UNIVERSIDAD COMPLUTENSE DE MADRID
FACULTAD DE CIENCIAS BIOLÓGICAS

TESIS DOCTORAL

Impact of landfills on breeding populations of White stork (Ciconia ciconia) and future perspectives

Impacto de los vertederos en las poblaciones reproductoras de Cigüeña blanca (Ciconia ciconia) y perspectivas de futuro

MEMORIA PARA OPTAR AL GRADO DE DOCTOR

PRESENTADA POR

Alejandro López García

DIRECTOR

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UNIVERSIDAD
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MADRID

DECLARACIÓN DE AUTORÍA Y ORIGINALIDAD DE LA TESIS PRESENTADA PARA OBTENER EL TÍTULO DE DOCTOR

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Impact of landfills on breeding populations of White stork (Ciconia ciconia) and future perspectives
Impacto de los vertederos en las poblaciones reproductoras de Cigüeña blanca (Ciconia ciconia) y perspectivas de futuro

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Alejandro López García ha sido beneficiario de un contrato predoctoral durante sus 2 últimos años de tesis doctoral dentro del programa “Universidad Complutense de Madrid. Contratos predoctorales de personal investigador en formación. Convocatoria 2019. CT63/19-CT64/19”.

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A mis padres por poner cada peldaño

Y a mi hermano, Diego, por darme la mano al subirlos

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En memoria de Bruno

ÍNDICE - CONTENTS

Abstract 15

Resumen 17

Introducción general 21

Material y métodos generales.. 31

Chapter 1 - Changes in white stork (*Ciconia ciconia*) distribution in the last four decades associated with landfills in Madrid, Spain.. 39

Chapter 2 - Influence of landfill use on breeding nest-site selection and breeding success in white storks (*Ciconia ciconia*)... 65

Chapter 3 - The trade-offs of foraging at landfills: Landfill use enhances hatching success but decrease the juvenile survival of their offspring on white storks (*Ciconia ciconia*).. 93

Chapter 4 - Back to the future: long term effects of landfill closure on breeding white stork (*Ciconia ciconia*) populations. 117

Discusión general... 141

Líneas de investigación futuras 149

Conclusions... 152

Conclusiones... 153

Referencias 155

Anexo I: Censo reproductor de Cigüeña blanca 2021 en Madrid 189

Glosario de tablas y figuras 205

ABSTRACT

Human pressure has substantially transformed the surface of Earth. Habitat transformation enhanced the decrease of many bird species over the centuries. However, some of these species have recovered in the last decades in association with the use of landfills. Feeding at these predictable anthropogenic food subsides reduces the cost of foraging and implies a positive effect on demographic parameters. The high food abundance at landfills may attract a huge number of birds with consequences at population and ecological level. In addition, feeding at landfills is not exempt of some risks. Poor quality food, ingestion of plastics and other pollutants, and proliferation of pathogens may negative affect actual or future life-history traits.

Knowing the effects of landfills on several aspects of the ecology of breeding populations is essential to predict and to understand the potential impact of landfill closure. In this way, we can assess management measures of bird populations.

The main objective of this thesis is to increase the knowledge about the effect of landfills over breeding populations in the short and the long term. Particularly, we aim to disentangle the impact of the use of landfills in the spatial distribution of breeding pairs, nest-site selection, offspring survival, and the potential effects of landfill closure on white storks (*Ciconia ciconia*; Linnaeus, 1758).

In the first two chapters, we work at a population level in the Province of Madrid, Spain. The **first chapter** confirms the effect of landfills on spatial distribution. Pairs bred closer to landfills over the last four decades, showing the attraction of such facilities for birds. Moreover, storks concentrate in higher densities in the vicinity of landfills over the last decades.

The **second chapter** focuses on how landfills affect nest-site selection in the close vicinity area of the nest. Pairs breeding near landfills nest in places surrounded by urban habitat and low-quality natural feeding areas (*i.e.*, arable lands) while breeders away from landfill show high preference for pastures and dehesas and forest near the nest. Our results also demonstrate that not only landfill but habitat quality in the surrounding area of the nest play a key role on breeding outcome.

In the **third chapter**, we deepen into the impact of landfills at an individual level. Using capture-recapture procedures, we show that feeding at landfills enhances offspring survival during nest stage but represents a negative impact on juvenile survival. These results suggest that foraging at landfills implies a trade-off between current and late survival. Consequently, using landfills as a complementary food source seems to be the best strategy.

In the **fourth chapter**, we use species distribution models (SDM) to explore the future impact of landfills closure on breeding white stork population in two different climate-change and urbanization scenarios. The models project a dramatic reduction of the breeding population range and a decrease in the nest density in the future. The future projections reveal that climate-change, landfills closure and increasing urbanization are the major threats for this species.

Overall, our results show the relevance of landfills on breeding white stork populations in the centre of the Iberian Peninsula. This thesis supports that landfills have been playing a key role in the recovery and thrive of the Western white stork populations. Nonetheless, this work alerts over the negative effects of these facilities on long-term survival. Moreover, this thesis highlights the importance of population monitoring programs to assess long-term effects of human activities and to make decisions on conservation measures or population management. To avoid this species to become threatened again, we propose to maintain the availability of non-anthropogenic foraging areas and a gradual transition during landfills closure.

RESUMEN

La presión humana ha transformado sustancialmente la superficie de la Tierra. La transformación de los hábitats ha favorecido la disminución de muchas especies de aves a lo largo de los siglos. Sin embargo, algunas de estas especies se han recuperado en las últimas décadas asociadas con el uso de los vertederos. La alimentación en estas fuentes de origen antrópico y predecibles reduce los costes del forrajeo e implica un efecto positivo en los parámetros demográficos. La gran abundancia de alimento en los vertederos podría atraer a un ingente número de aves con consecuencias tanto a nivel poblacional como ecológico. Además, forrajear en los vertederos no está exento de riesgos. La baja calidad del alimento, la ingesta de plásticos y otros contaminantes, y la proliferación de patógenos podrían afectar negativamente sus rasgos vitales, actuales y futuros.

Conocer el efecto de los vertederos sobre distintos aspectos de la ecología de las poblaciones reproductoras es esencial para predecir y entender el posible impacto debido a la clausura de los vertederos y así, poder determinar las medidas de gestión que se podrán llevar a cabo sobre las poblaciones de aves.

El principal objetivo de esta tesis es incrementar el conocimiento del efecto de los vertederos en las poblaciones reproductoras a corto y largo plazo. En concreto, se ha estudiado el impacto del uso de los vertederos en la distribución espacial de las parejas reproductoras, la selección del lugar de nidificación, la supervivencia de la descendencia y los efectos potenciales del cierre de los vertederos en las cigüeñas blancas (*Ciconia ciconia*; Linnaeus, 1758).

En los primeros dos capítulos, se ha realizado el estudio a nivel poblacional. En el **primer capítulo** se ha confirmado el efecto de los vertederos en la distribución espacial. Las parejas crían más cerca de los vertederos con el paso del tiempo, mostrando la atracción de las aves por estas instalaciones. Además, las cigüeñas se concentran en grandes densidades en las proximidades de los vertederos a lo largo de las últimas décadas.

El **segundo capítulo** se centra en como los vertederos afectan a la selección del lugar de nidificación en el área circundante al nido. Las parejas que crían cerca de los vertederos

nidifican en lugares rodeados por hábitat urbano y áreas de alimentación natural de baja calidad (como tierras de labor) mientras que los reproductores alejados de los vertederos muestran una fuerte preferencia por pastos, dehesas y bosques en las proximidades del nido. Los resultados muestran que no sólo el vertedero, si no la calidad del hábitat en el área circundante al nido, tienen un efecto importante sobre los parámetros de la reproducción.

En el **tercer capítulo**, se profundiza en el estudio del impacto de los vertederos a nivel individual. Para ello, se han usado modelos de captura-recaptura, y se muestra que, gracias a la alimentación en los vertederos, se aumenta la supervivencia durante la etapa en el nido, pero representa un impacto negativo en la supervivencia juvenil. Estos resultados sugieren que el forrajeo en los vertederos implica un equilibrio entre la supervivencia actual y la futura. En consecuencia, el uso de los vertederos como fuente de alimentación complementaria parece ser la mejor estrategia.

En el **cuarto capítulo**, se usaron modelos de distribución de especies (SDM) para explorar el impacto futuro del cierre de los vertederos en las poblaciones reproductoras de cigüeñas blancas en dos escenarios diferentes de cambio climático y crecimiento urbano. Los modelos proyectan una reducción dramática del rango de distribución de la población reproductora y una disminución en la densidad de nidos en el futuro. Las proyecciones a futuro revelan que el cambio climático, la clausura de los vertederos y el creciente proceso de urbanización, son las mayores amenazas para esta especie.

En conjunto, nuestros resultados muestran la relevancia de los vertederos en las poblaciones reproductoras de cigüeña blanca en el centro de la Península Ibérica. Esta tesis sustenta que los vertederos han jugado un papel clave en la recuperación e incremento de las poblaciones de cigüeña blanca occidentales. No obstante, en este trabajo se alerta de los efectos negativos de estas instalaciones en la supervivencia a largo plazo. Además, en esta tesis se destaca la importancia de los programas de seguimiento de poblaciones para evaluar los efectos de las actividades humanas a largo plazo y la toma de decisiones sobre medidas de conservación o gestión de las poblaciones. Para evitar que esta especie vuelva a estar amenazada, proponemos mantener la disponibilidad de las áreas de forrajeo no antrópicas y una transición paulatina del cierre de los vertederos.

INTRODUCCIÓN GENERAL

Las actividades desarrolladas por el ser humano han influido significativamente en la biodiversidad desde hace miles de años (Vitousek *et al.*, 1997; Lewis & Maslin, 2015; Boivin *et al.*, 2016). Se estima que aproximadamente dos tercios de la superficie terrestre han sido alterados por el ser humano respecto a su estado previo (Venter *et al.*, 2016). La presión humana sobre la biodiversidad se ha incrementado en los últimos años. Tan sólo en las últimas 2 décadas casi una décima parte del tercio *inalterado* del globo terráqueo, considerado como áreas naturales, ha dejado de serlo (Venter *et al.*, 2016; Watson *et al.*, 2016). Y esta situación puede empeorar en el futuro con el crecimiento de la población humana (Pereira *et al.*, 2010; Powers & Jetz, 2019). La degradación y pérdida de hábitats debido al cambio climático, la sobreexplotación de los recursos y el incesante incremento de la urbanización y los recursos requeridos asociados a la misma, han sido las principales causas de la actual pérdida de biodiversidad (Dirzo *et al.*, 2014; Di Marco *et al.*, 2018). No en vano, algunos investigadores llaman a este periodo el Antropoceno (Crutzen, 2002; Lewis & Maslin, 2015).

En los ambientes modelados por el ser humano, no sólo se han alterado la distribución y disponibilidad de los recursos tróficos, sino que han surgido nuevas fuentes de alimentación (Burns *et al.*, 2021). Estos nuevos recursos surgieron hace miles de años, en los albores de la domesticación, cuando los desperdicios dejados por los campamentos de los seres humanos eran explotados por animales silvestres (Axelsson *et al.*, 2013; Beckman *et al.*, 2022). Desde entonces, las especies que han sido capaces de aprovechar estos recursos, han prosperado. Estas especies a menudo comparten características comunes: predisposición a utilizar nuevos recursos, dieta generalista y oportunista, tolerancia a las perturbaciones del ser humano y adaptación o plasticidad de sus comportamientos y fisiología a los entornos humanizados (Kark *et al.*, 2007; Tuomainen & Candolin, 2011). Con el crecimiento de la población humana y el desarrollo de las sociedades sedentarias, se ha acrecentado el desperdicio de alimentos, que en 2011 se estimaba en 1.300 millones de toneladas anuales en el mundo (FAO, 2011) (Fig. 1). Este fenómeno ha fomentado el estudio de la fauna silvestre en relación a las denominadas fuentes de alimentación predecibles antrópicas (PAFS, del inglés "Predictable Anthropogenic Food Subsides") (Oro *et al.*, 2013). Los PAFS son un conjunto de fuentes de alimentación proporcionadas a los animales por el ser humano de forma intencionada o no y cuya disponibilidad se repite en el tiempo y el espacio (por ejemplo: los descartes pesqueros, los comederos para aves o los vertederos (Oro *et al.*, 2013)). Entre los PAFS de los ambientes terrestres destacan los vertederos por su abundancia y su carácter global (Oro *et al.*, 2013; Chen *et al.*, 2020).

