grass height. Subsequently, we assessed the response of grassland bird specialists. The findings of our study will contribute to filling the significant knowledge gap regarding faunal responses in areas with intermediate levels of grazing pressure, bridging the gap between high grazing intensity and non-grazing livestock systems.

3.4. Materials and methods

3.4.1. Study area

The Rio de la Plata Grasslands encompass two main sub-regions: Pampas and Campos (Fig. 1). These sub-regions are distinguished based on their physiognomic, geomorphologic, and edaphic characteristics. The Pampas can be described as an extensive and continuous plain of grasslands with minimal tree formations, while the Campos in Uruguay and southern Brazil are characterized by steeper terrains and a higher occurrence of tree formations and wetlands (Soriano et al. 1992, Oyarzabal et al. 2020).

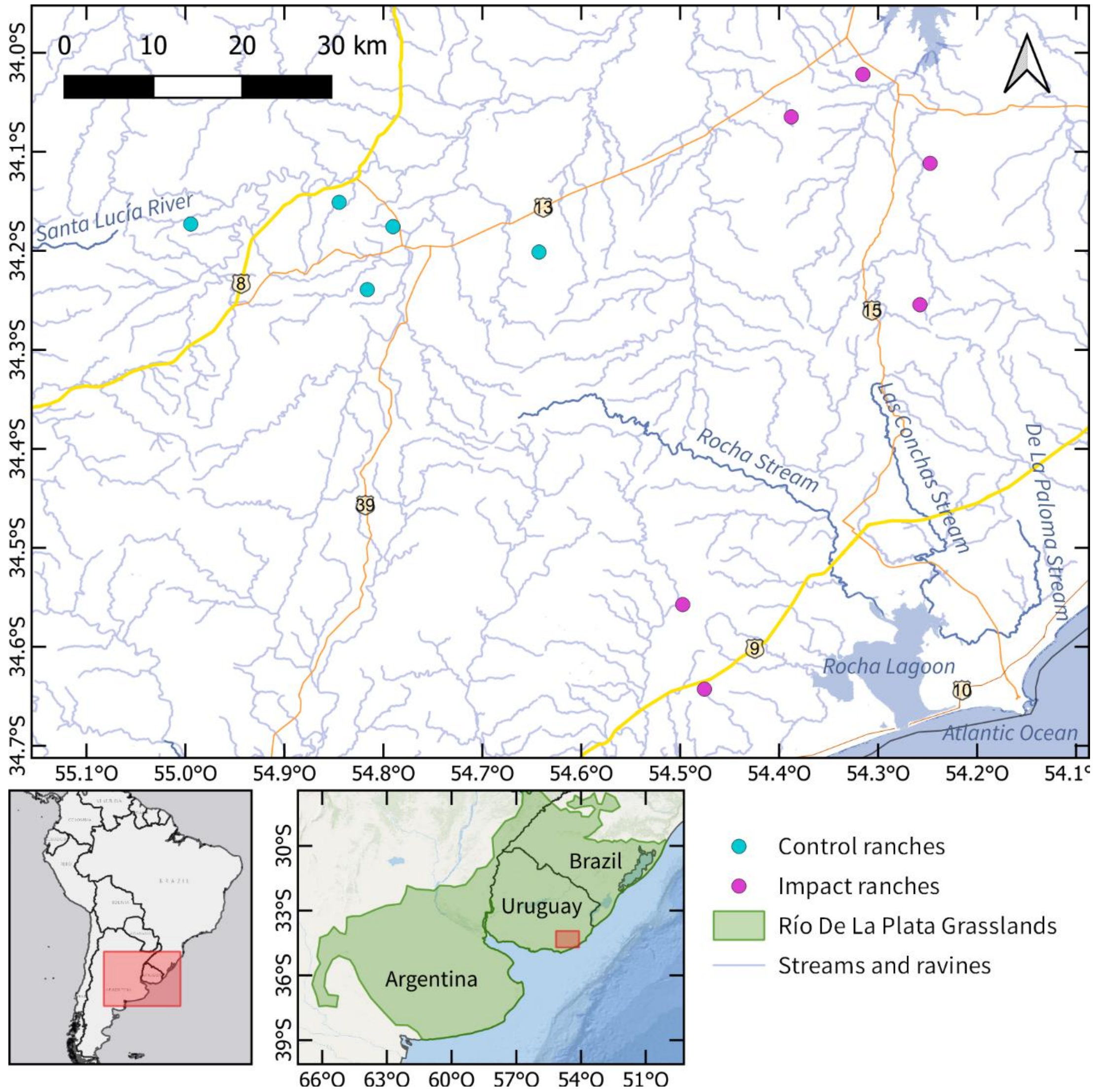

Figure 1. Study area and management categories in the Rio de la Plata Grasslands. The map shows the location of the Rio de la Plata Grasslands in the Southern Cone of South America. The study area encompasses both impact ranches, where cattle management changes were implemented, and control ranches, where no changes in management occurred.

This particular study was conducted in the Campos ecoregion, located in southeastern Uruguay, in the Eastern Hill grasslands region (Lezama et al. 2019). The research focused on 11 livestock ranches that are predominantly covered by natural grasslands (mean natural grassland area per ranch was 70%; range: 55%-95%). According to Lezama et al. (2019) the grass communities in our studied region were characterized by *Andropogon ternatus, Aira elgantissima, Danthonia rhizomata. Chevreulia sarmentosa, Bothriochloa laguroides, Chaptalia exscapa* and *Danthonia montevidensis*. In the area, there is also a relevant forb component, dominated by *Eryngium horridum*, as well as a sub-shrub component. The vegetation presents a double stratum physiognomy, with a low stratum of ca. 5 cm tall constituted by prostrate grasses and forbs, and a 30 cm tall stratum of erect grasses and sub-shrubs (Lezama et al. 2019). Frequent tussock species are *Erianthus angustifolius* and *Paspalum quadrifarium* (Aguerre and Albicete, 2018). Although spatial variation in grass species composition has been documented in the region (Lezama et al. 2019), the selected ranches are in the same geomorphological region and landscape unit (Panario et al. 2014). The area where all ranches were located is considered an ornithological unit ("Sabana esteparia serrana" according to Arballo and Cravino 1999). The average size of the ranches was 217 hectares, with a range of 90 to 413 hectares. The ranches were further divided into paddocks, with an average size of 23.7 hectares and a range of 2.7 to 104 hectares.

These ranches specialize in animal reproduction, specifically cow-calf production systems, and rely on the utilization of natural grasslands. However, their management practices traditionally maintain pasture heights below their productive potential, with grass heights typically below 5 cm.

Fifty-two grassland bird species were recorded in the studied area (Pírez and Aldabe, 2022) where the most common species were Molothrus bonariensis, Furnarius rufus, Nothura maculosa, Tyrannus savana, and Vanellus chilensis. All of these species inhabit short grasslands. Additionally, tall grassland species, such as Embernagra platensis, Phacellodomus striaticollis, and Sicalis luteola, were frequently observed in this region (Pírez and Aldabe, 2022). Other bird species capable of utilizing tall grasslands and tussocks, but recorded in the region with low frequency, included Donacospiza albifrons, Xanthopsar flavus, Xolmis dominicanus, Emberizoides ypiranganus and Hymenops perspicillatus. Occasional records have been obtained for tall grass and threatened species, such as Sporophila palustris (Azpiroz, 2003), and Sporophila cinnamomea (Blumetto and Castagna, 2019).

3.4.2. Grazing management changes

In our study, we selected a total of eleven ranches located in eastern Uruguay. Out of these, six ranches implemented cattle management changes and were considered as the "Impact" group, while the remaining five ranches maintained their existing management practices and served as the "Control" group.

To increase grass height in the impact group ranches, we undertook several actions. Firstly, the main action was to redistribute stocking rate among the paddocks. In each paddock, we adjusted the stocking rate according to the grass height and the cattle' requirements. In this sense, cattle with lower requirements were assigned to paddocks with lower height (between 3 and 5 cm), while animals with higher requirements were assigned to paddocks with a height greater than 5 cm, allowing relief in other paddocks. This adjustment was carried out on a monthly basis over the course of one year. We also decreased stocking rate at the beginning of the study. On average, ranches decreased their stocking rate from 0.84 cattle units (range: 0.6-1.24) to 0.79 cattle units (range: 0.53-0.95).

Secondly, reducing the sheep to cattle ratio was one of the strategies to increase forage height. In four out of the six impact ranches, the sheep to cattle ratio was decreased. This decision was based on the understanding that higher relationships between numbers of sheeps and cattles tends to result in shorter grass height (Morris et al. 1999).

Lastly, the reproductive strategy of ranches was changed. Previously, in most ranches, the mating used to be all year round or in two different seasons. Consequently, there were calves all year round or in two different seasons. This contributed to shorten grass height because of an overlap of the highest nutritional requirements of cattle (cows with calves at their side) with the season when grass grows very little (winter). Therefore, we restricted the mating period to 90 days during spring and summer. Consequently, calves were born in the spring, which is characterized by the highest grass growth. This strategic timing aimed to support higher grass height during critical periods.

It is worth noting that the specific management actions applied depended on the circumstances of each ranch. When all the paddocks in a particular ranch had short grass, priority was given to reduce stocking rate. Most livestock management changes occurred between April and December 2017.

3.4.3. Before-after control-impact design

In our study, we employed a before-after control-impact (BACI) design (Green 1979), which is a widely used approach to assess the effects of management changes on ecosystems when random assignment of treatment and control sites is not possible. The BACI design incorporates time and impact factors, comparing data collected before and after the management changes at both the control and impact sites.

By using the BACI design, we aimed to evaluate the specific impacts of grazing management changes on grass height, grass heterogeneity, and bird abundances of different species. This design allows us to account for potential confounding factors and reduce the influence of unmeasured variables that could affect the observed effects.

The control sites served as a reference for comparison, representing the conditions and dynamics of the grassland ecosystem without the implemented management changes. The impact sites, on the other hand, represented the areas where the specific management actions were applied to increase grass height and promote grassland bird conservation.

By collecting data before and after the management changes at both control and impact sites, we can analyze and compare the changes in grass height, grass heterogeneity, and bird abundances over time. This approach provides a robust framework to assess the effectiveness of the grazing management changes and their potential impact on the ecosystem.