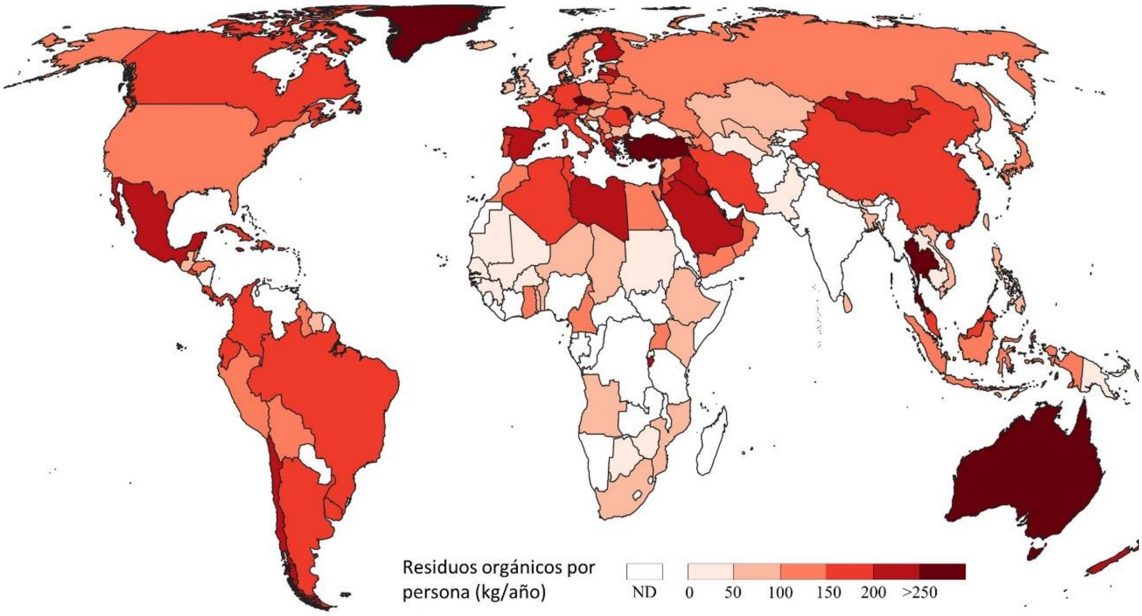


Fig. 1. Mapa de la cantidad de residuos orgánicos generados (Kg) por habitante al año en cada país del mundo. Fuente: What A Waste Global Database (The World Bank).

Los vertederos controlados o sanitarios (en inglés "landfills") surgen de la necesidad de gestionar los vertidos de residuos sólidos aislados descontrolados y, así, reducir la contaminación del suelo, del agua y del aire del entorno (Real Decreto 646/2020, Directiva 1999/31/CE). En estas superficies de terreno perimetradas y con una base de sustentación impermeable para recoger los lixiviados, se dispone la basura orgánica junto a determinados materiales inorgánicos que, tras alcanzar una franja de grosor determinada, son recubiertos por una capa de áridos y comprimidos de forma sucesiva (Nanda & Berruti, 2021). De esta manera, la basura orgánica queda expuesta hasta ser cubierta por una nueva capa de arena y es compactada.

Sólo en los vertederos europeos se vierten más de 88 millones de toneladas de basura orgánica al año (Stenmarck *et al.*, 2016). Esta cantidad ingente de recursos atrae a multitud de vertebrados, mayoritariamente aves, por su facilidad para acceder a estas instalaciones (Plaza & Lambertucci, 2017; Arnold *et al.*, 2021). La abundancia y predictibilidad en el tiempo y en el espacio de los recursos presentes en los vertederos produce la alteración del comportamiento de las aves. Como resultado de la alimentación en estas zonas, se acorta o se suprime el comportamiento migratorio (Flack *et al.*, 2016; Rotics *et al.*, 2016, 2017) o se modifican los desplazamientos diarios (Spelt *et al.*, 2021). La concentración de dichos residuos en un punto fijo y su renovación constante a lo largo del tiempo disminuye tanto el coste energético por la búsqueda de alimento como el tiempo mismo de forrajeo (Gilbert *et al.*, 2016; Soriano-Redondo *et al.*, 2021). Por consiguiente, aumenta la inversión parental dedicada a otras tareas, que junto a la casi ilimitada disponibilidad trófica, repercute directamente en los parámetros de la reproducción (Moritzi *et al.*, 2001). Las aves que crían cerca de los vertederos tienen mayores tamaños de puesta, mayor viabilidad de los huevos, mejor condición corporal y mayor supervivencia de la descendencia durante su fase de crecimiento en el nido (Tortosa *et al.*, 2002; Auman *et al.*, 2008; Steigerwald *et al.*, 2015; Djerdali *et al.*, 2016a; Pineda-Pampliega *et al.*, 2021). En última estancia, todo esto podría incrementar el número de efectivos reclutados, incidiendo en la dinámica de la población.

Este efecto de los vertederos sobre la dinámica poblacional ha permitido la recuperación de varias especies amenazadas o en peligro de extinción (Tauler-Ametller *et al.*, 2017). Siguiendo la Teoría del Forrajeo Óptimo ("Optimal Foraging Theory"), cada vez más individuos buscarán alimento en los vertederos, e incluso, criarán más cerca de los mismos al ser parches de alimentación abundante y constante (Schoener, 1971; Stephens & Krebs, 1986). A largo plazo, puede que el aumento de efectivos (*i.e.,* atracción de migrantes e incremento de descendientes) desplace la capacidad de carga del sistema basado en los recursos tróficos naturales, puesto que la disponibilidad de alimento queda asegurada por el aporte constante de los vertederos y no depende de condicionantes extrínsecos (por ejemplo: condiciones climáticas, enfermedades de especies presa, etc.) (Duhem *et al.*, 2008). De hecho, en periodos de escasez de alimento natural se incrementa el uso de las fuentes de alimentación antrópicas (Evans & Gawlik, 2020; O'Hanlon *et al.*, 2022). La situación comentada anteriormente puede favorecer la abundancia monoespecífica resultando en el incremento de interacciones con el ser humano y cambiando la percepción de la sociedad. Este cambio podría ser un factor determinante en la conservación de las especies (Castillo-Huitrón *et al.*, 2020) puesto que podrían pasar a ser consideradas especies super-abundantes o especies plaga (Duhem *et al.*, 2008; Payo-Payo *et al.*, 2015; Plaza & Lambertucci, 2017). La proliferación de estas especies puede desplazar a otras o afectar a las redes tróficas, resultando en sistemas con comunidades más homogéneas y alterando la funcionalidad del ecosistema (Newsome *et al.*, 2015; Arnold *et al.*, 2021).

Además del impacto a nivel ecosistémico, forrajear en los vertederos implica una serie de riesgos para los propios individuos (Plaza & Lambertucci, 2018; Tauler-Ametllller *et al.*, 2019; Pineda-Pampliega *et al.*, 2020). En estas instalaciones los residuos orgánicos a veces se encuentran mezclados con elementos tóxicos o nocivos, como plásticos, cordeles y metales pesados, que pueden ocasionar la intoxicación y/o ahogamiento de las aves (Peris, 2003; Smits *et al.*, 2005; Henry *et al.*, 2011; Pérez-López *et al.*, 2016). Además, los desperdicios a menudo se encuentran en avanzado estado de descomposición, lo que produce la proliferación de diversos microorganismos, e incluso algunos de ellos patógenos (Camacho *et al.*, 2016; Plaza *et al.*, 2019; Höfle *et al.*, 2020; Martín-Maldonado *et al.*, 2020). Estos últimos podrían ser trasladados por sus hospedadores a distintas áreas, lo que ligado al uso de masas de agua o campos de cultivo, que realizan muchas de las especies de aves que se alimentan en los vertederos, podría suponer un motivo de preocupación para la salud pública y la sanidad animal (Hatch, 1996; Navarro *et al.*, 2019).

No obstante, no todos los individuos de una misma población comparten los mismos rasgos (Bolnick *et al.*, 2003; Araújo *et al.*, 2011). La variabilidad intra-específica en los rasgos morfológicos, fisiológicos o comportamentales, afectan a la capacidad de los individuos para explotar diversos recursos tróficos. Por ejemplo, las cigüeñas procedentes del resto de Europa hacen un uso más intenso de los vertederos durante la invernada que los individuos locales del sur de España, los cuáles prefieren alimentarse en los arrozales (Sanz-Aguilar *et al.*, 2015). Por tanto, los individuos de una misma especie pueden no estar expuestos en la misma medida a los efectos nocivos de la alimentación en los vertederos o pueden presentar diversas estrategias para lidiar con dichos efectos.

Pese a que el tratamiento actual de los residuos en los vertederos supone una mejora sustancial respecto al vertido descontrolado, tal como se ha señalado anteriormente, aún existe una serie de impactos negativos sobre el medio ambiente, junto con el potencial efecto nocivo para la salud pública y la sanidad animal (Matejczyk *et al.*, 2011; Navarro *et al.*, 2019; Vaverková, 2019). La Comisión Europea, por tanto, ha dictaminado una serie de directivas (Directiva 1999/31/EC y 2018/850/EC) con el propósito de mejorar la gestión de los residuos, como la reducción de los residuos orgánicos generados, el reciclaje y reaprovechamiento de aquellos residuos destinados a vertedero además de limitar o impedir el acceso de animales a dichas instalaciones.

Estructura de la tesis

En esta tesis se han analizado los efectos de una fuente de alimentación antrópica predecible, como son los vertederos, sobre distintos aspectos de las poblaciones de cigüeña blanca (*Ciconia ciconia* Linnaeus, 1758) durante el periodo de reproducción (Fig.2). Mientras que la mayoría de los estudios se han centrado en el efecto de los vertederos sobre los patrones de movimiento y el periodo de invernada o en los efectos directos sobre los pollos (ya comentados anteriormente), este estudio profundiza en los efectos de esta fuente de alimentación antrópica sobre la población reproductora y su descendencia a medio-largo plazo. Se ha estructurado esta tesis en cuatro capítulos.

En los primeros dos capítulos se aborda el cambio en la distribución espacial de la población reproductora y en los requerimientos de hábitat en distintos momentos a lo largo de un periodo de casi 40 años. Dada la atracción que esta fuente de alimentación representa para los individuos y sus beneficios asociados con el incremento de la productividad, cabe esperar que existan cambios en la distribución de la especie tras el comienzo del uso de los vertederos, con la agregación de los individuos reproductores entorno a ellos. Esta hipótesis se aborda en el primer capítulo gracias a la recopilación de información de censos históricos, que ha permitido evaluar los cambios en la distribución de la especie, así como en las características de las áreas ocupadas tras el comienzo de la alimentación en los vertederos.

A partir de datos de los censos tras la apertura de los vertederos, el segundo capítulo se centra en el estudio de los efectos de la distancia a los vertederos en la selección del lugar de nidificación y el éxito reproductor. Los individuos reproductores dependerán menos de los recursos tróficos naturales del entorno al disponer de los vertederos como fuente de alimentación complementaria (Evans & Gawlik, 2020; Zorrozua *et al.*, 2020a; O'Hanlon *et al.*, 2022). Por tanto, los criterios de la selección del lugar de nidificación pueden cambiar progresivamente hacía otros recursos más escasos (Moreira *et al.*, 2017; Spelt *et al.*, 2019).

Aunque los efectos de los vertederos sobre la descendencia han sido ampliamente estudiados en aves, apenas se tiene constancia de los efectos de este tipo de alimentación a medio-largo plazo. La presencia de contaminantes y productos tóxicos entre los residuos orgánicos (De la Casa-Resino *et al.* 2014, 2015a, b) podría tener efectos negativos sobre la supervivencia de los descendientes a largo plazo. En el tercer capítulo se ahonda en los efectos a nivel individual del vertedero en la supervivencia de la descendencia. Para ello, se ha usado un análisis de captura-recaptura (Lebreton *et al.*, 1992) de datos de individuos marcados, pertenecientes a una única colonia, en la que se ha realizado un seguimiento a lo largo de 20 años.

Por último, los resultados previos parecen señalar una fuerte dependencia entre el crecimiento y expansión de la población estudiada y el uso de vertederos como fuente de alimentación. Por tanto, la aplicación de la Directiva Europea de Vertederos (Directiva 1999/31/EC y 2018/850/EC), en la que se pretende mejorar la gestión de los residuos, podría tener consecuencias en las dinámicas poblacionales de las especies que se favorecen de los recursos tróficos de estas instalaciones. Sin embargo, las consecuencias de esta situación sólo han sido estudiadas en gaviotas hasta el momento (Steigerwald *et al.*, 2015; Zorrozua *et al.*, 2020a; Delgado *et al.*, 2021). Por tanto, en el cuarto capítulo se investiga el posible efecto del cierre de los vertederos bajo dos posibles escenarios de cambio climático y crecimiento de la población humana. Se supone que la desaparición de dicha fuente antrópica suplementaria podría producir una reducción en el tamaño poblacional y, posiblemente, en la distribución de la especie. Esta reducción podría ser aún más pronunciada debido al incremento de los fenómenos climáticos adversos en el futuro. El uso de modelos de distribución de especies (SDM, "Species

Distribution Models", por sus siglas en inglés) permite realizar predicciones en la población estudiada (Thuiller, 2004; Barbet-Massin *et al.*, 2012). Determinar el efecto del cierre de los vertederos es crucial para la gestión de las especies que se alimentan en estas instalaciones.

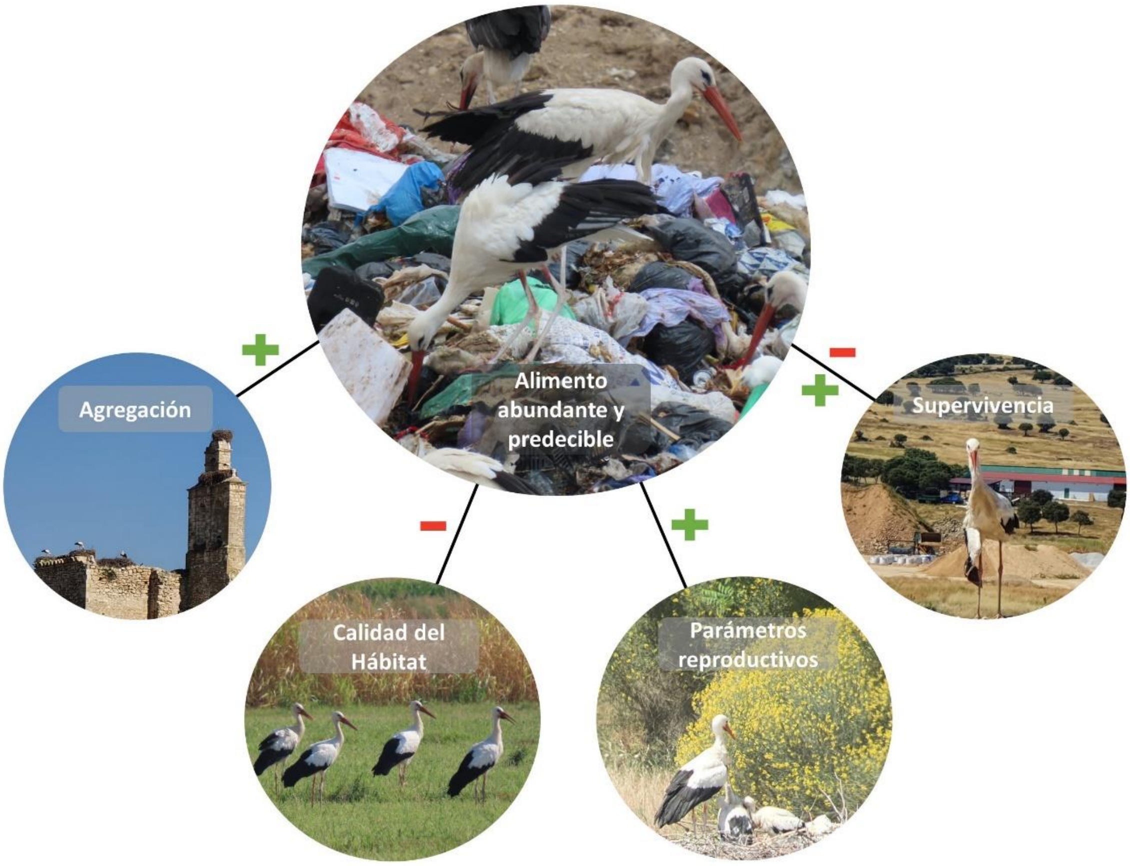


Fig. 2. Efectos del uso de los vertederos en las poblaciones reproductoras de cigüeña blanca analizados en esta tesis. Se señala el incremento (+) o disminución (-) en cada parámetro.

Los resultados y las conclusiones obtenidas nos proporcionan una perspectiva global e integradora acerca de las implicaciones de los vertederos sobre las poblaciones reproductoras de cigüeña blanca en el centro de la Península Ibérica (Fig.2) a la par que nos permite sugerir una serie de recomendaciones para la gestión de la especie llegado el momento de clausurar los vertederos actuales.

General objectives

The general objective of this study is to provide a relevant contribution to the impact of landfills on the distribution, nest-site selection and breeding success of a model species such as the White stork. As a framework to cover this objective, we have used a well-known region where historical data were available before and after landfill exploitation. Moreover, we particularly:

1. Analysed the role of landfills in the distribution of the breeding population of white storks in Madrid in the last four decades. We explored historical distribution patterns before and after the use of this food subside (**Chapter 1** - Changes in white stork (*Ciconia ciconia*) distribution in the last four decades associated with landfills in Madrid, Spain).
    a. Are pairs breeding closer to landfills over the years?
    b. Do landfills promote the aggregation of individuals?
    c. Characterization of newly occupied and abandoned areas.
2. Determined the impact of distance to landfill on nest-site selection and its consequences on breeding success (**Chapter 2** - Influence of landfill use on nest-site selection and breeding success in white storks (*Ciconia ciconia*)).
3. Evaluated the effects of individual differences in the intensity of landfill use during breeding season on offspring survival at several stages (**Chapter 3** - The trade-offs of foraging at landfills: Landfill use enhances hatching success but decrease the juvenile survival of their offspring on white storks (*Ciconia ciconia*)).
4. Deepened into the potential future perspective of White Stork population after landfill closure under different climate change and population growth scenarios (**Chapter 4** - Back to the future: long term effects of landfill closure on breeding white stork (*Ciconia ciconia*) populations)).

Objetivos generales

El objetivo general de esta tesis es contribuir de forma relevante al estudio del impacto de los vertederos sobre distintos aspectos de las poblaciones reproductoras: la distribución, la selección del lugar de nidificación y el éxito reproductor de una especie modelo, como es la cigüeña blanca. Para cubrir este objetivo, se ha empleado una región bien estudiada en la que disponíamos de datos históricos del antes y el después del uso de los vertederos. Más en detalle, en este trabajo se pretende:

1. Analizar el papel de los vertederos en la distribución de la población reproductora de cigüeñas blancas en Madrid en las últimas cuatro décadas. Se han explorado los patrones de distribución históricos previos al uso de esta fuente de alimentación y aquellos posteriores. (**Capítulo 1** - Cambios en la distribución de cigüeña blanca en las últimas cuatro décadas asociadas con los vertederos en Madrid, España).
    a. ¿Las parejas crían más cerca de los vertederos a lo largo de los años?
    b. ¿Promueven los vertederos la agregación de individuos?
    c. Caracterización de las nuevas áreas ocupadas y de las abandonadas.
2. Determinar el impacto de la distancia al vertedero en la selección del lugar de nidificación y sus consecuencias en el éxito reproductivo (**Capítulo 2** - Influencia del uso del vertedero en la selección del lugar de nidificación y el éxito de cría en las cigüeñas blancas (*Ciconia ciconia*)).
3. Evaluar los efectos de las diferencias individuales en la intensidad de uso del vertedero durante la temporada de cría en la supervivencia de la descendencia en diferentes etapas (**Capítulo 3** – Compensación de la alimentación en vertederos: el uso del vertedero incrementa el éxito de eclosión, pero disminuye la supervivencia juvenil de su descendencia en las cigüeñas blancas (*Ciconia ciconia*)).
4. Profundizar en la potencial perspectiva de futuro de la población de cigüeña blanca tras la clausura de los vertederos bajo diferentes escenarios de cambio climático y crecimiento de la población (**Capítulo 4** – Regreso al futuro: efectos a largo plazo de la clausura de los vertederos en las poblaciones reproductoras de cigüeña blanca (*Ciconia ciconia*)).

MATERIAL Y MÉTODOS GENERALES

Especie y área de estudio

La cigüeña blanca es una especie modelo ideal para el estudio del impacto de los vertederos sobre la fauna. Esta especie, oportunista y generalista, se alimenta en humedales, pastos, prados húmedos y dehesas (Alonso *et al.*, 1991; Radović *et al.*, 2015; Zurell *et al.*, 2018; Bialas *et al.*, 2021) dónde captura lombrices, insectos, peces, anfibios, reptiles e, incluso, pequeños vertebrados (Antczak *et al.*, 2002; Chenchouni, 2016; Orłowski *et al.*, 2018). Estos hábitats se consideran zonas de forrajeo de buena calidad debido a las elevadas tasas de éxito de captura y a la abundancia de presas de alto valor nutricional (Alonso *et al.*, 1991; Tryjanowski *et al.*, 2005; Zbyryt *et al.*, 2020; Golawski & Kasprzykowski, 2021). La alta adaptabilidad de la especie permite además explotar zonas de forrajeo sub-óptimas que requieren de una mayor inversión energética debido a la mayor dispersión y menor disponibilidad de recursos tróficos, como los campos arables (Nowakowski, 2003; Orłowski *et al.*, 2019).

Debido a los cambios en el uso del suelo por el deterioro de los humedales, la intensificación de la agricultura y el auge de la urbanización, se produjo un declive de la población occidental europea a comienzos de la segunda mitad del siglo XX, que alcanzó su mínimo poblacional en la década de los 80 (Bernis, 1981; Candeias & Araujo, 1989; Barlein, 1991). Ante esta situación, la población occidental de la especie (Luthin, 1987) se incluyó en catálogos de especies amenazadas y listados nacionales e internacionales, amparados en regímenes jurídicos básicos de conservación, uso sostenible, mejora y restauración de su hábitat. En la actualidad se encuentra incluida en el Anexo I de la Directiva Aves, por lo que es objeto de estudio y debe poseer medidas de conservación especiales en cuanto a su hábitat, con el fin de asegurar la supervivencia y la reproducción en su área de distribución a nivel europeo. También a nivel internacional, se encuentra listada como Especie Estrictamente Protegida en el Anexo II del Convenio de Berna, relativo a la Conservación de la Vida Silvestre y del Medio Natural en Europa, e incluida además en el Anexo II del Convenio de Bonn relativo a la Conservación de Especies Migratorias (CMS). Tras este fuerte declive generalizado se produjo una progresiva e incesante recuperación de las poblaciones en el suroeste de Europa

(Portugal, España y parte de Francia) (Schulz, 1999) asociado al comienzo de la explotación de los vertederos como fuente trófica por parte de esta especie junto con las medidas de conservación aplicadas (Blanco, 1996; Tortosa *et al.*, 2002; Massemin-Challet *et al.*, 2006; Moreira *et al.*, 2017) (Fig. 1). Por ello, a nivel nacional se excluyó en 2011 del Catalogo Nacional de Especies Amenazadas y no ha sido incluida en el Libro Rojo de los Aves Reproductoras de España (Madroño *et al.*, 2004), aunque se mantiene incluida en el Anexo IV de la Ley 42/2007, del Patrimonio Natural y de la Biodiversidad, en el Listado de Especies Silvestres en Régimen de Protección Especial (EEPE, Real Decreto 139/2011).

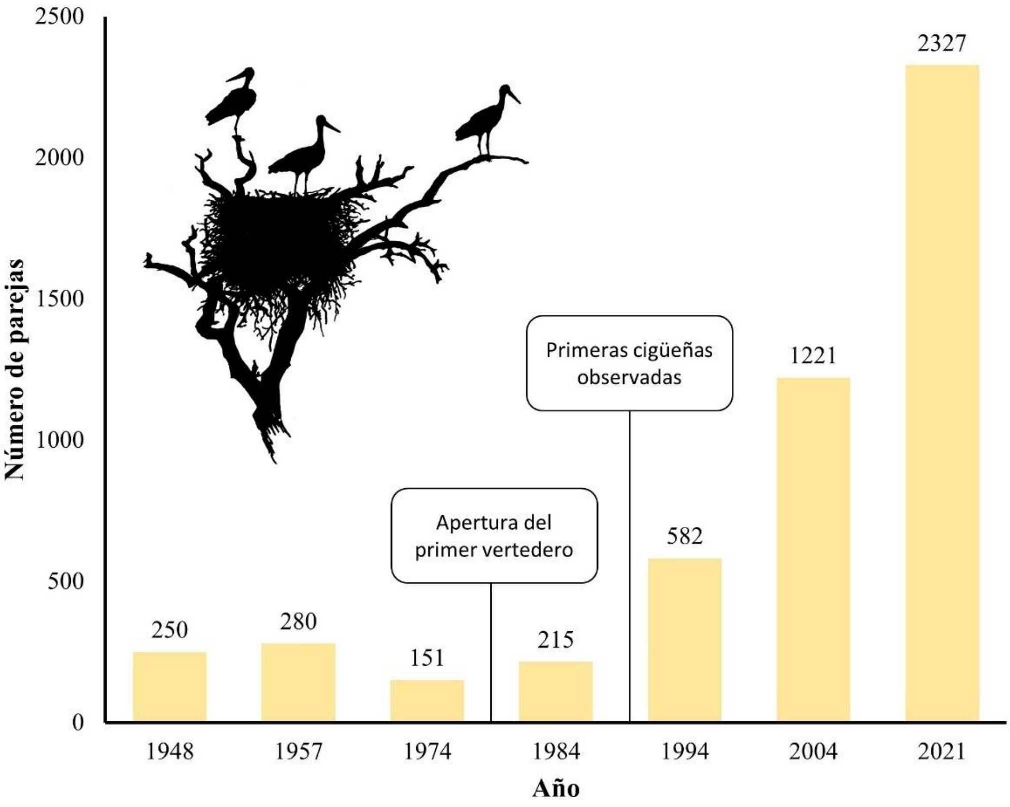


Fig. 1. Número de parejas reproductoras de cigüeña blanca en la Comunidad de Madrid desde 1948. Se señala el momento en el que produce la apertura del primer vertedero en la región y el periodo en el que se comienzan a observar las primeras cigüeñas en los vertederos. **Nota:** la metodología empleada en los censos anteriores a 1984 es diferente. **Fuente:** Bernis, 1981. Lázaro *et al.*, 1986; Informe SEO/Birdlife, 1995; Molina & Del Moral, 2004; López-García & Aguirre, *In Press*. Imagen: Carlos Hernáez.

En la Comunidad de Madrid, área de estudio de esta tesis, la población de cigüeña blanca siguió esta tendencia generalizada de disminución y posterior recuperación. Por ello, se incluyó en el Catálogo Regional del Especies Amenazadas con la categoría de Vulnerable (Decreto 18/1992). La población reproductora de Madrid pasó de 215 parejas en 1984

(Lázaro *et al.* 1986) a 582 parejas en 1994 (SEO/Birdlife 1995) y siguió aumentando hasta las 1221 parejas registradas en 2004 (Molina and Del Moral 2005), tendencia que se mantiene en la actualidad hasta alcanzar las 2327 parejas reproductoras en 2021 (López & Aguirre, *en prensa*) (Fig. 1). A su vez, el primer vertedero de residuos sólidos urbanos entró en funcionamiento en 1978, con el fin de regular el vertido de residuos hasta ese momento descontrolado (Ley 42/1975). Este plan de gestión promovió que los desperdicios de las ciudades se concentraran en grandes cantidades en estas instalaciones (Tabla 1). A este vertedero le siguieron otros tres (Alcalá de Henares, 1984; Colmenar Viejo, 1985; y Pinto, 1986) (Fig. 2, Tabla 1). Unos pocos años después, a finales de la década de los 80, se observaron las primeras cigüeñas alimentándose en los vertederos de esta región (Manuel Fernández Cruz, datos sin publicar). A principio de los 90 se confirmó el papel de los vertederos como importantes puntos de parada durante la migración y lugares de concentración de individuos durante la invernada (Marchamalo, 1994; Blanco, 1996; Marchamalo *et al.*, 1997; Vergara *et al.*, 2004) (Tabla 1).