Our sampling unit was the paddock, while management changes were implemented at the ranch level. We randomly selected a total of 71 paddocks from the 11 ranches. All paddocks were natural grasslands (i.e. we avoided artificial pastures). At the beginning of the study, all paddocks had relatively short grass on average. Subsequently, we selected a subset of 42 paddocks in 6 ranches where we implemented the described cattle management changes. These 42 paddocks were designated as the treatment group, referred to as "impact paddocks" henceforth. The remaining 29 paddocks, located in 5 ranches, maintained their traditional management practices throughout the study and served as the control group. We assessed grass height and bird abundances in all 71 paddocks before and one year after the implementation of the management changes.

3.4.4. Grass height measurements

Grass height was assessed in each paddock by measuring it at regular intervals along a 300-meter transect, with measurements taken every 50 meters. This resulted in a total of six measurements per paddock. A metal ruler was used to determine the maximum height at which the majority of the grass biomass was concentrated (Fig 2). The standard deviation of grass height along each transect was utilized as an indicator of grassland heterogeneity.

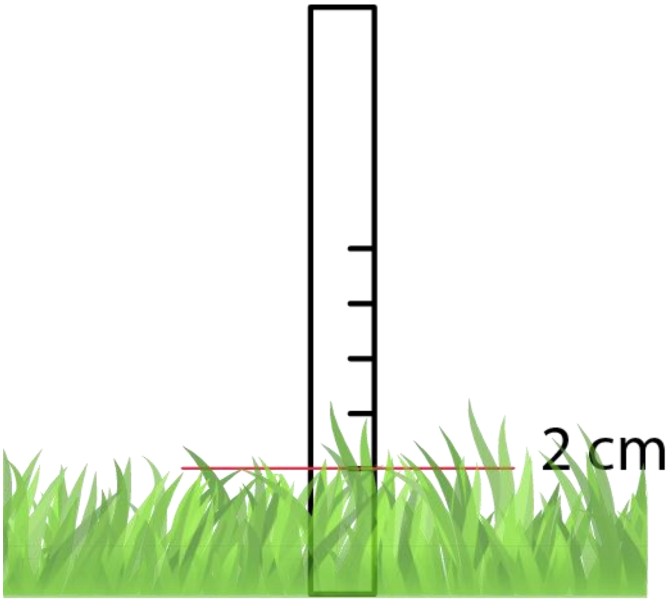

Figure 2: Measurement and recording of grass height. Grass height at each sampling point was measured using a ruler. The measurement was taken to determine the maximum height at which the majority of the grass biomass was accumulated. This value was then recorded for further analysis. In the case of this figure we show, as example, a grass height of 2 cm.

3.4.5. Bird Surveys

We conducted our study on grassland bird species that depend entirely on grasslands for their whole life cycle (Vickery et al. 1999). The continuous conversion of grasslands into alternative land uses in the Rio de la Plata Grasslands (Baeza and Paruelo 2020) presents a significant risk to these species. Hence, conserving the species that rely exclusively on native grasslands becomes a matter of utmost importance (Azpiroz et al. 2012) .

In each paddock, we conducted bird identifications and counts before (December 2017) and after (December 2018) implementing cattle grazing management changes. We selected December to do bird surveys as this is the breeding season and when the birds are more active and easier to detect. Furthermore, grass height differences between control and impact paddocks are expected to be higher in December, immediately after the highest grass growing rate season. To sample obligate grassland birds, we used linear transects of 300 m without a fixed bandwidth. A unique transect line was randomly placed within each paddock, and two observers walked along each transect, recording and identifying individual birds. We only recorded birds that were actively using the habitat within the transect, such as feeding, resting, or perching, excluding those birds that were merely flying over the surveyed area. Bird surveys were conducted during the first 4 hours of the morning or the last 4 hours of the afternoon. To account for detection heterogeneity (Miller et al., 2015), the timing of bird surveys was randomized across ranches and paddocks. To estimate the detection probability of bird species (MacKenzie et al. 2002), we repeatedly surveyed birds in the same transects established in different paddocks (between 5 and 7 transects in each ranch), within a short period of time (2 to 4 times over a 3-day period).

Our study focused on analyzing changes in the abundance of individual bird species before and after implementing grazing management changes. This approach at the species level was chosen based on evidence from previous studies that have highlighted the significance of specific compositional changes and the responses of individual species to grazing in informing off-reserve conservation on rangelands (Neilly et al. 2016, Dias et al. 2017). We recognized that assessing the responses of individual species may provide valuable insights that go beyond overall biodiversity measures.

3.4.6. Data modeling

In a model for data from a BACI design, there are two fixed effects: Before-After (BA) and Control-Impact (CI). However, considering these main effects individually may not provide informative results for impact assessment. Differences in time (BA) could be due to natural temporal variation, and CI effects could be influenced by natural spatial random variation, indicating inherent differences between control and impact sites. It is the interaction term between BA and CI that indicates whether the experimental treatment has truly influenced the impact sites. When the interaction term shows a significant effect, it suggests that the time (BA) effect depends on the site treatment level (CI) (Mcdonald et al. 2000).

We distinguished between Control and Impact paddocks using a variable called PadTreat, which had two levels: "Control" and "Impact." Additionally, we identified seasons as either Before or After the treatment using a seasonal paddock-specific variable called SeasonBA, with the levels "Before" and "After." Both PadTreat and SeasonBA factors were covariates that varied at the paddock level. It's important to note that all paddocks within a ranch received the same treatment, either Control or Impact.

To estimate each bird species abundance for paddock i ($i = 1, \ldots 141$) and survey j ($j = 1, \ldots 4$) we specified the following binomial N-mixture model, that allowed us to estimate each species abundance considering imperfect detection (Royle 2004):

$$C_{i,j} \sim Binomial(N_i, p_{i,j})$$

$$N_i \sim Poisson(\lambda_i)$$

$$\begin{aligned} log(\lambda_i) = \beta_{0,R(i)} + \beta_1 \times SeasonBA + \beta_2 \times PadTreat \\ + \beta_3 \times SeasonBA.PadTreat \end{aligned}$$

$$\beta_{0,R} \sim Normal(\mu_R, \sigma_R^2)$$

In our analysis, we used a random slope N-mixture model to estimate the latent abundance of each species. The count at paddock i during survey j is denoted as $C_{i,j}$. Ni represents the realized abundance at paddock i and p the probability of detection in survey j in paddock i. We modeled the expected latent abundance Ni using a generalized linear model (GLM) with one set of random effects. Specifically, on the log scale, Ni was expressed as a sum of a random ranch effect (R), (β_1), treatment (β_2) and of the interaction of these two factors (β_3). We employed a 95% credible interval for estimating the uncertainty in the model parameters. We used the ubms R package (Kellner et al. 2022) to fit the random slope N-mixture model, fitting one model per species.

To model the effect of treatment on grass height, we specified the following model:

$$GrassHght_{h,i,R} \sim Normal\ (\mu_{iR}, \sigma_R^2)$$

$$\mu_{i,R} = v_{0,R} + l_{0,[R]i} + \beta_1 \times SeasonBA + \beta_2 \times SiteTreat \\ + \beta_3 \times SeasonBA.SiteTreat$$

$$v_{0,R} \sim Normal(\mu_R, \sigma_R^2)$$

$$l_{0,[R]i} \sim Normal(\mu_i, \sigma_i^2)$$

where $GrassHght_{h,i,R}$denotes the grass height observation h at paddock i in ranch R. $\mu_{i,R}$ represents the expected grass height at paddock i in ranch R. Grass height was modeled according to a GLM with two sets of random effects: $v_{0,R}$ corresponds to the random ranch effect, and $l_{0,[R]i}$ corresponds to the random paddock effects nested within the random ranch effect. The fixed effects in the model were the season (SeasonBA), treatment (SiteTreat), and the interaction of these two factors (SeasonBA.SiteTreat), denoted β_1, β_2) and β_3), respectively. To fit this model, the lme4 package was used (Bates et al. 2015) . By estimating the parameters in this model, we can determine the effect of treatment on grass height while considering the random effects of the ranch and paddock.

To model the effect of treatment on the standard deviation of grass height (grass height heterogeneity), we specified the following model:

$$GrassHet_{i,R} \sim Normal\ (\mu_i, \sigma_R^2)$$

$$\mu_i = v_{0,R} + \beta_1 \times SeasonBA + \beta_2 \times SiteTreat + \beta_3 \times SeasonBA.SiteTreat$$

$$v_{0,R} \sim Normal(\mu_R, \sigma_R^2)$$

Here, $GrassHet_{R(i)}$, denotes the standard deviation of grass height in paddock i in ranch R. μ_i was expressed as the sum of a random ranch effect ($v_{0,R}$) and effects of the season (β_1), treatment (β_2) and of the interaction of these two factors (β_3). We used lme4 package to fit this model.

The random ranch effect, $(v_{0,R})$, follows a normal distribution with mean μ_R and variance σ_R^2. By incorporating the random ranch effect and the fixed effects, we can estimate the expected standard deviation of grass height in each paddock, accounting for the effects of season and treatment. Parametric modeling assumptions as well as model fitting were tested using posterior predictive checks and residual analysis.

3.5. Results

Our study found that the implemented grazing management change had a positive impact on grass height. Specifically, we observed a significant increase in grass height in the impact paddocks, whereas no significant change was observed in the control paddocks (Fig. 3). The statistical model we used showed that the interaction between the season (SeasonBA) and treatment (PadTreat) had a significant effect on grass height (df=842, F=31.3, p<<0.01). Grass height in the impact paddocks increased from an initial average of 6 cm to 12 cm, while grass height in the control paddocks remained below 6 cm (Fig. 3). These findings provide strong evidence that the grazing management change led to an increase in grass height in the impact paddocks compared to the control paddocks.

Additionally, we observed a significant effect of treatment on the standard deviation of grass height, which represents grass patchiness. Our statistical model revealed a significant interaction effect between the season (SeasonBA) and treatment (PadTreat) on the standard deviation of grass height (df=133, F=4.82, p<0.05; Fig. 4). In the impact paddocks, the standard deviation of grass height increased from an initial value of 3 cm to 6.5 cm after the management change, indicating a higher level of patchiness. Conversely, in the control paddocks, the standard deviation remained relatively stable at around 4 cm throughout the study period (Fig.4). These results suggest that the grazing management change influenced the spatial heterogeneity of grass height, leading to increased patchiness in the impact paddocks compared to the control paddocks.