Tabla 1. Principales Vertederos de Residuos Sólidos Urbanos de la Comunidad de Madrid hasta la actualidad, ordenados según fecha de apertura. Se incluye el año de apertura y clausura para cada instalación, señalando como "Activo" aquellos que se encuentran en funcionamiento durante el año 2022. Se indica la cantidad de vertido (en toneladas) para cada vertedero en los años de los censos de **2001, 2004 y 2021**; excepto para el año **1984,** puesto que no se dispone de dicha información. En el caso de los vertederos señalados con asterisco la cantidad de vertidos corresponde a 2018 (*)

Nombre	**Apertura**	**Clausura**	**2001**	**2004**	**2021**	**Cigüeñas OBSERVADAS**
Valdemingomez/Las Dehesas (Madrid)	1978	Activo	701199	816429	782757	Sí
Alcalá de Henares	1984	2021	130672	133715	204000	Sí
Colmenar Viejo	1985	Activo	313564	330146	299237*	Sí
Pinto	1986	Activo	823660	889840	815028*	Sí
Colmenar de Oreja	1986	2002	35721	Clausurado	Clausurado	Nunca
Nueva Rendija (San Fernando de Henares)	1995	2009	141464	174399	Clausurado	Menos de 20 cigüeñas observadas

Fuente: Estrategia de residuos de la Comunidad de Madrid 2006-2016 (Informe de la Comunidad de Madrid), Informes de El Medio Ambiente en la Comunidad de Madrid (Informe de la Comunidad de Madrid), y Memoria anual de generación y gestión de residuos de competencia municipal (Informe del Ayuntamiento de Madrid).

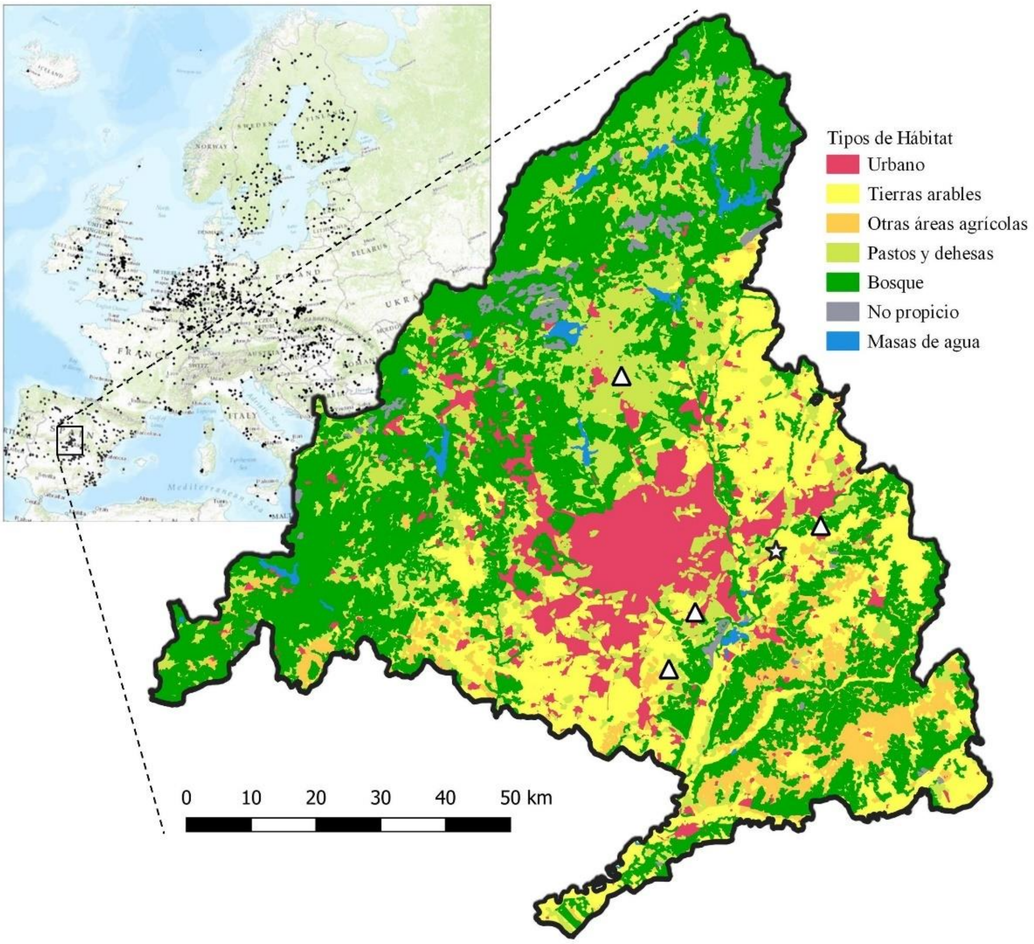


Fig. 2. Mapa de los vertederos y los tipos de hábitat principales de la Comunidad de Madrid. Los vertederos de la Comunidad se representan mediante triángulos blancos (de norte a sur: Colmenar Viejo, Alcalá de Henares, Valdemingómez/Las dehesas y Pinto). El vertedero de Nueva Rendija está representado por un asterisco. En la esquina superior derecha se representa los vertederos (puntos) de la Unión Europea incluidos en la capa "Corine Land Cover 2018", destacando la ubicación de la Comunidad de Madrid. Mapa de los vertederos Europeos modificado de: Szabóa *et al.* 2017.

En el área de estudio, el periodo de reproducción comienza en febrero con la formación de las parejas reproductoras (Bernis, 1981; Aguirre & Vergara, 2009). Los huevos eclosionan en abril-mayo y los pollos permanecen en el nido durante aproximadamente dos meses. A partir de los 45 días se yerguen sobre sus patas traseras lo que facilita su conteo durante los censos (Bernis, 1981). Esto permite obtener un valor de productividad de cada pareja, aunque no un valor de éxito reproductor, puesto que resulta complicado contabilizar el número de huevos dada la altura a la que se encuentran la mayoría de los nidos. Al igual que otras muchas especies de aves altriciales, la movilidad de las parejas de cigüeña blanca está limitada durante toda la temporada de reproducción debido a la necesidad constante de aporte de calor y la ceba de los pollos (Schoener, 1971; Ricklefs, 1979). Estudios previos demuestran que durante la temporada de reproducción la distancia de forrajeo no suele superar los 2 kilómetros de distancia (Alonso *et al.*, 1991; Johst *et al.*, 2001; Olsson & Bolin, 2014; Zurell *et al.*, 2018). La explotación de fuentes de alimentación antrópica supera ampliamente dicha distancia, pero rara vez superan los 25-30 kilómetros (Gilbert *et al.*, 2016; Bécares *et al.*, 2019; Soriano-Redondo *et al.*, 2021). Además, la probabilidad e intensidad de uso del vertedero disminuye a medida que este se encuentra más alejado del lugar de nidificación (Gilbert *et al.*, 2016; Soriano-Redondo *et al.*, 2021).

Seguimiento de las poblaciones: censos y marcado

El seguimiento a largo plazo de las poblaciones animales proporciona una excelente herramienta para evaluar el impacto del ser humano sobre las mismas (Koskimies, 1989; Balmford *et al.*, 2003). En este aspecto, las aves han demostrado ser buenos bioindicadores por ser ubicuos, fáciles de detectar, ampliamente estudiados y ser sensibles a los cambios ambientales (Gregory *et al.*, 2005). De esta forma, el estudio de las poblaciones reproductoras y de los parámetros de la reproducción son muy útiles a la hora de detectar las perturbaciones en el ambiente a corto-medio plazo (Siriwardena *et al.*, 1998). Por su antigüedad y su eficacia demostrada, los censos y el marcado mediante anillamiento científico constituyen metodologías destacadas para el seguimiento de poblaciones.

Entre los diversos censos de aves realizados en España, los de cigüeña blanca destacan por su larga tradición y popularidad (Bernis, 1981; Chozas, 1983; Lázaro *et al.*, 1986). Se añade a esto la estandarización de la metodología de muestreo desde hace más de 40 años en toda Europa, que facilita la comparación entre regiones y a lo largo del tiempo (Schulz, 1999; Molina & Del Moral, 2005). A menudo, el elevado coste de estos censos y la falta de interés de las administraciones dificultan la continuidad de este tipo de seguimientos a largo plazo (Loss *et al.*, 2015). Es importante resaltar la ausencia de participación de España en el censo mundial de 2014 (Thomsen *et al.*, 2017). Por ello, viene siendo habitual en los últimos años que se requiera de la participación ciudadana desinteresada en los programas de seguimiento a gran escala (Greenwood, 2007; Tulloch *et al.*, 2013).

La popularidad de esta especie, unido a su ubicuidad y fácil identificación, hace que sea una excelente candidata para proyectos de ciencia ciudadana (Tulloch *et al.*, 2013; Frigerio *et al.*, 2018). No obstante, presenta una serie de limitaciones e inconvenientes, que en ningún caso parece que pueda sustituir la labor de los profesionales (Frigerio *et al.*, 2018). Estos últimos son necesarios en la formación previa de las personas que participen en la toma de datos y en la supervisión de los mismos, además de la revisión de los resultados, especialmente aquellos más dudosos (Loss *et al.*, 2015; Frigerio *et al.*, 2018). A diferencia de otros proyectos de ciencia ciudadana, los datos obtenidos en el censo de 2021 (Anexo I de esta tesis), así como en censos anteriores en la región de Madrid, han sido recogidos mayoritariamente por personas con formación y/o amplia experiencia en la realización de este tipo de censos, lo que sustenta la calidad de los datos. La difusión de la información recabada mediante voluntarios o ciencia ciudadana debería ser una prioridad. Es por ello que en el Anexo de esta tesis se incluye los resultados del Censo de la población reproductora de cigüeña blanca en la Comunidad de Madrid (Anexo I).

Igualmente, el desarrollo de las nuevas tecnologías ha supuesto un gran avance en la recogida y almacenamiento de la información (Sullivan *et al.*, 2009). Programas específicos como ZamiaDroid (http://biodiver.bio.ub.es/zamiaDroid/index.jsp), usado para recoger gran parte de los datos del censo de 2021 (Anexo I), facilitan la toma y procesamiento de los datos mientras que el uso de repositorios permite almacenar y compartir dicha información.

Mientras que estos censos han permitido analizar los cambios a nivel poblacional producidos por los vertederos en un intervalo prolongado de tiempo, el análisis de captura-recaptura con datos de anillamiento científico ha permitido determinar el impacto del vertedero a nivel individual (Figura 1). Aunque el marcado mediante anillamiento científico surge originalmente como una herramienta para estudiar los patrones migratorios (Bairlein, 2003), ha demostrado ser muy útil en otros muchos campos, al permitir la individualización de los miembros de una población a un bajo coste (Arizaga *et al.*, 2021).

Puesto que este sistema de marcaje no está exento de riesgo para las aves, requiere de un propósito que justifique claramente el impacto directo sobre los organismos a estudiar (Arizaga *et al.*, 2021). En el caso de especies de gran tamaño y difíciles de recapturar, como las cigüeñas, el anillamiento científico oficial (consistente en la colocación de una anilla metálica con un código alfanumérico y un remite indicando el país de origen) por sí solo carece de sentido. En estas especies las anillas oficiales deberían ser sustituidas o complementadas por otras marcas fácilmente identificables a distancia que posibilitan la identificación del individuo sin interferencia con sus procesos naturales, como en el caso de las ELSA ("European Laser-Signed Advanced") o al menos ir siempre acompañadas de marcas de lectura a distancia u otros elementos como radioemisores o dispositivos GPS, que permitan el seguimiento de los individuos tras abandonar el nido (Martínez-Miranzo *et al.*, 2016; Galbraith *et al.*, 2017). Lejos de ser metodologías excluyentes, anillas y dispositivos GPS se complementan en diversas áreas de estudio. Mientras que los dispositivos GPS permiten un nivel de detalle mucho mayor y un seguimiento continuo del individuo, su vida útil es mucho menor que las anillas de lectura a distancia (Fiedler, 2009; Thorup *et al.*, 2014). Estas últimas permiten un seguimiento a largo plazo, como el realizado en el Capítulo 3, esencial en especies de vida larga (Figura 3). El inconveniente de este sistema es que requiere de un gran esfuerzo de muestreo, puesto que los análisis de captura-recaptura están limitados al número de observaciones (recapturas) obtenidas y al potencial sesgo de los puntos de observación (Choquet *et al.*, 2013; Tourani, 2022). Una vez más, los programas de ciencia ciudadana pueden ayudar a elaborar una red de observadores que sigan una metodología similar.

Con el objetivo de mantener una metodología estandarizada en la identificación de los adultos y de los juveniles para los análisis de captura-recaptura, se han realizado visitas periódicas y regulares al vertedero de Colmenar Viejo y a una de las colonias (localizada entre los municipios de Soto del Real y Manzanares el Real). La lectura de las anillas se ha realizado mediante observación directa con prismáticos o telescopio terrestre, o con cámaras de fototrampeo situadas en los nidos. Además, se ha recopilado información de las observaciones realizadas por personas anónimas o colaboradores habituales.

Con el fin de centralizar y relacionar todas las observaciones realizadas, en el pasado y en la actualidad, se ha elaborado una base de datos, en la que se ha eliminado los duplicados y se ha generado un identificador único para cada individuo. Esta base de datos cuenta con más de 49.000 observaciones y cerca de 11.000 datos de anillamiento.