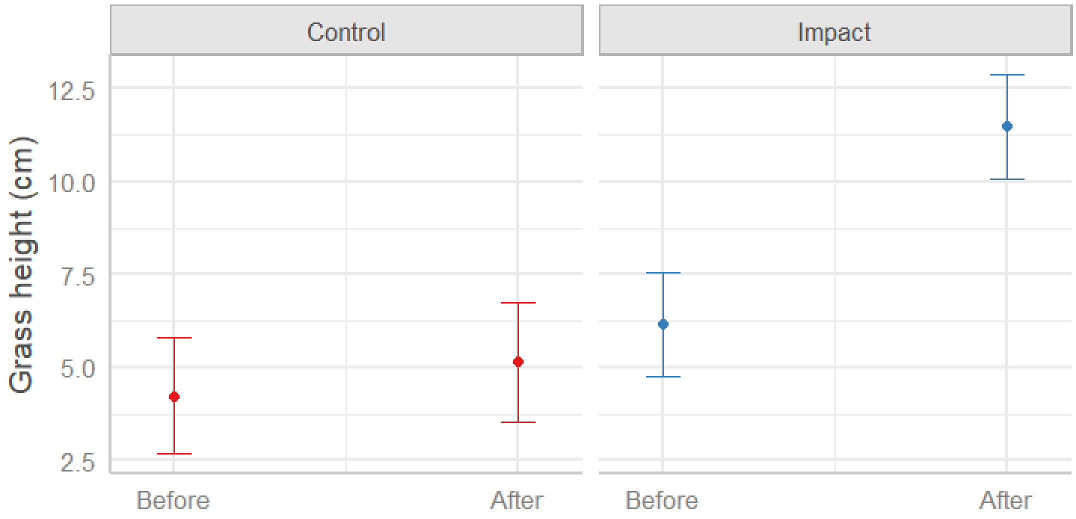

Figure 3. Variation in grass height over the study period for both control and impact paddocks. The mean values of grass height are represented by the points, while the solid lines depict the corresponding confidence intervals.

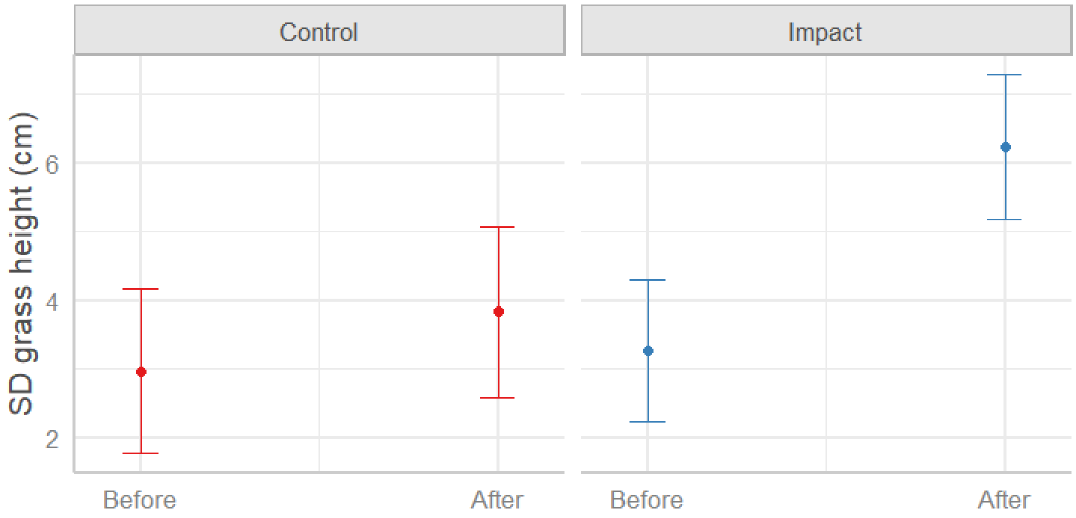

Figure 4. Standard deviation of grass height, representing patchiness, at control and impact paddocks. The mean values of the standard deviation are indicated by the points, while the solid lines represent the corresponding confidence intervals for both treatments.

According to our records, a total of 11 obligate grassland bird species were recorded during the study (Table 1). However, only 7 of these species had a sufficient number of records to model differences in abundance (Table 1). The Anthus genus and Xanthopsar flavus had an insufficient number of records to estimate their abundances accurately.

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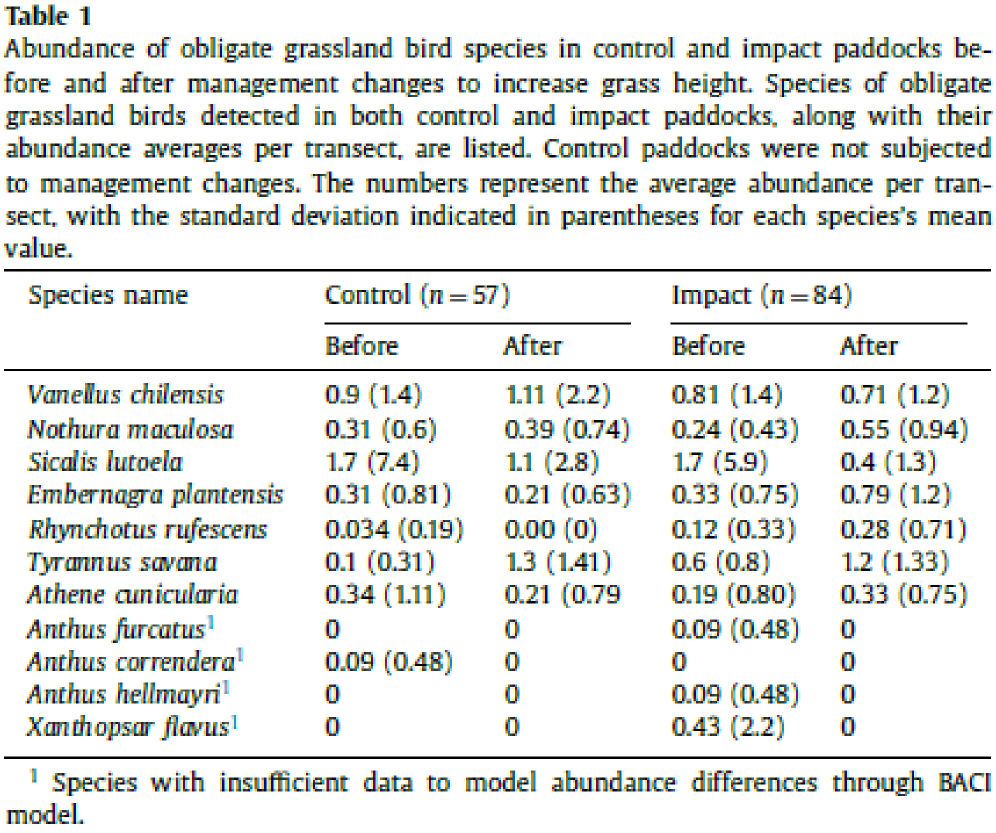

Table 1
Abundance of obligate grassland bird species in control and impact paddocks before and after management changes to increase grass height. Species of obligate grassland birds detected in both control and impact paddocks, along with their abundance averages per transect, are listed. Control paddocks were not subjected to management changes. The numbers represent the average abundance per transect, with the standard deviation indicated in parentheses for each species's mean value.

Species name	Control ($n = 57$)		Impact ($n = 84$)	
	Before	After	Before	After
Vanellus chilensis	0.9 (1.4)	1.11 (2.2)	0.81 (1.4)	0.71 (1.2)
Nothura maculosa	0.31 (0.6)	0.39 (0.74)	0.24 (0.43)	0.55 (0.94)
Sicalis lutoela	1.7 (7.4)	1.1 (2.8)	1.7 (5.9)	0.4 (1.3)
Embernagra plantensis	0.31 (0.81)	0.21 (0.63)	0.33 (0.75)	0.79 (1.2)
Rhynchotus rufescens	0.034 (0.19)	0.00 (0)	0.12 (0.33)	0.28 (0.71)
Tyrannus savana	0.1 (0.31)	1.3 (1.41)	0.6 (0.8)	1.2 (1.33)
Athene cunicularia	0.34 (1.11)	0.21 (0.79	0.19 (0.80)	0.33 (0.75)
Anthus furcatus[1]	0	0	0.09 (0.48)	0
Anthus correndera[1]	0.09 (0.48)	0	0	0
Anthus hellmayri[1]	0	0	0.09 (0.48)	0
Xanthopsar flavus[1]	0	0	0.43 (2.2)	0

[1] Species with insufficient data to model abundance differences through BACI model.

All 7 modeled species were recorded in both seasons, indicating their presence throughout the study period. Interestingly, no new species were observed following the change in management.

The interaction between SeasonBA (season) and PadTreat (treatment) did not have a significant effect on the abundances of any of the modeled species. This suggests that the management change did not result in noticeable differences in species abundances between the before and after periods in the impact paddocks (Fig. 5).

However, two species, *R. rufescens* and *E. platensis*, exhibited a slight increasing trend in their abundances in the impact paddocks, indicating a potential positive effect of the implemented management (Fig. 5, Table 1).

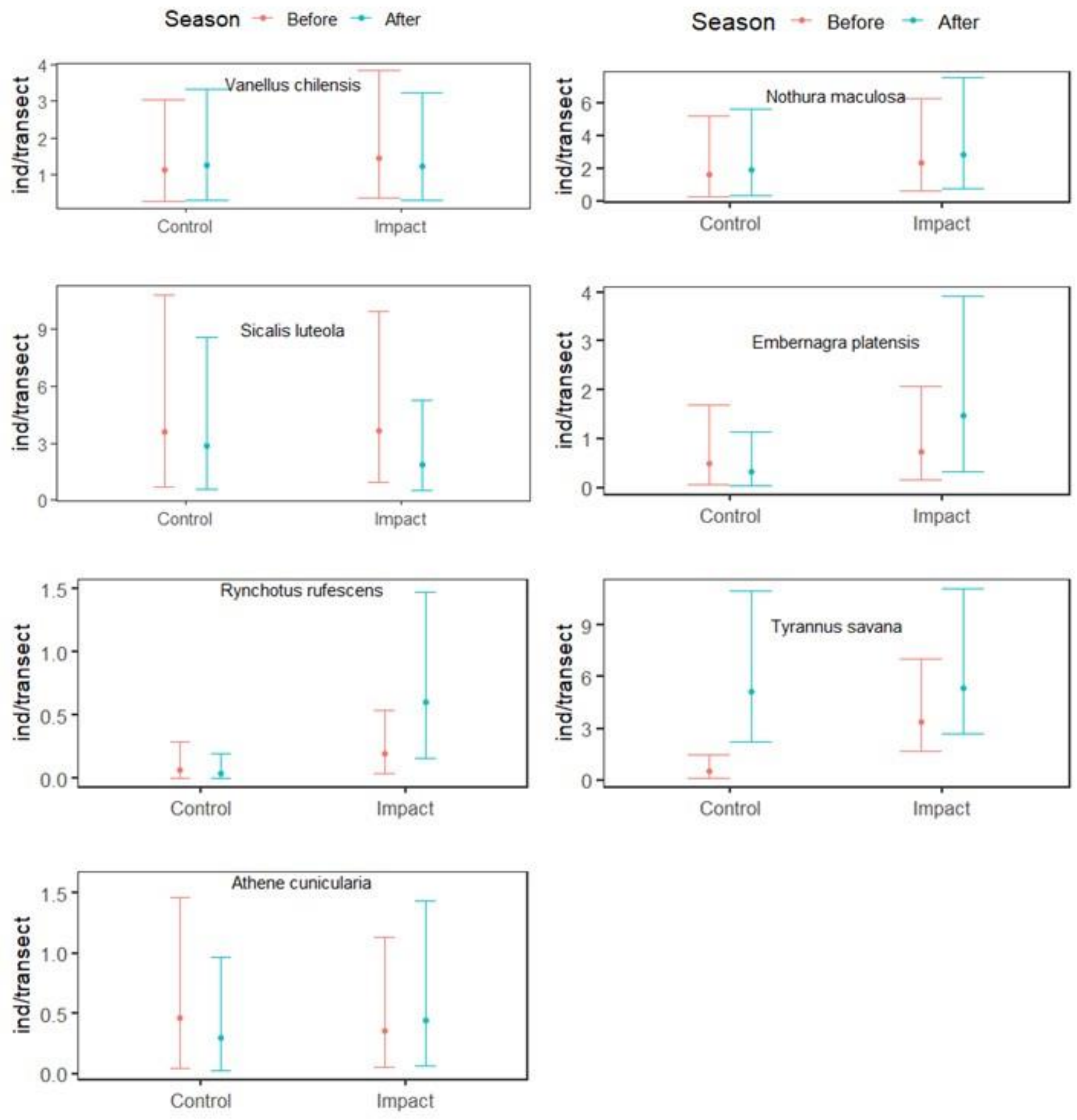

Figure 5. Predicted number of individuals per transect for each of the modeled species. The figure displays the predicted number of individuals per transect for each of the species modeled. The data is presented for both control paddocks, where no cattle management change was implemented, and impact paddocks, where grass height was increased through changes in cattle management. The "Before" category represents the time of bird counts conducted prior to the implementation of the cattle management changes, while the "After" category represents the time of bird counts conducted after the changes were made. The error bars represent the 95% credible intervals.

3.6. Discussion

Our findings demonstrate that it is indeed possible to increase grass height while maintaining grassland bird abundances stable. The average grass height resulting from our management practices aligns with levels known to maximize cattle dry matter intake rate, thus potentially enhancing livestock production (Da Trindade et al., 2017). These results suggest a positive synergy between increased grass height, the maintenance of seven grassland bird specialists abundances and livestock production. However, it is important to note that, coincident with Blumetto and Castagna (2019), the grass height achieved in our study was not sufficient to support the needs of other grassland specialists that require taller grasslands, such as the globally endangered *Sporophila cinnamomea*.

In our study, we did not record several species of grassland bird specialists, specifically those belonging to the *Sporophila*, *Emberizoides*, *Polystictus* and *Cistothorus* genera. Species in these genera are known to select grasslands with heights exceeding 50 cm (Azpiroz et al. 2012, Dias et al. 2014) and have been reported in the vicinity of our study area. However, the majority of our grass height measurements were well below 50 cm; this likely explains why we did not detect these species in our field data. Therefore, while the management changes we implemented may increase livestock production, they were insufficient to meet the habitat requirements of grassland bird species that rely on taller grasslands.

Despite inadequate grass height for endangered tall grass species in our study, the persistence over time of these management changes suggest a long-term increase in grass height compared to pre-study conditions. Therefore, in the long-term the applied grazing management may promote taller grass and habitat for tall grass birds. However, this may take several years. To address this limitation and fasten the grass height growing process, complementary actions could be undertaken, such as implementing temporary grazing exclusion in specific areas of the ranches (Cozzani and Zalba 2009, Blumetto and Castagna 2019). By temporarily excluding grazing from targeted areas, the grass height in those areas can be allowed to reach the desired levels for the specialized bird species. *Saccharum angustifolium* and *Paspalum quadrifarium* are widespread tussock grass species in this region (Lezama et al. 2019; López-Mársico 2021) that could provide the required vegetation height for birds. This targeted approach can provide suitable habitats for these species while still allowing for productive grazing as some tussock cover of 50 cm or more in a paddock has been reported to not affect livestock production (Ruggia 2021). Furthermore, grazing exclusions increase shrub cover (Lezama et al. 2014), enhancing structural heterogeneity and thus biodiversity (Benton et al. 2003).

Associated with the increase in grass height, we also observed an increase in grass height heterogeneity, which is consistent with findings from previous studies (Da Trindade et al. 2016, Ruggia et al. 2021). The importance of grass height heterogeneity for the conservation of grassland birds has been emphasized in numerous studies in the Rio de la Plata Grasslands (Isacch and Cardoni 2011, Codesido et al. 2013, Azpiroz and Blake 2016, Dias et al. 2017). However, the practical implementation of measures to generate this heterogeneity in rangelands has received limited attention (Pírez and Aldabe 2022). Here, we found that adjusting the stocking rate and breeding strategy contributed to the promotion of grass height heterogeneity. This could be attributed to the fact that with taller grass and increased forage availability, cattle tend to selectively graze on the more palatable plants, leaving behind the less digestible, and thus less selected, plants, creating heterogeneity in grass height (Do Carmo et al., 2018; Tonn et al., 2019). However, although grass height heterogeneity significantly increased after the management modifications, we did not observe changes in the abundance of grassland bird specialists. This lack of effect may be due to the relatively short time interval between the implementation of the management changes and the evaluation of the response of grassland bird specialists. It is possible that the expression of grass height heterogeneity in a magnitude sufficient to impact bird abundance requires a longer period of time.

In this sense, tussocks, which provide habitat heterogeneity for grassland birds (Comparatore et al. 1996, Pretelli et al. 2018), are expected to increase with decreasing grazing intensity as they are less selected by cattle (Ruggia et al. 2021). However, the limited time frame of our study may not have allowed enough time for these plants to sufficiently grow and contribute to increased heterogeneity. Future studies should consider evaluating the effects of grass height heterogeneity and bird responses over longer periods to better understand the dynamics and time frame required for these relationships to fully manifest.

Grass height is known to be indicative of grazing intensity, with shorter grass heights associated with higher grazing intensities and taller grass heights reflecting lower or no grazing intensity (Da Trindade et al. 2016). Previous review research on the effects of grazing intensities on birds by Barzan et al. (2021), found that high grazing intensities negatively impacted bird abundances, while moderate or low intensities had no effect. In contrast to these findings, our study did not detect any difference in bird abundances between short and taller grass heights when we changed management from higher (i.e., short grass) to lower grazing intensity (i.e., taller grass). This contrast might be explained by the fact that most studies reviewed by (Barzan et al. (2021) have used qualitative descriptors of grass height to classify grazing intensity into categories (e.g., low-profile vegetation to define high grazing intensity; Dias et al., 2017). However, the specific grazing intensity level associated with these descriptors may vary between studies, making it difficult to compare grazing intensity across different research efforts. Therefore, we suggest that future biodiversity studies employ objective and quantitative measures of grazing intensity.

One possible tool for quantifying grazing intensity is forage allowance, which refers to the relationship between forage dry matter per unit area and the number of animal units (or total body weight) at a specific point in time (Do Carmo et al., 2018b). Forage allowance offers a more precise and quantifiable parameter for assessing grazing intensity and can also be a valuable predictor of cattle performance. The consideration of forage allowance in biodiversity studies can provide a holistic approach that incorporates both food production and biodiversity conservation objectives, leading to more comprehensive and informative research outcomes.

The management approach we implemented, which resulted in increased grass height, may have positive implications for the nesting success of recorded grassland specialist birds. Previous studies have demonstrated that taller grass height provides better concealment and protection for bird nests, thereby enhancing reproductive success (Fondell and Ball 2004, Winter et al. 2005, Fogarty et al. 2017). Conversely, lower vegetation cover can increase nest visibility and predation rates (Zalba and Cozzani 2004). Specifically, species such as *Nothura maculosa, Embernagra plantensis, and Sicalis lutoela* would likely benefit from taller grasslands in terms of nesting success. This further suggests that there could be an increase in the abundance of these species over longer time periods, considering the favorable conditions for nesting provided by taller grasslands. Therefore, it would be valuable for future studies to evaluate whether our management practices promote nesting success.

Implications

Previous research in the field has predominantly focused on comparing intensive grazing with no grazing, leaving a gap in understanding the effects of intermediate grazing intensities. Our study addresses this knowledge gap by examining faunal responses under intermediate levels of grazing, which present a promising opportunity to achieve both production and conservation objectives. By exploring these intermediate situations, we contribute to a better understanding of the impacts of grazing intensity on biodiversity.

Our findings align with a win-win production-conservation approach, demonstrating that by strategically managing grass height and grazing intensity, it is possible to favor both livestock productivity and biodiversity conservation. Moreover, raising the height of grass may yield a favorable influence on the diversity of grass species (Altesor et al., 2005), the accumulation of soil organic carbon (Zhang et al., 2018), and diminished soil erosion rates (Modernel et al., 2016). This integrated approach holds great potential for protecting biodiversity on larger geographical scales while supporting sustainable agricultural practices. Our study adds valuable insights to the growing body of research promoting the coexistence of productive and conservation-oriented land management strategies.

*Nota adicional posterior a la publicación del artículo: el número de lecturas de altura en las transectas de 300 metros de largo puede ser insuficiente para entender la relación de la estructura del forraje del campo natural con la diversidad de aves del campo natural (Do Carmo 2020).

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3.7. References

Agra, M., Bilenca, D., Codesido, M., 2015. Responses of birds to planting of Lotus tenuis pasture in the Flooding Pampas, Argentina. Emu 115. https://doi.org/10.1071/MU14075

Aguerre, V., Albicette, M.M., 2018. Co-innovando para el desarrollo sostenible de sistemas ganaderos familiares de Rocha-Uruguay, Serie Técn. ed. INIA, Montevideo.