Fig. 3. Cigüeña marcada con anilla de lectura a distancia alimentándose en vertedero. En España, estas anillas constan de un código alfa-numérico de 4 caracteres. La foto corresponde al individuo F323. Foto original de David González del Portillo.

CHAPTER 1 - CHANGES IN WHITE STORK (*CICONIA CICONIA*) DISTRIBUTION IN THE LAST FOUR DECADES ASSOCIATED WITH LANDFILLS IN MADRID, SPAIN

This chapter is based on the manuscript: López-García, A., Aguirre, J.I. Impact of landfills on density and distribution of the White stork (*Ciconia ciconia*) in the last four decades in Central Spain. *Under review*.

Abstract

Human-induced environmental changes are the main drivers of the ongoing redistribution of biodiversity. The availability of millions of tonnes of organic waste that is added daily to landfills can increase the carrying capacity of ecological systems with direct effects on species´ population size and/or distribution. Understanding the effect of landfill on bird distribution is essential to assess management decisions. In this study, we found that landfills have altered the distribution of the breeding white stork (*Ciconia ciconia*) population over the last four decades in the province of Madrid, Spain. These results emphasise the impact of landfills on the population dynamics of bird species. We found that birds occupied new nesting sites near landfills independently of habitat quality. Nest density was higher near landfills and increased after the landfill began to be utilised by this species. Population growth and extremely high breeding densities may translate into conflicts with humans, particularly when new nesting sites are in urban areas, and possibly alter the perception of this bird species by the human population. The mandatory implementation of the European Landfill legislation is an opportunity to reduce the effects of landfills on animal populations, and reduce human-wildlife conflicts. However, there must be a process of transition and a preliminary evaluation of habitat quality and suitability in the region to avoid a dramatic decline of the white stork population.

Resumen

Los cambios inducidos por los seres humanos son los principales factores subyacentes de la redistribución de la biodiversidad en la Tierra. La disponibilidad de millones de toneladas de basura orgánica dispuesta diariamente en los vertederos puede desembocar en un incremento de la capacidad de carga de los ecosistemas con repercusiones directas en el tamaño y/o distribución de la población. Comprender el efecto de los vertederos en la distribución de las aves es esencial para evaluar las decisiones en la gestión de la fauna. En este estudio, se ha encontrado que los vertederos han alterado la distribución de la población reproductora de cigüeña blanca *(Ciconia ciconia)* a lo largo de las últimas cuatro décadas en la región de Madrid, España. Estos resultados enfatizan el impacto de los vertederos en la dinámica de poblaciones de las aves. Se observó que los reproductores ocupaban nuevos lugares de nidificación cerca de los vertederos independientemente de la calidad del hábitat. La densidad de los nidos fue mayor cerca de los vertederos y se incrementó tras el uso del vertedero por parte de esta especie en la región. El crecimiento poblacional y la extremadamente elevada densidad de parejas reproductoras puede dar lugar a conflictos con los seres humanos, particularmente cuando los lugares de nidificación se localizan en áreas urbanas, cambiando posiblemente la percepción de las personas sobre esta especie. La implementación obligatoria de la legislación europea sobre vertederos podría ser una oportunidad para reducir los potenciales efectos negativos de los vertederos en las poblaciones animales y además reducir los conflictos entre la fauna y los seres humanos.

Sin embargo, debería existir un proceso de transición, así como una evaluación previa de la calidad e idoneidad del hábitat en la región para evitar una dramática disminución de la población de cigüeña blanca.

Introduction

Humans have deeply modified the size and distribution of populations of species around the world (Boivin *et al.*, 2016; Bar-On *et al.*, 2018). Habitat fragmentation, overexploitation of natural resources, land-use change, and an increasing human global pollution are behind dramatic population declines and shrinking distributions of many species (Dirzo *et al.*, 2014; Ceballos *et al.*, 2015). However, some anthropogenic activities have triggered a spectacular population explosion of a small group of species. In addition to land-use transformation and climate change, predictable anthropogenic food subsidies (PAFS) alter population dynamics by increasing food availability in a particular location (Oro *et al.*, 2013). Landfills are the main unintentional PAFS in terrestrial ecosystems. The continuous and abundant generation of organic waste attracts the attention of several species (Plaza & Lambertucci, 2017). Animals cover long distances and modify their movement patterns to reach these PAFS (Soriano-Redondo *et al.*, 2021; Spelt *et al.*, 2021). Migratory birds use these facilities as important stop-over sites or will shorten their migrations both in time and distance (Gilbert *et al.*, 2016; Arizaga *et al.*, 2018; Cheng *et al.*, 2019). Ensuring food availability also encourages opportunistic bird species to breed near landfills (Duhem *et al.*, 2008; Monsarrat *et al.*, 2013; Tauler-Ametller *et al.*, 2017). During the breeding season, foraging at a predictable anthropogenic resource reduces foraging costs associated with seeking food and increases parental care of offspring (Moritzi *et al.*, 2001; Soriano-Redondo *et al.*, 2021). For instance, foraging at landfills enhances clutch size, egg size, body condition, and survival of offspring (Tortosa *et al.*, 2002; Steigerwald *et al.*, 2015; Djerdali *et al.*, 2016a; Pineda-Pampliega *et al.*, 2020). The increase in breeding success and survival may promote population growth that results in a higher density of individuals, an expansion of population range, or both. In addition, an elevated aggregation of breeders may lead to an increase in intra-specific competition (Gilchrist & Otali, 2002; Denac, 2006; Djerdali *et al.*, 2016b) and conflicts with humans (Belant, 1997). This human-induced increase in population numbers is often considered to be a negative impact in the so-called overabundant species (Bino *et al.*, 2010; Payo-Payo *et al.*, 2015), but positive in endangered species (Tauler-Ametller *et al.*, 2017; Plaza & Lambertucci, 2018). Thus, understanding the effects of landfills on population dynamics is essential to accurately address the management and conservation of both endangered and overabundant species.

White storks (*Ciconia ciconia*) have been known to utilise landfills in the Iberian Peninsula since the 1980s (Blanco, 1996; Tortosa *et al.*, 2002), when the south-western breeding population reached the minimum number of registered pairs after a period of constant decline (Bernis, 1981; Barlein, 1991). First, they were observed wintering or roosting in the surrounding areas (Blanco, 1996; Archaux *et al.*, 2004). As the white stork breeding population continued to grow, this growth was associated with the use of landfills and implementation of conservation measures (Schulz, 1999; Tortosa *et al.*, 2002; Massemin-Challet *et al.*, 2006). Although many studies have analysed the positive effect of landfills on population size (which include: an increase in juvenile survival by shortening migration (Flack *et al.*, 2016; Rotics *et al.*, 2017; Cheng *et al.*, 2019), improvement to body condition (Pineda-Pampliega *et al.*, 2021), and changes to reproductive parameters (Tortosa *et al.*, 2002; Massemin-Challet *et al.*, 2006; López-García *et al.*, 2021), few studies have explored the effects of landfills to the distribution of the breeding population and the occupation of new sites (Bialas *et al.*, 2020). Furthermore, in the case of species with a marked nest-site fidelity (Barbraud *et al.*, 1999; Vergara *et al.*, 2006) and philopatry (Del Hoyo *et al.*, 2014), such as the white stork, previous distribution is a determinant factor in the future expansion of the species. Furthermore, it is possible that the most breeding suitable areas or feeding areas overlap the new anthropogenic food resource, just by chance. However, the studies focused on the effects of landfills on white stork distribution often overlooked the potential effects of historical distribution due to the lack of available data before the use of the new feeding resources began (Djerdali *et al.*, 2016b; Bialas *et al.*, 2020; Hmamouchi *et al.*, 2020b,a). One of the strong points of this study is that we took into account the historical distribution of the breeding population.

Our main objective was to evaluate the effects of landfills on the distribution of the growing population of white storks over the last 40 years. Our first purpose was to analyse which variables were involved in the probability of nest occupation before the use of landfills. We then evaluated the effects of landfills on the aggregability of this species, controlling for other potential confounding variables. Our final aim was to characterise the expansion process by identifying the differences between previous nesting sites and the new nesting sites. We hypothesized that landfill attracted white stork pairs which breed nearer these facilities independently of the habitat quality. Moreover, we propose that the abundance of food provided by landfills support higher densities of breeders.

Materials and methods

Study area and data collection

We surveyed the entire province of Madrid (8030 km^2), Spain during the 2021 breeding season searching for white stork occupied nests (López-García & Aguirre, In press). The areas in the north and west of this region are characterised by mountain ranges, forest, and pastures usually devoted to cattle. The eastern area of the region is characterised by arable lands and crop fields. The south is similar to the east, but with more mixed habitats, water, and permanent crops associated with rivers (Cuevas, 2003). Madrid city has around 3.5 million inhabitants and is located at the centre of this region. The white stork has never bred in the south-eastern area of Madrid (Molina & Del Moral, 2005).

To evaluate the effects of landfills on the white stork distribution, we also compiled data on nest occupancy from two previous censuses in 1984 (Lázaro *et al.*, 1986) and 2004 (Molina & Del Moral, 2005). The nest location accuracy in 1984 was less precise, as some locations were based on description only, and not on GPS locations. Because the precision error was between tens of hundreds of meters in 1984, we uniformized resolution across sampling periods to a 1-km grid. All the censuses in our study area were based on the same methodology of direct assessment that was used in all the White stork European censuses (Schulz, 1999; Aguirre & Vergara, 2009). This involves three visits to detect the occupancy of nests between March to June or, at least one visit in May. A nest was considered as an occupied nest when we observed an adult, a pair or fledglings on the surface of the nest.

In the Madrid region, white storks were first observed feeding on the landfill in the 1980s (unpublished data and Chozas 1983, Blanco 1996). The 4 main landfills in this region were first opened in the 1980s and are active in 2021. The use of these four landfills

(Alcalá de Henares, 40.457619 N, 3.36308 W; Colmenar Viejo, 40.664071N, 3.72797W; Pinto, 40.257118N, 3.63755W; and Valdemingomez also named Las Dehesas, 40.336427N, 3.590375W) by white storks was confirmed in situ by Manuel Fernández Cruz and the authors (See localization in Fig. S1). Two smaller landfills (Nueva Rendija and Colmenar de Oreja), were closed at the beginning of 2000. They were not included in our analysis due to the absence of observations of white stork feeding on these sites. Therefore, the distribution of nests in 1984 was defined as the distribution of the breeding population in the region before white storks started using the landfills.

It was not possible to identify individual nest re-occupation between the censuses due to the time elapsed between censuses and differences in the location accuracy (Barbraud *et al.*, 1999). Instead, we divided the occupied nests into two variables to explore the direction of the expansion of the breeding population in relation to the spatial distribution of the nests. The nearest occupied nest to a nest recorded in the previous census (at maximum distance of 1 km) was considered as a remained nest (0). We defined *new nesting sites* as all other nests (1). Therefore, new nesting sites represent areas of population expansion.

To evaluate the effects of landfills and other environmental variables on the probability of a breeding area being abandoned between censuses, we defined an analogous binomial variable, following Bialas *et al.* 2020: either the nest was occupied in one census and also occupied in the next census (remained nests, 0), versus the nest was occupied in the first census but unoccupied in the following census (1). This variable was called *abandoned nesting sites*.

Both of these variables explore changes in occupancy in two periods of time 1984-2004 and 2004-2021.

Spatial analysis

We selected the following environmental variables known to affect the distribution of breeding white stork populations (Carrascal *et al.*, 1993; Radović *et al.*, 2015; Orłowski *et al.*, 2019; Hmamouchi *et al.*, 2020b; Bialas *et al.*, 2021): percentage of land cover by arable lands (CLC class 21), percentage of land cover by other agricultural areas (CLC class 22 and 24, except 244), percentage of land cover by forestry areas (CLC class 31 and 32, except 321), percentage of land cover by pastures, meadows & dehesas (*i.e.,* Agro-forestry areas with a particular management in the Mediterranean area) (CLC class 231, 321 and 244), distance to the nearest water bodies, and degree of urbanization. Because of the collinearity with the rest of variables, we did not include the percentage of water bodies (CLC class 41 and 51) or percentage of highly altered urban areas (CLC class 1).

These land cover types were obtained by grouping the original Corine Land Cover (CLC) classes based on previous accumulated knowledge on this species, following the methodology used in Bialas *et al.* 2021. Information about land cover was extracted from the Corine Land Cover vector dataset (https://centrodedescargas.cnig.es). Then, we calculated the percentage of each land cover type in each grid cell of 1x1 km.

We obtained the dataset of the degree of urbanization in a resolution of 30 arc-seconds (equivalent to 1x1 km) from the first hierarchical level of the Global Human Settlement Layer - Settlement Model Grid (GHSL-SMOD) from the Socioeconomic Data and Applications Center (https://sedac.ciesin.columbia.edu/data/set/ghsl-population-built-up-estimates-degree-urban-smod). This layer classifies each grid cell in rural, periurban, or urban areas, as a function of population density and the percentage of built-up areas (Florczyk *et al.*, 2019). We use this information as a representation of imperviousness soil and potential food scarcity in cities, and human perturbance.

Water bodies information were downloaded from the BTN100 database of Organismo Autónomo Centro Nacional de Información Geográfica (https://centrodedescargas.cnig.es).

Our grid resolution was constrained by our lowest accuracy of occupied nest location (1984 census). Therefore, we used a grid resolution of 1x1 km for the community of Madrid to extract the values of our environmental variables for each cell. We measured distance from centroids of each cell of the grid to the nearest landfill and nearest body of water using *st_distance* function in the *sf* package (Sumner *et al.*, 2022). To obtain the nest density, we counted the number of nests in each grid cell (nests per km^2).