Alkemade, R., Reid, R.S., Van Den Berg, M., De Leeuw, J., Jeuken, M., 2013. Assessing the impacts of livestock production on biodiversity in rangeland ecosystems. Proc Natl Acad Sci U S A 110, 20900–20905. https://doi.org/10.1073/pnas.1011013108

Altesor, A., Oesterheld, M., Leoni, E., Lezama, F., Rodrı, C., 2005. Effect of grazing on community structure and productivity of a Uruguayan grassland 83–91. https://doi.org/10.1007/s11258-004-5800-5

Arballo, E., Cravino, J., 1999. Aves del Uruguay. Manual Ornitológico. Hemisferio Sur, Montevideo.

Azpiroz, A.B., Blake, J.G., 2016. Associations of grassland birds with vegetation structure in the Northern Campos of Uruguay. Condor 118, 12–23. https://doi.org/10.1650/CONDOR-15-49.1

Azpiroz, A.B., Isacch, J.P., Dias, R.A., Di Giacomo, A.S., Fontana, C.S., Palarea, C.M., 2012. Ecology and conservation of grassland birds in southeastern South America: A review. J Field Ornithol 83, 217–246. https://doi.org/10.1111/j.1557-9263.2012.00372.x

Azpiroz, A.B., 2003. Primeros registros del capuchino de collar (Sporophila zelichi) en Uruguay. Ornitol Neotrop 14, 117–119.

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, 1–22. https://doi.org/10.3390/rs12030381

Barzan, F.R., Bellis, L.M., Dardanelli, S., 2021. Livestock grazing constrains bird abundance and species richness: A global meta-analysis. Basic Appl Ecol 56, 289–298. https://doi.org/10.1016/j.baae.2021.08.007

Bates, D., Mächler, M., Bolker, B.M., Walker, S.C., 2015. Fitting linear mixed-effects models using lme4. J Stat Softw 67. https://doi.org/10.18637/jss.v067.i01

Benton, T.G., Vickery, J.A., Wilson, J.D., 2003. Farmland biodiversity: Is habitat heterogeneity the key? Trends Ecol Evol 18, 182–188. https://doi.org/10.1016/S0169-5347(03)00011-9

Blumetto, O., Castagna, A., 2019. Linking livestock production and wild biodiversity: Contribution of pastoral production systems to the habitat of bird priority conservation species, in: 2nd Euro-Mediterranean Conference for Environmental Integration. Sousse, Tunisia.

Claramunt, M., Fernández-Foren, A., Soca, P., 2018. Effect of herbage allowance on productive and reproductive responses of primiparous beef cows grazing on Campos grassland. Anim Prod Sci 58, 1615–1624. https://doi.org/10.1071/AN16601

Codesido, M., González-Fischer, C.M., Bilenca, D.N., 2013. Landbird Assemblages in Different Agricultural Landscapes: A Case Study in the Pampas of Central Argentina. Condor 115, 8–16. https://doi.org/10.1525/cond.2012.120011

Comparatore, V.M., Martinez, M.M., Vassallo, A.I., Barg, M., Isacch, J.P., 1996. Abundancia y Relaciones con el Hábitat de Aves Y Mamíferos En Pastizales de Paspalum quadrifarium (paja colorada) manejados con fuego (Provincia de Buenos Aires, Argentina). Interciencia 21, 228–237.

Cortés-Capano, G., Hanley, N., Sheremet, O., Hausmann, A., Toivonen, T., Garibotto-Carton, G., Soutullo, A., Di Minin, E., 2021. Assessing landowners' preferences to inform voluntary private land conservation: The role of non-monetary incentives. Land use policy 109, 105626. https://doi.org/10.1016/j.landusepol.2021.105626

Cozzani, N., Zalba, S., 2009. Estructura de la vegetación y selección de hábitats reproductivos en aves del pastizal pampeano. Ecologia Austral 19, 35–44.

Da Trindade, J.K.D., Neves, F.P., Pinto, C.E., Bremm, C., Mezzalira, J.C., Nadin, L.B., Genro, T.C.M., Gonda, H.L., Carvalho, P.C.F., 2016. Daily Forage Intake by Cattle on Natural Grassland: Response to Forage Allowance and Sward Structure. Rangel Ecol Manag 69, 59–67. https://doi.org/10.1016/j.rama.2015.10.002

Dias, R.A., Bastazini, V.A.G., Gianuca, A.T., 2014. Bird-habitat associations in coastal rangelands of southern Brazil. Iheringia Ser Zool 104, 200–208. https://doi.org/10.1590/1678-476620141042200208

Dias, R.A., Gianuca, A.T., Vizentin-Bugoni, J., Gonçalves, M.S.S., Bencke, G.A., Bastazini, V.A.G., 2017. Livestock disturbance in Brazilian grasslands influences avian species diversity via turnover. Biodivers Conserv 26, 2473–2490. https://doi.org/10.1007/s10531-017-1370-4

Do Carmo, Martín, Cardozo, G., Jaurena, M., Soca, P., 2018a. Demonstrating control of forage allowance for beef cattle grazing Campos grassland in Uruguay to improve system productivity. Tropical Grasslands 7, 35–47. https://doi.org/10.17138/TGFT(7)35-47

Do Carmo, M, Cardozo, G.A., Ruggia, A., Soca, P.M., 2018. Prediction of herbage mass in Campos grasslands based on herbage height, in: 10th International Symposium on the Nutrition of Herbivores 2018 Herbivore Nutrition Supporting Sustainable Intensification and Agro-ecological Approaches.

Do Carmo, Martín, Sollenberger, L.E., Carriquiry, M., Soca, P., 2018b. Controlling herbage allowance and selection of cow genotype improve cow-calf productivity in Campos grasslands. Prof Anim Sci 34, 32–41. https://doi.org/10.15232/pas.2016-01600

Ellis, E.C., 2019. To Conserve Nature in the Anthropocene, Half Earth Is Not Nearly Enough. One Earth 1, 163–167. https://doi.org/10.1016/j.oneear.2019.10.009

Fogarty, D.T., Elmore, R.D., Fuhlendorf, S.D., Loss, S.R., 2017. Influence of olfactory and visual cover on nest site selection and nest success for grassland-nesting birds. Ecol Evol 7, 6247–6258. https://doi.org/10.1002/ece3.3195

Fondell, T.F., Ball, I.J., 2004. Density and success of bird nests relative to grazing on western Montana grasslands. Biol Conserv 117, 203–213. https://doi.org/10.1016/S0006-3207(03)00293-3

Franklin, J.F., Lindenmayer, D.B., 2009. Importance of matrix habitats in maintaining biological diversity.

Fuhlendorf, S.D., Fynn, R.W.S., McGranahan, D.A., Twidwell, D., 2017. Heterogeneity as the Basis for Rangeland Management, in: Briske, D.D. (Ed.), Rangeland Systems. Springer International Publishing, Cham, pp. 169–196. https://doi.org/10.1007/978-3-319-46709-2

Fuhlendorf, S.D., Harrell, W.C., Engle, D.M., Hamilton, R.G., Davis, C.A., Leslie Jr., D.M., 2006. Should heterogeneity be the basis fo conservation? Grassland bird response to fire and grazing. Ecological Applications 16, 1706–1716.

Gennet, S., Spotswood, E., Hammond, M., Bartolome, J.W., 2017. Livestock grazing supports native plants and songbirds in a California annual grassland. PLoS One 12. https://doi.org/10.1371/journal.pone.0176367

Green, R.H., 1979. Sampling design and statistical methods for environmental biologists. Wiley, New York; Chichester.

Gurney, G.G., Darling, E.S., Ahmadia, G.N., Agostini, V.N., Ban, N.C., Blythe, J., Claudet, J., Epstein, G., Himes-cornell, A., Jonas, H.D., Armitage, D., Campbell, S.J., Cox, C., Friedman, W.R., Gill, D., Lestari, P., Mangubhai, S., Mcleod, E., Muthiga, N.A., Ranaivoson, R., Wenger, A., Yulianto, I., Jupiter, S.D., 2021. Biodiversity needs every tool in the box : use OECMs. Nature 595, 646–649.

Hoekstra, J.M., Boucher, T.M., Ricketts, T.H., Roberts, C., 2005. Confronting a biome crisis: Global disparities of habitat loss and protection. Ecol Lett 8, 23–29. https://doi.org/10.1111/j.1461-0248.2004.00686.x

Isacch, J.P., Cardoni, D.A., 2011. Different Grazing Strategies Are Necessary to Conserve Endangered Grassland Birds in Short and Tall Salty Grasslands of the Flooding Pampas. Condor 113, 724–734. https://doi.org/10.1525/cond.2011.100123

Kellner, K.F., Fowler, N.L., Petroelje, T.R., Kautz, T.M., Beyer, D.E., Belant, J.L., 2022. ubms: An R package for fitting hierarchical occupancy and N-mixture abundance models in a Bayesian framework. Methods Ecol Evol 13, 577–584. https://doi.org/10.1111/2041-210X.13777

Lezama, F., Pereira, M., Altesor, A., Paruelo, J. M., 2019. Grasslands of Uruguay: Classification based on vegetation plots. Phytocoenologia, 49(3), 211–229. https://doi.org/10.1127/phyto/2019/0215

Lezama, F., Baeza, S., Altesor, A., Cesa, A., Chaneton, E.J., Paruelo, J.M., 2014. Variation of grazing-induced vegetation changes across a large-scale productivity gradient. Journal of Vegetation Science 25, 8–21. https://doi.org/10.1111/jvs.12053

López-Mársico, L., Lezama, F., Altesor, A., 2021. Heterogeneity decreases as time since fire increases in a South American grassland. Appl Veg Sci 24, e12521. https://doi.org/https://doi.org/10.1111/avsc.12521

MacKenzie, D.I., Nichols, J.D., Lachman, G.B., Droege, S., Royle, A.A., Langtimm, C.A., 2002. Estimating site occupancy rates when detection probabilities are less than one. Ecology 83, 2248–2255. https://doi.org/10.1890/0012

Mcdonald, T.L., Erickson, W.P., Mcdonald, L.L., 2000. Analysis of count data from before-after control-impact studies. J Agric Biol Environ Stat 5, 262–276. https://doi.org/10.2307/1400453

Modernel, P., Picasso, V., Do Carmo, M., Rossing, W.A.H., Corbeels, M., Soca, P., Dogliotti, S., Tittonell, P., 2019. Grazing management for more resilient mixed livestock farming systems on native grasslands of southern South America. Grass and Forage Science 74. https://doi.org/10.1111/gfs.12445

Modernel, P., Rossing, W.A.H., Corbeels, M., Dogliotti, S., Picasso, V., Tittonell, P., 2016. Land use change and ecosystem service provision in Pampas and Campos grasslands of southern South America. Environmental Research Letters 11. https://doi.org/10.1088/1748-9326/11/11/113002

Morris, C.D., Derry, J.F., Hardy, M.B., 1999. Effec of cattle and sheep grazing on the structure of Highland Souveld swards in South Africa. Tropical Grasslands 33, 111–121.