We used the *randomPoints* function in the *dismo* package (Hijmans *et al.*, 2017) to generate the pseudoabsences within the area of Community of Madrid. We generated as many pseudoabsences as nests were found in 1984 (215 nests). Pseudoabsences were generated up to 1 km from each nest, avoiding to include the same variables of occupied grid cells.

Given that land cover and population density have changed over the last 40 years, we chose the layer temporally closest to each year of census data (*i.e.,* CLC 1990 to 1984 census, CLC 2006 to 2004 census, and 2018 CLC to 2021 census).

Spatial analyses were performed with QGIS 3.16.11 open-source software (QGIS Development Team, 2022) and 4.1.2 R version (R Core Team, 2020).

Statistical analysis

To assess whether the average distance from nests to the nearest landfill varied over the last 37 years, we built a Linear Model (LM) with a gaussian error structure, with distance to a landfill as the response variable and the year of the census (as factor) as predictor variables.

To identify which of several environmental variables determine the distribution of white stork breeding populations before landfills were used by this species in our study region, we used a Generalized Linear Model (GLM) with a binomial response and logit link function. We defined the occupied nests identified in the 1984 census as presences (1) and the random points generated in the study area as pseudoabsences (0). We include the percentage of other agricultural areas (Agricultural areas called as "Agri" hereinafter), percentage of arable lands (Arable lands), percentage of forest (Forest), percentage of pasture & dehesas (Pasture & Dehesas), degree of urbanization (Urbanization), and the distance to the nearest body of water (Water) as predictors.

We performed a GLM with a negative binomial error structure and log link function to evaluate the differences in nest density (nests/km^2) in relation to the distance to landfills and the census year. We also evaluated the interaction of year and landfill distance, as well as interaction of year and environmental variables (Agri, Arable lands, Forest, Pasture & Dehesas and Water) in this analysis. To control for their potential confounding effects, we include the followings predictor variables: Agri, Arable, Forest, Pasture & Dehesas, Urbanization, and Water. We also evaluated landfill distance and Pastures & Dehesas as a quadratic term.

In order to consider terms which may not have a linear relationship with the outcomes in our GLMs, we also include quadratic term in the GLMs for both the probability of nest occupation and nest density. Although we tried all the different combination of quadratic terms for all the continuous variables (Agri, Arable, Forest, Pasture & Dehesas and Water), only the ones that improved the Akaike's Information Criterion corrected (ΔAICc) in the model selection process were included. Thus supports its inclusion in the final model (Table S1), avoiding a high increase of multicollinearity.

To analyse the relationships between distance to landfills and various environmental variables on new nesting sites and abandoned nesting sites we built GLMs with binomial error structures and logit link functions. We built separate models for each period of time 1984-2004 (2004) and 2004-2021; named 2004 and 2021 for the new nesting site and 1984, and 2004 for the abandoned nesting sites respectively. Beside distance to landfill, the following predictor variables were included in the analysis: Agri, Arable, Forest, Pasture & Dehesas, Urbanization, and Water.

We used the Akaike's Information Criterion corrected by sample size (AICc) for model selection (Burnham & Anderson, 2002). We generated models with all possible combinations of our predictor variables and ranked them based on the differences between the AICc of a given model and the AICc of the model with the lowest AICc (ΔAICc). We interpreted results of the model with the lowest AICc which was considered to be the best model. When there were two or more equally plausible models with less than 2 units of AICc of difference, we applied conditional model averaging among these equivalent models using the *model.avg* function in the *MuMIn* package (Barton, 2022).

We conducted pairwise comparisons with Tukey post hoc tests to explore differences in urbanization degree or years. We report the results of the post-hoc analysis only when the Analysis of Variance (ANOVA) tests used first were significant.

Collinearity was evaluated using variance inflation factor (VIF) statistics. Multicollinearity was acceptable in all our analysis (VIF < 3).

All statical analysis were performed with R version 4.1.2 (R Core Team, 2020).

Results

The mean distance of occupied nests to the nearest landfill was 17.515 ± 0.893 km in 1984, 15.096 ± 0.314 km in 2004 and 14.063 ± 0.220 km in 2021.

On average, nests were significantly closer to the landfills in 2021 than in 2004 (Tukey HSD test, Estimate ± SE=2.42 ± 0.81, T-student=3.45, p= 0.007) and 1984 (Tukey HSD test, Estimate ± SE=3.45 ± 0.78, T-student = 4.44, p<0.001). Similarly, we found that nests were closer to landfills in 2004 than in 1984 (Tukey HSD test, Estimate ± SE = 1.03 ± 0.39, T-student=1.03, p= 0.02).

According to the model selection the following variables predicted the nest occupation in 1984: a low percentage of agricultural areas, arable lands, and forest areas; more than 25% coverage of pastures & dehesas; and proximity to water bodies (Table 1). The probability of nesting in 1984 in rural areas was significantly higher than in urban areas (Tukey HSD test, Estimate ± SE=1.34 ± 0.55, Wald=2.427, p= 0.040) (Fig.1).

Table 1. Output from the GLM for probability of nest occupation before white storks began to use landfills (1984). Values for the predictors: percentage of other agricultural fields (Agri), arable lands (Arable), forest (Forest), pasture and dehesas (PD), degree of urbanization (Rural, Periurban or Urban), and distance to water bodies (Water). Estimates (Est), standard errors (SE) and 95% confidence intervals (CI) were calculated by model averaging for models with a ΔAIC < 2.

	Est	**SE**	**Wald**	**p-value**	**2.5%CI**	**97.5%CI**
Agri	-1.492	0.468	3.188	0.001***	-2.410	-0.575
Arable	-0.906	0.452	2.010	0.045*	-1.792	-0.021
Forest	-3.342	0.658	5.081	<0.001***	-4.630	-2.053
PD	2.475	1.058	2.338	0.019*	0.400	4.549
PD^2	-2.606	0.965	2.702	0.007**	-4.497	-0.716
Rural	0.653	0.327	1.990	0.046*	0.011	1.295
Periurban	0.298	0.230	1.300	0.195	-0.153	0.750
Water	-1.872	0.51	3.672	<0.001***	-2.871	-0.873

Note: Significance level: ***p<0.001, **p<0.01, *p<0.05.

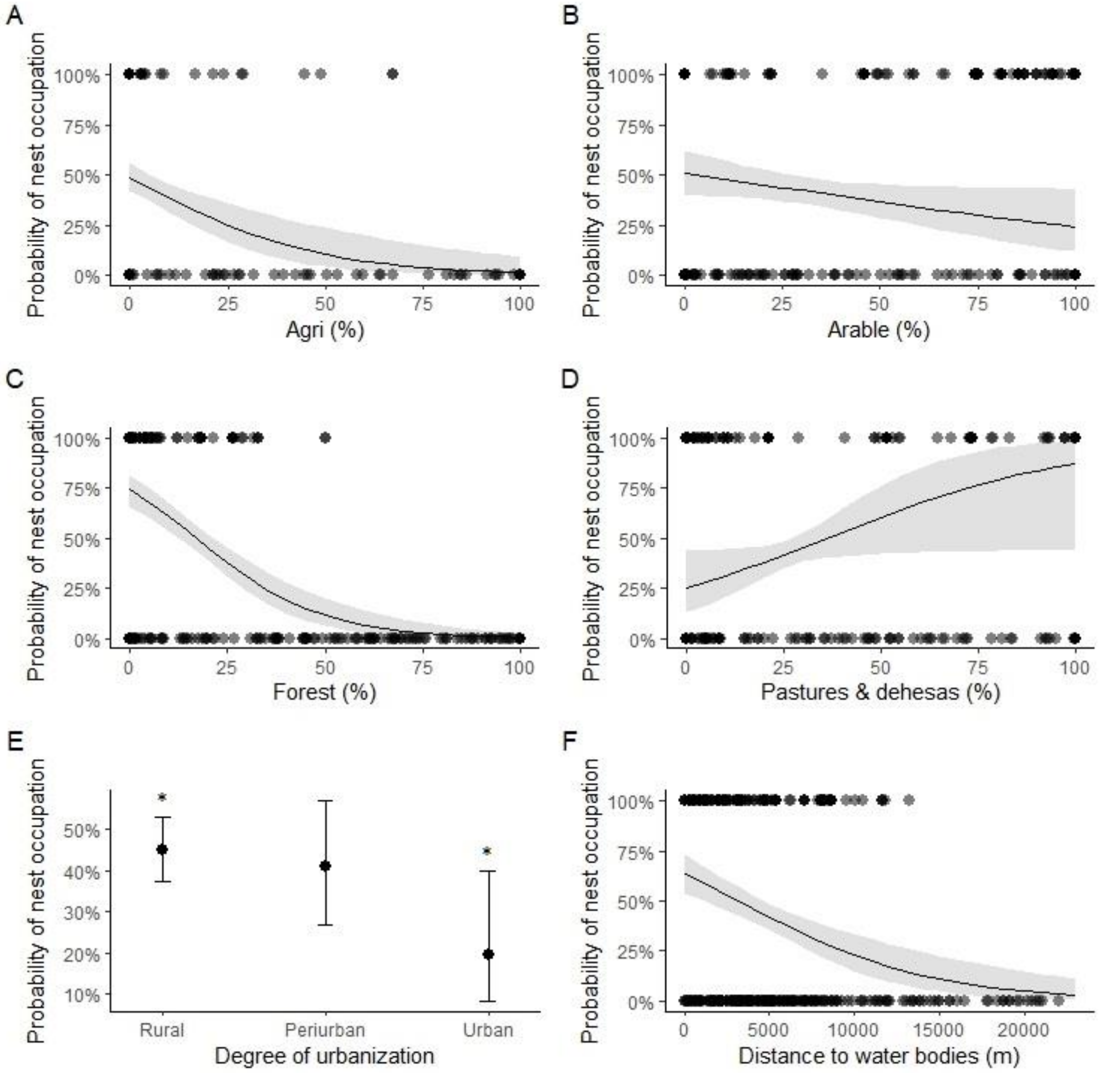


Fig. 1 The probability of nest occupation in relation to the percentage of other agricultural fields (A), arable lands (B), forest (C), pastures & dehesas (D), degree of urbanization (E) and, distance to water bodies (F). Shading is the 95% confidence intervals. In the degree of urbanization (E), the whiskers show the 95% CI and the asterisk designates the groups with significant differences.

Landfill distance negatively affected nest density depending on the year of the census (Table 2). The nest density increased near landfills in 2004 and 2021 compared to 1984 (both Tukey HSD test, p<0.001) (Fig. 2). Moreover, nest density had a negative relationship with the percent coverage of agricultural fields and arable lands, and increased near water bodies (Table 2). In addition, nest density was lower in periurban areas than in rural areas (Tukey HSD test, Estimate=-0.52 ± 0.14, Wald =3.746, p<0.001) (Table 2, Fig. 3).

Table 2. Nest density relationship to the landfill distance, controlling by percentage of other agricultural fields (Agri), arable lands (Arable), forest (Forest), pasture and dehesas (PD), degree of urbanization (Rural, Periurban or Urban), and distance to water bodies (Water). Estimates (Est), Standard Errors (SE) and 95% confidence intervals (CI) were calculated by model averaging of models with a ΔAIC < 2.

	Est	**SE**	**Wald**	**p value**	**2.5%CI**	**97.5%CI**
Agri	-0.011	0.004	2.626	0.009**	-0.019	-0.003
Arable	-0.011	0.005	2.107	0.035*	-0.021	-0.001
Landfill	-0.067	0.011	6.097	<0.001***	-0.089	-0.046
Landfill^2	0.030	0.011	2.671	0.008**	0.0079	0.051
PD	0.007	0.006	1.181	0.237	-0.004	0.018
Rural	0.014	0.005	3.059	0.002**	0.005	0.023
Periurban	-0.010	0.003	2.944	0.003**	-0.017	-0.003
Water	-0.017	0.004	4.452	<0.001***	-0.025	-0.010
1984	-0.041	0.013	3.098	0.002**	-0.068	-0.015
2004	0.011	0.013	0.853	0.394	-0.014	0.036
Arable*1984	0.010	0.007	1.403	0.161	-0.004	0.023
Arable*2004	0.004	0.006	0.655	0.513	-0.008	0.015
Landfill*1984	0.019	0.011	1.734	0.083	-0.002	0.040
Landfill*2004	-0.022	0.010	2.196	0.028**	-0.043	-0.002
PD*1984	0.004	0.009	0.481	0.631	-0.014	0.023
PD*2004	0.014	0.008	1.859	0.063	-0.001	0.030
Forest	0.003	0.003	0.905	0.365	-0.004	0.011

Note: Significance level: ***p<0.001, **p<0.01, *p<0.05.