Neilly, H., Vanderwal, J., Schwarzkopf, L., 2016. Balancing Biodiversity and Food Production: A Better Understanding of Wildlife Response to Grazing Will Inform Off-Reserve Conservation on Rangelands. Rangel Ecol Manag 69, 430–436. https://doi.org/10.1016/j.rama.2016.07.007

Oldekop, J.A., Holmes, G., Harris, W.E., Evans, K.L., 2016. A global assessment of the social and conservation outcomes of protected areas. Conservation Biology 30, 133–141. https://doi.org/10.1111/cobi.12568

Oyarzabal, M., Andrade, B., Pillar, V.D., Paruelo, J., 2020. Temperate Subhumid Grasslands of Southern South America. Encyclopedia of the World's Biomes 577–593. https://doi.org/10.1016/b978-0-12-409548-9.12132-3

Panario, D., Gutiérrez, O., Sánchez-Bettuchhi, L., Peel, E., Oyhantçabal, P., Rabassa, J., 2014. Ancient Landscapes of Uruguay, in: Rabassa, J., Ollier, C. (Eds.), Gondwana Landscapes in South America. Springer, Ushuaia, pp. 161–199.

Pírez, F., Aldabe, J., 2022. Comparison of the bird community in livestock farms with continuous and rotational grazing in eastern Uruguay. Ornithology Research. https://doi.org/10.1007/s43388-022-00113-1

Pretelli, M.G., Isacch, J.P., Cardoni, D.A., 2018. Species-Area Relationships of Specialist Versus Opportunistic Pampas Grassland Birds Depend on the Surrounding Landscape Matrix. Ardeola 65, 3–23. https://doi.org/10.13157/arla.65.1.2018.ra1

Royle, J.A., 2004. N-Mixture Models for Estimating Population Size from Spatially Replicated Counts. Biometrics 60, 108–115.

Ruggia, A., Dogliotti, S., Aguerre, V., Albicette, M.M., Albin, A., Blumetto, O., Cardozo, G., Leoni, C., Quintans, G., Scarlato, S., Tittonell, P., Rossing, W.A.H., 2021. The application of ecologically intensive principles to the systemic redesign of livestock farms on native grasslands: A case of co-innovation in Rocha, Uruguay. Agric Syst 191, 103148. https://doi.org/10.1016/j.agsy.2021.103148

Soriano, A., León, R., Sala, O., Lavado, R., Deregibus, V., Cauhepé, M., Scaglia, O., Velázquez, C., Lemcoff, J., 1992. Rio de la Plata Grasslands, in: Coupland, R. (Ed.), Ecosystems of the World 8A. Natural Grasslands: Introduction and Western Hemisphere. Elsevier, Amsterdam, pp. 367–407.

Tonn, B., Raab, C., Isselstein, J., 2019. Sward patterns created by patch grazing are stable over more than a decade. Grass and Forage Science 74, 104–114. https://doi.org/10.1111/gfs.12389

Vickery, P.D., Tubaro, P.L., Da Silva, J.M.C., Peterjohn, B.G., Herkert, J.R., Cavalcanti, R.B., 1999. Conservation of grassland birds in the western hemisphere. Studies in Avian Biology 2–26.

Winter, M., Johnson, D.H., Shaffer, J.A., 2005. Variability in Vegetation Effects on Density and Nesting Success of Grassland Birds. Journal of Wildlife Management 69, 185–197. https://doi.org/10.2193/0022-541x(2005)069<0185:viveod>2.0.co;2

Zalba, S.M., Cozzani, N.C., 2004. The impact of feral horses on grassland bird communities in Argentina. Anim Conserv 7, 35–44. https://doi.org/10.1017/S1367943003001094

Zhang, M., Li, X., Wang, H., Huang, Q., 2018. Comprehensive analysis of grazing intensity impacts soil organic carbon: A case study in typical steppe of Inner Mongolia, China. Applied Soil Ecology 129, 1–12. https://doi.org/https://doi.org/10.1016/j.apsoil.2018.03.008

4. Discusión general y conclusiones

En este trabajo estimamos la respuesta de 69 especies de aves de pastizal a la altura del pasto, la presencia de matas cespitosas (*e. g.*, pajonales), la cobertura de árboles y la implantación de praderas artificiales. Estas variables están relacionadas con decisiones de manejo por parte de los productores; por lo tanto, la información generada puede servir para el diseño de sistemas productivo que integren objetivos de conservación de aves silvestres (capitulo 2). El número de especies evaluadas representa más del 50 % de las especies de aves que utilizan los pastizales en la región del Río de la Plata (Azpiroz et al., 2012; Pírez y Aldabe, 2022). Por lo tanto, el presente trabajo constituye un aporte muy relevante al conocimiento de la ecología de aves de la región, con implicancias directas para el diseño de sistemas de pastoreo que favorezcan su conservación. Además, generamos información sobre cómo las especies responden según sus rasgos morfológicos y comportamentales, una dimensión muy poco explorada en las aves de pastizal, que permite generar inferencias de las respuestas de aves con rasgos similares en otras regiones del mundo. Por último, este trabajo (capítulo 3) mostró que es posible incrementar la altura del pasto y su heterogeneidad estructural, y potencialmente aumentar la productividad ganadera, manteniendo constante la abundancia de aves especialistas de pastizal. En línea con Jaurena et al. (2021), esto sugiere que existe margen de intensificación en los sistemas pastoriles sobre campo natural, sin afectación negativa sobre aves que dependen estrictamente del pastizal. No obstante, varias especies de aves que son de prioridad para la conservación en el mundo no colonizaron los predios en los que se aumentó la altura del pasto, posiblemente porque la magnitud del cambio no fue suficiente. Lapsos más prolongados con el manejo aplicado permitirían una mayor magnitud de cambio.

Si bien los cambios en la altura del pasto fueron notorios en los predios en los que se realizaron cambios de manejo (capítulo 3), las especies de aves no cambiaron su abundancia. Estos resultados empíricos en general no coincidieron con las predicciones teóricas sobre la respuesta de esas aves a cambios de altura del pasto (capítulo 2). El modelo ajustado en el capítulo 2 predijo que todas las especies, excepto la tijereta (*Tyrannus savana)* y la martineta (*Rynchotus rufescens*), responden sustancialmente a cambios en la altura del pasto. En este modelo, especies como el tero (*Vanellus chilensis*) y la lechucita de campo (*Athene cunicularia*) mostraron una respuesta negativa a incrementos en la altura del pasto, mientras que el verdón (*Embernagra platensis*), el misto (*Sicalis luteola*) y la perdiz (*Nothura maculosa*) mostraron una respuesta positiva a los incrementos en la altura del pasto. Sin embargo, cuando se realizaron cambios de manejo para aumentar la altura del pasto en 6 predios (capítulo 3), no se detectaron diferencias en la abundancia respecto a la situación con pasto más corto. Esta falta de respuesta puede deberse a que la magnitud de cambio de altura del pasto (relacionada con la corta duración del estudio) no fue suficiente para que las aves mostraran cambios en su abundancia. Además, tiempos más prolongados podrían permitir un incremento de abundancia por nacimientos y colonización.

El modelo predictivo (capítulo 2) mostró que varias especies respondieron a la altura del pasto: algunas respondieron positivamente a incrementos de la altura del pasto, mientras que otras lo hicieron de manera negativa o no mostraron una respuesta. Identificamos que las especies que utilizan el suelo tendieron a preferir pastos más cortos (*e. g.*, el tero *Vanellus chilensis,* la calandria *Mimus saturninus*). Estas aves se alimentan de artrópodos y semillas que detectan visualmente en el suelo. Similar a otras aves de pastizales en otras regiones, el pasto corto facilita esta detección (Butler y Gillings, 2004). No obstante, hubo especies que también usan el suelo de manera frecuente, pero no respondieron positivamente a pastos cortos. Un ejemplo es la perdiz *Nothura maculosa*. Si bien esta especie se alimenta en el suelo a partir de semillas y artrópodos que detecta con la visión, necesita pastos más altos para refugiarse, dado que su estrategia de escape contra depredadores se basa en pasar desapercibida y volar repentinamente cuando el depredador está muy próximo al ave. Esto genera una respuesta de *freezing* en el depredador, que queda paralizado por un lapso que la perdiz aprovecha para alejarse y escapar. Entonces, nuestros resultados sugieren que las respuestas de las diferentes especies a la altura del pasto pueden estar relacionadas con la estrategia de forrajeo y con la estrategia antidepredador, o son el resultado de un *trade-off* entre ambos rasgos. Sin embargo, es importante señalar que para 60 % (aproximadamente) de las especies no detectamos una respuesta a la altura del pasto, siendo la gran mayoría de estas especies facultativas pastizal. Entonces, si bien la altura del pasto ha sido ampliamente considerada en la literatura como la principal variable ecológica para explicar la abundancia y distribución de las aves de pastizal (Fisher y Davis, 2010), esta variable no afecta a todas las especies. Como se mostró aquí (capítulo 2), otras variables como la presencia de pajonales y árboles pueden contribuir a explicar los patrones de ocurrencia.

Por otro lado, la presencia de plantas cespitosas mostró un efecto en general positivo sobre numerosas especies. Sin embargo, la baja calidad forrajera de los pajonales impone una restricción para su conservación, dado que los productores tienden a cortarlos o quemarlos (Herrera et al., 2009). Sin embargo, varios trabajos reportan que un pastoreo continuo con intensidades moderadas da como resultado rendimientos productivos adecuados en potreros con hasta un 30 % de cobertura de matas de pajonal (Da Trindade et al., 2016). En este sentido, sería recomendable proveer asistencia técnica a los productores ganaderos para informarles sobre cómo proteger los pajonales y al mismo tiempo lograr un rendimiento adecuado de la producción ganadera (Ruggia et al., 2021). Además, los pajonales pueden ofrecer servicios valiosos para la producción ovina con pariciones al aire libre. Uno de los problemas más acuciantes en este rubro es la mortandad de corderos debido a tormentas, bajas temperaturas y vientos fuertes (Abella et al., 2017). Los pajonales podrían auspiciar de abrigo para estos animales y evitar importantes mortandades (Pereira-Machín, comunicación personal, 2022).

Otra de las variables que mostró tener incidencia sobre varias especies fue la cobertura de árboles. Varias especies se beneficiaron, al tiempo que otras se perjudicaron. Las especies que se perjudicaron fueron, en general, especialistas de pastizal, mientras que las especies que respondieron positivamente fueron generalistas como el hornero *Furnarius Rufus* y la calandria *Mimus saturninus*. La cobertura de árboles también está influenciada por el manejo del pastoreo, con coberturas arbóreas más bajas en áreas con mayores presiones de pastoreo (Bernardi et al., 2019). Entonces, a través del manejo del pastoreo y su efecto sobre la cobertura arbórea, es posible incidir sobre el ensamble de aves. Si el objetivo es incrementar la riqueza de especies, entonces la presencia de coberturas arbóreas va a ser deseable. Además, estas coberturas son de gran valor para dar abrigo y sombra al ganado. No obstante, para favorecer especies de aves especialistas de pastizales y especies amenazadas como el dragón *Xanthopsar flavus* y la viudita blanca grande *Xolmis dominicanus*, es necesario generar pastizales abiertos con baja cobertura arbórea. Dependiendo del contexto social y biológico, se podrá optar por alguna de estas alternativas.

En línea con otros trabajos, las praderas artificiales mostraron un efecto global negativo sobre las aves (Agra et al., 2015; Fontana et al., 2016). Sin embargo, las praderas son muchas veces clave para los sistemas productivos (Jaurena et al., 2021). De hecho, si se destina una proporción relativamente pequeña del predio a la implantación de praderas, se puede beneficiar la conservación del campo natural y sus servicios ecosistémicos, dado que estas áreas ofrecen reservas de forraje en períodos de baja productividad de pasto, que permiten hacer ajustes de carga para evitar el sobrepastoreo del campo natural (Jaurena et al., 2021). Por lo tanto, si bien las praderas mostraron un impacto global negativo sobre las aves, pequeñas áreas pueden favorecer indirectamente la biodiversidad en los predios.

4.1. Conclusiones y perspectivas de investigación

Finalmente, es importante destacar que se puede trabajar con más pasto y mejorar la producción sin afectar la abundancia de especies de aves especialistas. Este resultado complementa la propuesta de intensificación ecológica (Ruggia et al., 2021) o intensificación sostenible desarrollada por Jaurena et al. (2021). En esta propuesta se argumenta que mediante modificaciones en el manejo del pastoreo se puede incrementar la rentabilidad con base en campo natural, conservando los servicios ecosistémicos como el control de la erosión, el almacenamiento orgánico en el suelo y el sostén de la biodiversidad. Aquí aportamos nueva información que sugiere que estas modificaciones de manejo son compatibles con la conservación de 7 especies de aves que son especialistas de pastizal. Además, trabajar con más pasto permite que las vacas seleccionen las pasturas más palatables, lo que permite disminuir presión de pastoreo sobre pajonales que beneficiarán a la avifauna, incluyendo especies amenazadas. Además, manejos del pastoreo que incrementen la altura del estrato herbáceo bajo pueden favorecer el éxito reproductivo de las aves, ya que los nidos quedan más ocultos y menos accesibles a los depredadores (Fondell y Ball, 2004). Esto se debería evaluar *in situ,* así como dar un seguimiento al desarrollo de altura de los pajonales a lo largo del tiempo.

5. Bibliografía

Abella, F., M. Cueto, y F. Moraes. (2017). Factores que afectan la supervivencia del cordero. *Revista Argentina de Producción Animal* 17:61-76.

Agra, M., D. Bilenca, y M. Codesido. (2015). Responses of birds to planting of Lotus tenuis pasture in the Flooding Pampas, Argentina. *Emu* 115:270-276

Albicette, M. M., C. Leoni, A. Ruggia, S. Scarlato, O. Blumetto, A. Albín, y V. Aguerre. (2017). Co-innovation in family-farming livestock systems in Rocha, Uruguay: A 3-year learning process. *Outlook on Agriculture* 46:92-98.

Aldabe, J., R. B. Lanctot, D. Blanco, P. Rocca, y P. Inchausti. (2019). Managing Grasslands to Maximize Migratory Shorebird Use and Livestock Production. *Rangeland Ecology and Management* 72:150-159.

Altesor, A., M. Oesterheld, E. Leoni, F. Lezama, y C. Rodríguez. (2005). Effect of grazing on community structure and productivity of a Uruguayan grassland. *Plant Ecology* 179:83-91.

Azpiroz, A. B., y J. G. Blake. (2016). Associations of grassland birds with vegetation structure in the Northern Campos of Uruguay. *The Condor* 118:12-23.

Azpiroz, A. B., J. P. Isacch, R. A. Dias, A. S. Di Giacomo, C. S. Fontana, y C. M. Palarea. (2012). Ecology and conservation of grassland birds in southeastern South America: A review. *Journal of Field Ornithology* 83:217-246.

Baeza, S., y J. M. Paruelo. (2020). Land use/land cover change (2000-2014) in the Rio de la Plata grasslands: An analysis based on MODIS NDVI time series. *Remote Sensing* 12:1–22.

Baeza, S., Vélez-Martin, E., de Abelleyra, D., Banchero, S., Gallego, F., Schirmbeck, J., Veron, S., Vallejos, M., Weber, E., Oyarzabal, M., Barbieri, A., Petek, M., Guerra Lara, M., Sarrailhé, S. S., Baldi, G., Bagnato, C., Bruzzone, L., Ramos, S., y Hasenack, H. (2022). Two decades of land cover mapping in the Río de la Plata grassland region: The Map Biomas Pampa initiative. *Remote Sensing Applications: Society and Environment*, 28, 100834.

Barzan, F. R., L. M. Bellis, y S. Dardanelli. (2021). Livestock grazing constrains bird abundance and species richness: A global meta-analysis. *Basic and Applied Ecology* 56:289-298.

Bernardi, R. E., M. Buddeberg, M. Arim, y M. Holmgren. (2019). Forests expand as livestock pressure declines in subtropical South America. *Ecology and Society* 2:19.

Berreta, E. J., D. F. Risso, F. Montossi, y G. Pigurina. (2000). *Campos in Uruguay*. Page 422. En G. Lemair, A. de Moraes, C. Nabinger, y P. C. F. Carvalho (eds.). Grassland Ecophisiology and Grazing Ecology. CABI Publishing, New York.

Bilenca, D. y Miñarro, F. (2004). *Identificación de Áreas Valiosas de Pastizal (AVPs) en las Pampas y Campos de Argentina, Uruguay y sur de Brasil*. Fundación Vida Silvestre Argentina, Buenos Aires.

Blumetto, O., y A. Castagna. (2019). Linking livestock production and wild biodiversity: Contribution of pastoral production systems to the habitat of bird priority conservation species. *2nd Euro-Mediterranean Conference for Environmental Integration*. Sousse, Tunisia.

Butler, S. J., M. J. Whittingham, J. L. Quinn, y W. Cresswell. (2005). Quantifying the interaction between food density and habitat structure in determining patch selection. *Animal Behaviour* 69:337-343.

Butler, S.J. y Gillings, S. (2004). Quantifying the effects of habitat structure on prey detectability and accessibility to farmland birds. *Ibis* 146:123-130.

Cardoni, D. A., J. P. Isacch, y O. Iribarne. (2015). Avian responses to varying intensity of cattle production in Spartina densiflora saltmarshes of South-Eastern South America. *Emu* 115(1):12-19.

Cartes, J. L., y A. A. Yanosky. (2020). Evaluation of the Paraguayan System of Protected Areas After 24 Years of Its Implementation. Revista de Ciencias Ambientales 54:147-164.

Carvalho, P. (2013). Harry Stobbs Memorial Lecture: Can grazing behavior support innovations in grassland management? *Tropical Grasslands-Forrajes Tropicales*, 1, 137–155.

Carvalho, P. C. F., M. O. Wallau, C. Bremm, O. Bonnet, J. K. Da Trindade, F. Q. Rosa, T. S. Freitas, y F. G. Moojen. (2019). *Boletim Nativão: + de 30 anos de pesquisa em campo nativo*. P. C. F. Carvalho, M. O. Wallau, C. Bremm, O. Bonnet, J. K. Da Trindade, F. Q. Rosa, T. S. Freitas, y F. G. Moojen (eds.). Editora Viapampa, Uruguaiana.

Claramunt, M., A. Fernández-Foren, y P. Soca. (2018). Effect of herbage allowance on productive and reproductive responses of primiparous beef cows grazing on Campos grassland. *Animal Production Science* 58:1615-1624.

Da Silva, T. W., G. Dotta, D. T. Gressler, y C. S. Fontana. (2015). Habitat use by grassland birds in natural areas and soybean fields in southern Brazil and Uruguay. *The Wilson Journal of Ornithology* 127:212-221.

Da Trindade, J. K. D., F. P. Neves, C. E. Pinto, C. Bremm, J. C. Mezzalira, L. B. Nadin, T. C. M. Genro, H. L. Gonda, y P. C. F. Carvalho. (2016). Daily Forage Intake by Cattle on Natural Grassland: Response to Forage Allowance and Sward Structure. *Rangeland Ecology and Management* 69:59-67.

Davis, K. P., D. J. Augustine, A. P. Monroe, J. D. Derner, y C. L. Aldridge. (2020). Adaptive rangeland management benefits grassland birds utilizing opposing vegetation structure in the shortgrass steppe. *Ecological Applications* 30:1-14.

Derner, J. D., W. K. Lauenroth, P. Stapp, y D. J. Augustine. (2009). Livestock as ecosystem engineers for Grassland Bird habitat in the Western Great Plains of North America. *Rangeland Ecology and Management* 62:111-118.

Dias, R. A., A. T. Gianuca, J. Vizentin-Bugoni, M. S. S. Gonçalves, G. A. Bencke, y V. A. G. Bastazini. (2017). Livestock disturbance in Brazilian grasslands influences avian species diversity via turnover. *Biodiversity and Conservation* 26:2473-2490.

Dias, R. A., V. A. G. Bastazini, y A. T. Gianuca. (2014). Bird-habitat associations in coastal rangelands of southern Brazil. *Iheringia. Série Zoologia* 104:200-208.

Dixon, A. P., D. Faber-Langendoen, C. Josse, J. Morrison, y C. J. Loucks. (2014). Distribution mapping of world grassland types. *Journal of Biogeography* 41:2003-2019.

Do Carmo, M., G. Cardozo, M. Jaurena, y P. Soca. (2018). Demonstrating control of forage allowance for beef cattle grazing Campos grassland in Uruguay to improve system productivity. *Tropical Grasslands* 7:35-47.