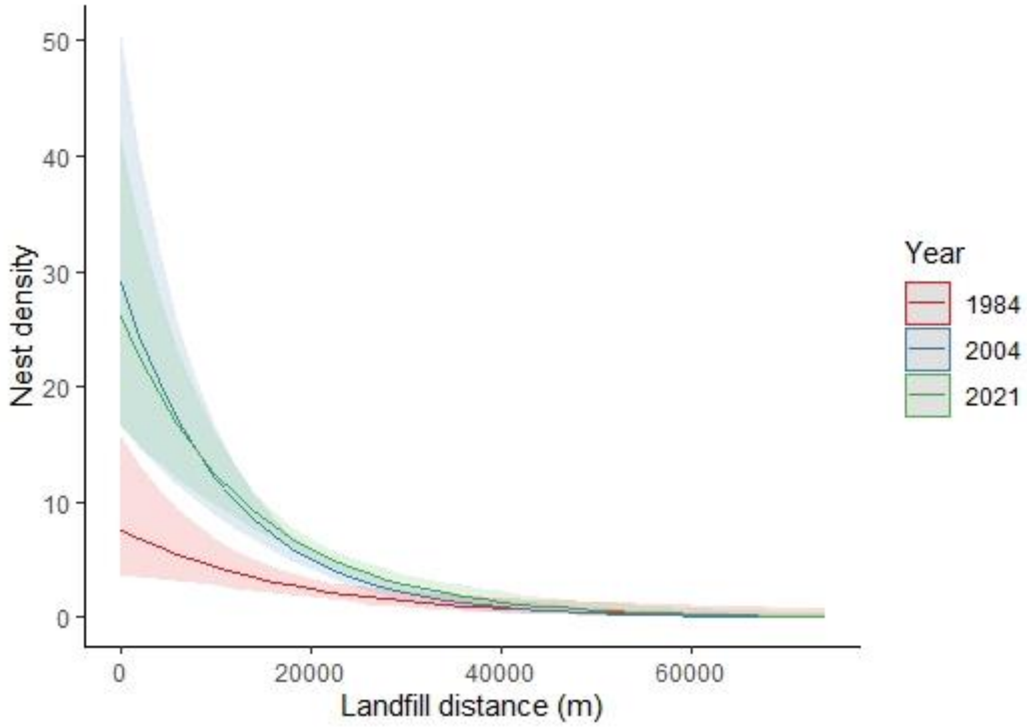


Fig. 2. Effect of landfill distance on nest density. Nest density is higher in 2021 and 2004 than in 1984, before white storks began utilizing landfills. Shaded areas represent the 95% confidence intervals.

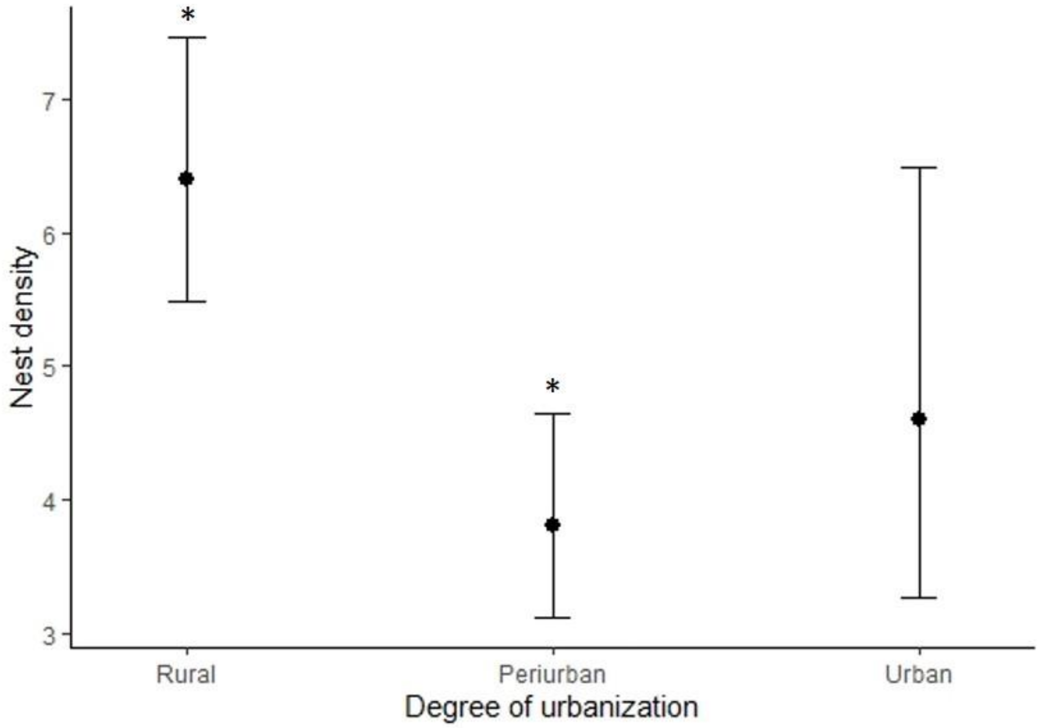


Fig. 3. Differences in nest density (nests/km^2) as a function of degree of urbanization (with 95% confidence intervals). Significant differences are marked with asterisks (*).

Model selection of new nesting sites in 2004 resulted in two equally good models (AICc<2) (Table S1). Selected models include all variables and quadratic terms for pastures & dehesas. Both models included agricultural areas, forest areas, landfill distance, pastures & dehesa, degree of urbanization, and distance to water bodies (Table 3). In 2004, *new nesting sites* were closer to landfills and in areas with a higher percentage of: agricultural coverage, forest, and pastures & dehesas, and closer in proximities to water bodies (Table 3, Fig. 4). Moreover, *new nesting sites* were ubicated in a higher proportion in urban areas than in rural areas (Tukey HSD test, Estimate=-1.12 ± 0.39, Wald = -2.899, p<0.011) (Fig. 4).

In 2021, *new nesting sites* were significantly further away from landfills and water bodies than *abandoned nesting sites*, and in areas with a higher proportion of arable lands (Table 3) (Fig. 4).

Table 3. Differences amongst permanent breeding areas to new nesting sites and abandoned nesting sites. Values for the predictors are: percentage of other agricultural fields (Agri), arable lands (Arable), forest (Forest), pasture and dehesas (PD), Landfill distance (Landfill), degree of urbanization (Rural, Periurban or Urban), and distance to water bodies (Water. Estimates (Est), standard error (SE) and 95% confidence intervals (CI) were calculated by model averaging of models with a ΔAIC < 2.

	Est	**SE**	**Wald**	**p value**	**2.5%CI**	**97.5%CI**
New nesting sites 2004						
Agri	0.824	0.344	2.397	0.017*	0.150	1.498
Forest	1.027	0.271	3.785	<0.001***	0.495	1.559
Landfill	-0.100	0.255	3.926	<0.001***	-1.499	-0.500
PD	1.016	0.343	2.963	0.003**	0.344	1.688
Rural	-1.028	0.338	3.04	0.002**	-1.690	-0.365
Periurban	0.005	0.260	0.018	0.986	-0.504	0.514
Water	-1.100	0.294	3.746	<0.001***	-1.675	-0.524
Arable	0.062	0.201	0.308	0.758	-0.428	0.832
New nesting sites 2021						
Arable	0.654	0.102	6.391	<0.001***	0.453	0.854
Landfill	0.293	0.100	2.915	0.004**	0.096	0.489
Rural	0.153	0.111	1.376	0.169	-0.065	0.371
Periurban	0.840	0.096	8.754	<0.001***	0.652	1.028

Water	1.591	0.120	13.223	<0.001***	1.355	1.827
Agri	-0.021	0.058	0.359	0.720	-0.265	0.094
PD	-0.008	0.053	0.157	0.875	-0.284	0.186
Forest	0.003	0.039	0.088	0.930	-0.166	0.209
Abandoned nesting sites 1984						
Agri	1.235	0.456	2.709	0.007**	0.341	2.129
Arable	1.196	0.923	1.296	0.195	-0.153	3.049
Forest	-0.825	0.923	0.894	0.371	-2.970	0.368
Landfill	0.578	0.751	0.769	0.442	-0.340	2.482
PD	0.148	0.546	0.271	0.787	-1.255	2.824
Water	0.050	0.277	0.18	0.857	-0.950	1.775
Abandoned nesting sites 2004						
Agri	0.219	0.283	0.776	0.438	-0.081	0.935
Arable	2.756	0.474	5.814	<0.001***	1.827	3.685
Forest	0.839	0.280	2.999	0.003**	0.291	1.387
Landfill	0.875	0.332	2.637	0.008**	0.225	1.525
PD	0.573	0.660	0.868	0.385	-0.242	2.131
Rural	0.498	0.494	1.007	0.314	-0.470	1.466
Periurban	-2.162	0.499	4.330	<0.001***	-3.140	-1.183
Water	-0.384	0.431	0.889	0.374	-1.390	0.133

Note: significance level: ***p<0.001, **p<0.01, *p<0.05.

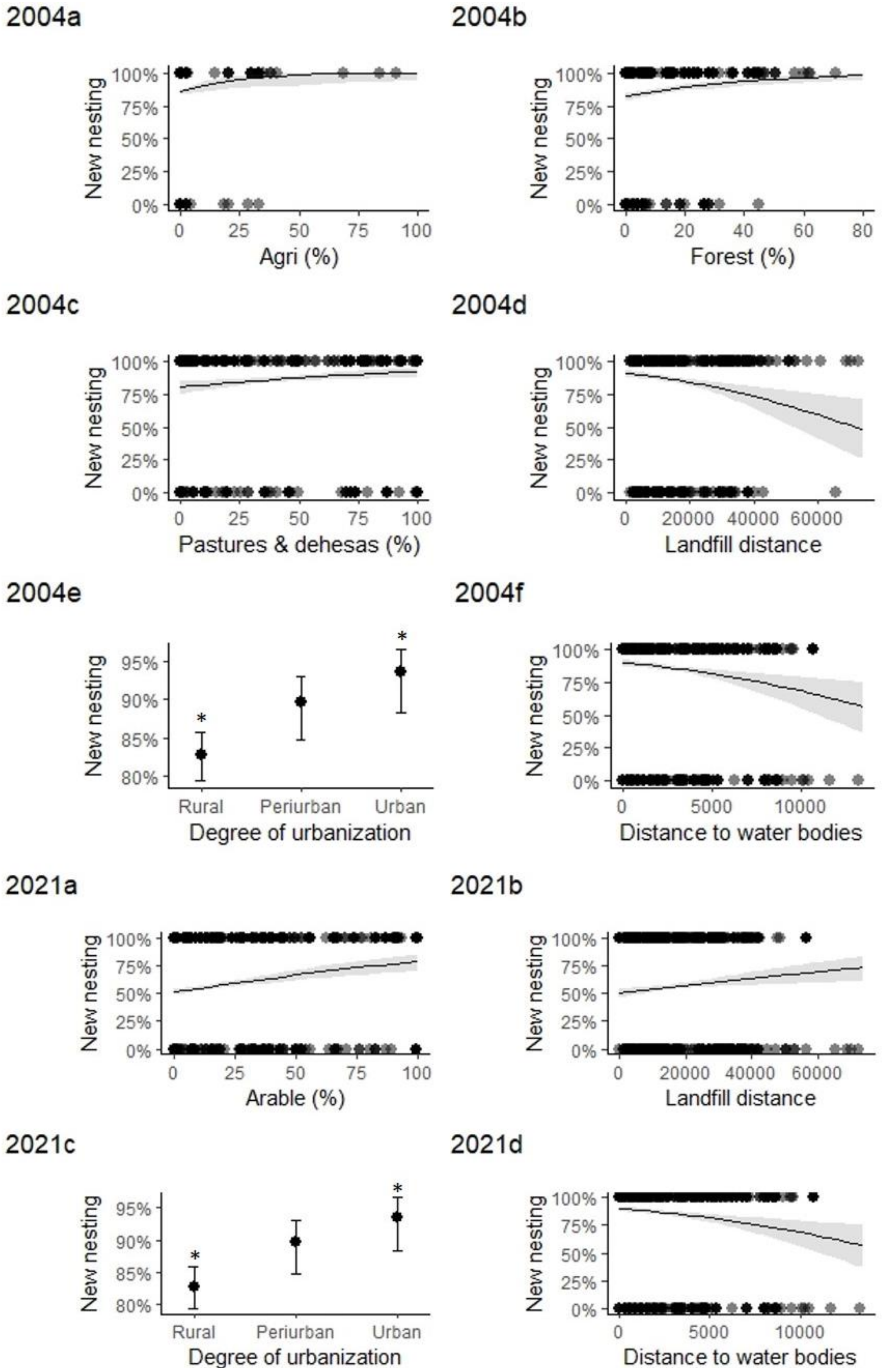


Fig. 4. The relationship between the probability of new nesting sites in 2004 and 2021 and the percentage of other agricultural fields (Agri), percentage of arable lands (Arable), percentage of forest (Forest), percentage of pastures & dehesas (Pastures & dehesas), distance to the nearest landfill (Landfill distance), degree of urbanization, and distance to water bodies. In general, the probability of new nesting sites was higher in 2004 than in 2021. Shaded areas show the 95% confidence intervals. In the degree of urbanization plot, whiskers show the 95% CI and asterisks represent significant differences. We only show significant differences.

In the period between 1984-2004, proximity to areas with a higher percentage of agricultural land increased the abandoned nesting sites of that area (Est = 1.235 ± 0.453, Wald = 2.709, p = 0.007), but this relationship was not found for any of the other variables (p>0.05) (Table 3). However, the abandoned nesting sites during the period of 2004-2021 was correlated positively with the distance to landfills (Fig. 5). abandoned nesting sites in this period also increased with the percentage of arable lands, forest, and in urban areas (Fig. 5, Table 3).