Do Carmo, M., M. Claramunt, M. Carriquiry, y P. Soca. (2016). Animal energetics in extensive grazing systems: Rationality and results of research models to improve energy efficiency of beef cow-calf grazing Campos systems. *Journal of Animal Science* 94:84-92.

Douglas, D. J. T., J. Waldinger, Z. Buckmire, K. Gibb, J. P. Medina, L. Sutcliffe, C. Beckmann, N. J. Collar, R. Jansen, J. Kamp, I. Little, R. Sheldon, A. Yanosky, y N. Koper. (2023). A global review identifies agriculture as the main threat to declining grassland birds. *Ibis* 165:1107-1128

Fedrigo, J. K., P. F. Ataide, J. A. Filho, L. V. Oliveira, M. Jaurena, E. A. Laca, G. E. Overbeck, y C. Nabinger. (2018). Temporary grazing exclusion promotes rapid recovery of species richness and productivity in a long-term overgrazed Campos grassland. *Restoration Ecology* 26:677-685.

Fisher, R. J., y S. K. Davis. (2010). From Wiens to Robel: A Review of Grassland-Bird Habitat Selection. *Journal of Wildlife Management* 74:265-273.

Fondell, T. F., y I. J. Ball. (2004). Density and success of bird nests relative to grazing on western Montana grasslands. *Biological Conservation* 117:203-213.

Fontana, C. S., G. Dotta, C. K. Marques, M. Repenning, C. E. Agne, y R. J. dos Santos. (2016). Conservation of grassland birds in South Brazil: a land management perspective. *Natureza e Conservação* 14:83-87.

Fuhlendorf, S. D., y D. M. Engle. (2001). Restoring heterogeneity on rangelands: Ecosystem management based on evolutionary grazing patterns. *BioScience* 51:625-632.

Fuhlendorf, S. D., R. W. S. Fynn, D. A. McGranahan, y D. Twidwell. (2017). Heterogeneity as the Basis for Rangeland Management. Pages 169-196 en D. D. Briske (ed). *Rangeland Systems*. Springer International Publishing, Cham.

Gavier-Pizarro, G. I., N. C. Calamari, J. J. Thompson, S. B. Canavelli, L. M. Solari, J. Decarre, A. P. Goijman, R. P. Suarez, J. N. Bernardos, y M. E. Zaccagnini. (2012). Expansion and intensification of row crop agriculture in the Pampas and Espinal of Argentina can reduce ecosystem service provision by changing avian density. *Agriculture, Ecosystems and Environment* 154:44-55.

Gibson, D. J. (2009). *Grasses and Grasslands*. Oxford University Press, Oxford.

Goijman, A. P., M. J. Conroy, V. D. Varni, J. J. Thompson, y M. E. Zaccagnini. (2020). Occupancy of avian foraging guilds in soybean fields and borders in Entre Ríos, Argentina: responses to vegetation structure and prey resources. *Avian Research* 11:1-14.

Herrera, L. P., P. Laterra, N. O. Maceira, K. D. Zelaya, y G. A. Martínez. (2009). Fragmentation status of tall-tussock grassland relicts in the Flooding Pampa, Argentina. *Rangeland Ecology and Management* 62:73-82.

Hoekstra, J. M., T. M. Boucher, T. H. Ricketts, y C. Roberts. (2005). Confronting a biome crisis: Global disparities of habitat loss and protection. *Ecology Letters* 8:23-29.

Isacch, J. P., y D. A. Cardoni. (2011). Different Grazing Strategies Are Necessary to Conserve Endangered Grassland Birds in Short and Tall Salty Grasslands of the Flooding Pampas. *The Condor* 113:724-734.

Jaurena, M., M. Durante, T. Devincenzi, J. V. Savian, D. Bendersky, F. G. Moojen, M. Pereira, P. Soca, F. L. F. Quadros, R. Pizzio, C. Nabinger, P. C. F. Carvalho, y F. A. Lattanzi. (2021). Native Grasslands at the Core: A New Paradigm of Intensification for the Campos of Southern South America to Increase Economic and Environmental Sustainability. *Frontiers in Sustainable Food Systems* 5:1-15.

Knopf, F. L. (1996). Prairie Legacies-Birds. Pages 135–148 in F. B. Samson y F. Knopf (eds.). *Prairie conservation: preserving North America's most endangered ecosystem*. Island Press, Washington, D.C.

Modernel, P., S. Dogliotti, S. Alvarez, M. Corbeels, V. Picasso, P. Tittonell, y W. A. H. Rossing. (2018). Identification of beef production farms in the Pampas and Campos area that stand out in economic and environmental performance. *Ecological Indicators* 89:755-770.

Modernel, P., W. A. H. Rossing, M. Corbeels, S. Dogliotti, V. Picasso, y P. Tittonell. (2016). Land use change and ecosystem service provision in Pampas and

Campos grasslands of southern South America. *Environmental Research* Letters 11:1-21.

Mouquet, N., Y. Lagadeuc, V. Devictor, L. Doyen, A. Duputié, D. Eveillard, D. Faure, E. Garnier, O. Gimenez, P. Huneman, F. Jabot, P. Jarne, D. Joly, R. Julliard, S. Kéfi, G. J. Kergoat, S. Lavorel, L. Le Gall, L. Meslin, …, M. Loreau. (2015), October 1. *Predictive ecology in a changing world*. Blackwell Publishing Ltd.

Oyarzabal, M., B. Andrade, V. D. Pillar, y J. Paruelo. (2020). Temperate Subhumid Grasslands of Southern South America. *Encyclopedia of the World's Biomes*:577-593.

Pinto, C. E., M. O. Wallau, y I. Boldrini. (2019). Estrutura da Vegetaçao e Composiçao Florística. Pages 16–20 in P. C. F. Carvalho, M. O. Wallau, C. Bremm, O. Bonnet, J. K. Da Trindade, F. Q. Rosa, T. S. Freitas, y F. G. Moojen (eds.). *Boletim Nativao: + de 30 anos de pesquisa em campo nativo*. Uruguaiana.

Pírez, F., y J. Aldabe. (2022). Comparison of the bird community in livestock farms with continuous and rotational grazing in eastern Uruguay. *Ornithology Research* 31:41-50.

Quintans, G., A. I. Vázquez, y K. A. Weigel. (2009). Effect of suckling restriction with nose plates and premature weaning on postpartum anestrous interval in primiparous cows under range conditions. *Animal Reproduction Science* 116:10-18.

Rinaldi, C. (1997). Manejo de la recría de tambo en campo natural. *Revista Cangüé* 11:12-16.

Rosa, F. Q., C. Bremm, y D. R. Machado. (2019). Efeito da Oferta de Forragem na Estrutura do Pasto. Pages 23-25 in P. C. F. Carvalho, M. O. Wallau, C. Bremm, O. Bonnet, J. K. Da Trindade, F. Q. Rosa, T. S. Freitas, y F. G. Moojen, editors. *Boletim Nativao: + de 30 anos de pesquisa em campo nativo*. Editora Viapampa, Uruguaiana.

Rosenberg, K. V, A. M. Dokter, P. J. Blancher, J. R. Sauer, A. C. Smith, P. A. Smith, J. C. Stanton, A. Panjabi, L. Helft, M. Parr, y P. P. Marra. (2019). Decline of the North American avifauna. *Science* 366:120-124.

Ruggia, A., S. Dogliotti, V. Aguerre, M. M. Albicette, A. Albin, O. Blumetto, G. Cardozo, C. Leoni, G. Quintans, S. Scarlato, P. Tittonell, y W. A. H. Rossing. (2021). The application of ecologically intensive principles to the systemic redesign of livestock farms on native grasslands: A case of co-innovation in Rocha, Uruguay. *Agricultural Systems* 191:103148.

Sharps, E., J. Smart, L. R. Mason, K. Jones, M. W. Skov, A. Garbutt, y J. G. Hiddink. (2017). Nest trampling and ground nesting birds: Quantifying temporal and spatial overlap between cattle activity and breeding redshank. *Ecology and Evolution* 7:6622-6633.

Soca, P., F. Olmos, A. Espasandín, D. Bentancur, F. Pereyra, V. Cal, M. Sosa, y M. Do Carmo. (2008). Impacto de cambios en la estrategia de asignación de forraje sobre la productividad de la cría con diversos grupos genéticos bajo pastoreo de campo natural. Pages 110-119 in G. Quintans, J. I. Velazco, y G. Roig (eds.). *Seminario de Actualización: Cría Vacuna*. Montevideo.

Song, X. P., M. C. Hansen, S. V. Stehman, P. V. Potapov, A. Tyukavina, E. F. Vermote, y J. R. Townshend. (2018). Global land change from 1982 to 2016. *Nature* 560:639-643.

Soriano, A., R. León, O. Sala, R. Lavado, V. Deregibus, M. Cauhepé, O. Scaglia, C. Velázquez, y J. Lemcoff. (1992). Rio de la Plata Grasslands. Pages 367–407 *in* R. Coupland, editor. *Ecosystems of the world 8A. Natural grasslands: Introduction and Western Hemisphere*. Elsevier, Amsterdam.

Spitzer, J. C., D. G. Morrison, R. P. Wettemann, y L. C. Faulkner. (1995). Reproductive responses and calf birth and weaning weights as affected by body condition at parturition and postpartum weight gain in primiparous beef cows. *Journal of Animal Science* 73:1251-1257.

Squires, V. R., J. Dengler, H. Feng, y L. Hua. (2018). *Grasslands of the world : diversity, management and conservation*. Page. V. Squires, J. Denglre, H. Feng, y L. Hua (eds.). CRC Press, Boca Raton-London-New York.

Vickery, P. D., P. L. Tubaro, J. M. C. Da Silva, B. G. Peterjohn, J. R. Herkert, y R. B. Cavalcanti. (1999). Conservation of grassland birds in the western hemisphere. *Studies in Avian Biology*:2-26.

Weyland, F., M. P. Barral, y P. Laterra. (2017). Assessing the relationship between ecosystem functions and services: Importance of local ecological conditions. *Ecological Indicators* 81:201-213.

Whittingham, M. J., S. J. Butler, J. L. Quinn, y W. Cresswell. (2004). The effect of limited visibility on vigilance behaviour and speed of predator detection: Implications for the conservation of granivorous passerines. *Oikos* 106:377-385.

Zalba, S. M., y N. C. Cozzani. (2004). The impact of feral horses on grassland bird communities in Argentina. *Animal Conservation* 7:35-44.