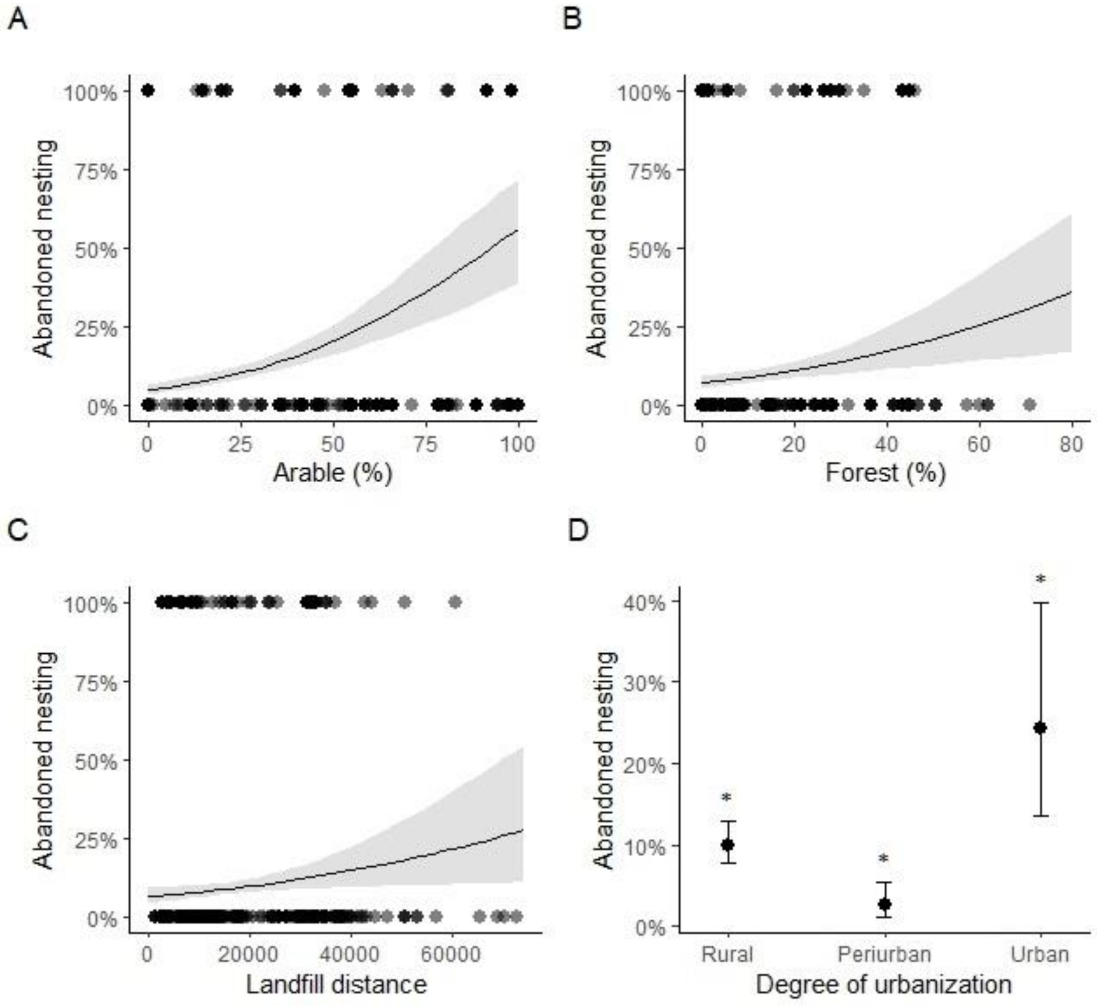


Fig. 5. The relationship between sites in 2004 and the percentage of other arable lands (A), percentage of forest (B), distance to the nearest landfill (C), and degree of urbanization (D). Shaded areas represent the 95% confidence intervals. Asterisks indicate significant differences.

Discussion

This study found that the increase of the breeding population of white storks was associated with the exploitation of landfills as feeding resources, which has also been found in other species (Duhem *et al.*, 2008; Tauler-Ametller *et al.*, 2017). The distribution of the expanding breeding population, as a consequence of the population growth, has not been random. After landfill exploitation began, pairs gradually breed near these facilities in spite of poor-quality habitat such as agricultural fields, forests or urban areas.

As seen in other species (Belant *et al.*, 1998; Monsarrat *et al.*, 2013; Tauler-Ametller *et al.*, 2017), white storks are attracted by the large amount of organic waste and the constant renewal of resources in landfills. White storks not only breed near landfills, but, since 1984, they are also more aggregated around these facilities. These almost unlimited anthropogenic food resources reduce the dependency for prey availability (Payo-Payo *et al.*, 2015; Evans & Gawlik, 2020), so it seems to reduce conflicts on feeding resources and enhances intra-specific tolerance (Restani *et al.*, 2001; Corman *et al.*, 2016), allowing individual aggregability that often results in colony formation in the Mediterranean region (Carrascal *et al.*, 1990; Massemin-Challet *et al.*, 2006; Djerdali *et al.*, 2016b). Thus, the probability and number of interactions between individuals are exponentially elevated in a colony as well as in places where food is concentrated in a limited area (*i.e.,* landfills or vulture restaurants). Since pathogens proliferate in landfills and aggregation of individuals enhances infectious disease transmission (Bradley & Altizer, 2007; Moyers *et al.*, 2018), the proliferation of potential avian foraging species associated with these facilities raises public health issues (Hatch, 1996; Navarro *et al.*, 2019; Höfle *et al.*, 2020). The increase of nesting density in periurban and urban areas in the last four decades (Fig. 4) also could result in greater wildlife-human conflicts, such as nuisance by noise or dirtiness, and damage to buildings and other human structures (Belant, 1997; Vergara *et al.*, 2007a; Zbyryt *et al.*, 2021). This situation may change the human perception of this species from an iconic bird to a pest species (Belant, 1997; Belant *et al.*, 1998).

The high nest density and low abandoned nesting sites in rural areas indicate that typically white storks avoid areas with high human densities and a high percentage of land cover transformation to impervious soil. In fact, an elevated degree of urbanization and human population density imply higher physiological stress (Ellenberg *et al.*, 2007; Strasser & Heath, 2013) and risks for breeding success such as collision with electricity poles, use of deterrents, or nest removal (Garrido & Fernandéz-Cruz, 2003; Moreira *et al.*, 2018; Marcelino *et al.*, 2021). Moreover, the highest nest densities in rural areas were possibly associated with proximity to landfills in 2004 and 2021, or possibly because of farms and zoos in 1984, where human feeding supply or the presence of cows increase feeding rates (Tryjanowski *et al.*, 2005; Massemin-Challet *et al.*, 2006; Hilgartner *et al.*, 2014; Zbyryt *et al.*, 2020). In contrast, two opposite ecological forces underlie the wide range of nest density that we found in urban areas. Breeding pairs avoid urban areas because of the nuisance to the nestlings and the scarcity of natural feeding resources (Strasser & Heath, 2013; Seress *et al.*, 2020). At the same time that the proximity to a landfill allows birds to occupy nests in urban areas, but limited to certain sites such as the highest structures (*i.e.,* churches, antennas, electricity poles) which increase nest density at particular buildings or sites (Bialas *et al.*, 2020).

According to previous studies, this species preferentially selects areas with pastures and dehesas, near water bodies and in rural areas while avoiding agricultural fields, arable lands, forests and urban areas (Alonso *et al.*, 1991; Carrascal *et al.*, 1993; Zurell *et al.*, 2018; Hmamouchi *et al.*, 2020b). However, in this study we found that nest density was not associated with the percentage of pastures and dehesas. Maybe, this habitat is a key factor to determine presence or absence of occupied nests but not their density, at least at the spatial resolution utilized here.

In the period between 1984-2004, birds primarily expanded to areas close to landfills independently of the habitat quality. Besides landfill distance, it is possible that several strategies converge to determine the occupation of new nesting sites in 2004. On the one hand, the astonishing population increase in this period led to a process of expansion to high-quality areas, with a high percentage of pastures and dehesas and near water bodies, while breeding pairs disappeared from areas with a high percentage of agricultural usage (Fig. 4). On the other hand, inexperienced young pairs might have been forced to breed in poor-quality areas in this expanding population, which resulted in a high percentage of occupied nests in other agricultural fields, forests, and in urban areas (Newton, 1992; Vergara & Aguirre, 2006). Nonetheless, future research should explore this question further.

The expansion process continued in the period between 2004-2021. After the occupation of the most favourable areas near landfills, it seems that new nesting sites were located in equivalent or sub-optimal quality patches in this period (Fig S1, Table 3). These areas further from landfills and water bodies with a high percentage of arable lands are possibly being utilized by younger breeders or low-quality individuals (Newton, 1992; Vergara & Aguirre, 2006). Arable lands may constitute an alternative and sub-optimal feeding area in recent decades due to habitat degradation, which has been supported in previous studies (Orłowski *et al.*, 2019; Bialas *et al.*, 2021). In agreement with these previous studies, we found that birds abandoned suboptimal areas with a higher percentage of arable lands, forests, urban areas, and areas far away from landfills in 2004.

Nevertheless, our findings show that the new food resource at landfills allows white storks to colonize new and previously unsuitable areas. However, previous research has indicated that the breeding success of these pairs is lower (López-García *et al.* in press, Bialas *et al.* 2021), and the extra food from landfills primarily increases the density of breeders in the surrounding areas.

In conclusion, the use of landfills by white storks enhances the population growth and has played an important role in the distribution of this species in the Community of Madrid. The changes in the management of landfills following the European directive (Directive 1999/31/EC and 2018/850/EC) provide an opportunity to reduce nearby nest density, thus reducing potential wildlife-human conflicts (Belant, 1997) as well as reducing potential risks to wildlife associated with an intensive use of landfills (Plaza & Lambertucci, 2018; López-García *et al.*, 2021). However, this should be a gradual process and managers must evaluate the nearby habitat quality and possibly support expanding suitable habitat in the region. Otherwise, white storks might in future decades be in the same critical situation as they were in 1950-1980.

Supplementary material

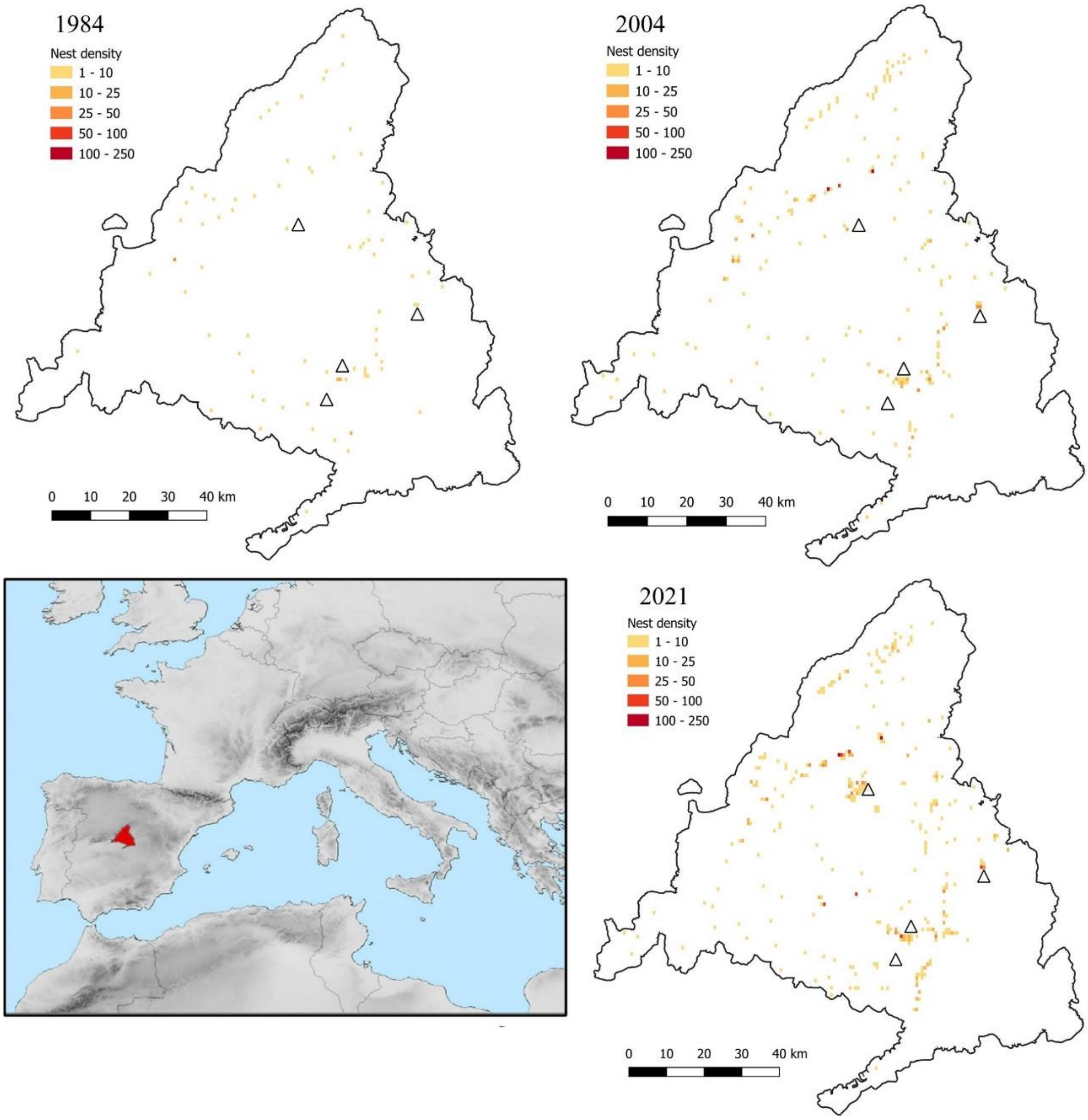


Fig. S1. Changes on nest density (nests/ km^2) and distribution of the breeding population of white storks in the province of Madrid. The location of landfills is marked by white triangles. The grid resolution is 1km^2. Map of Mediterranean area with our study region (Madrid) highlighted in red (in the left down corner).

Table S1. Model selection based on the Akaike Information Criterion adjusted for small sample sizes (AICc) in probability of nest occupation before the use of landfills (1984), nest density, new nesting sites and abandoned nesting sites. Models in bold (within 2 AICc units of the best model) were used in the model averaging. We include additive (+) and interaction (∗) effects. Squared terms were only included in nest-site selection and nest density.

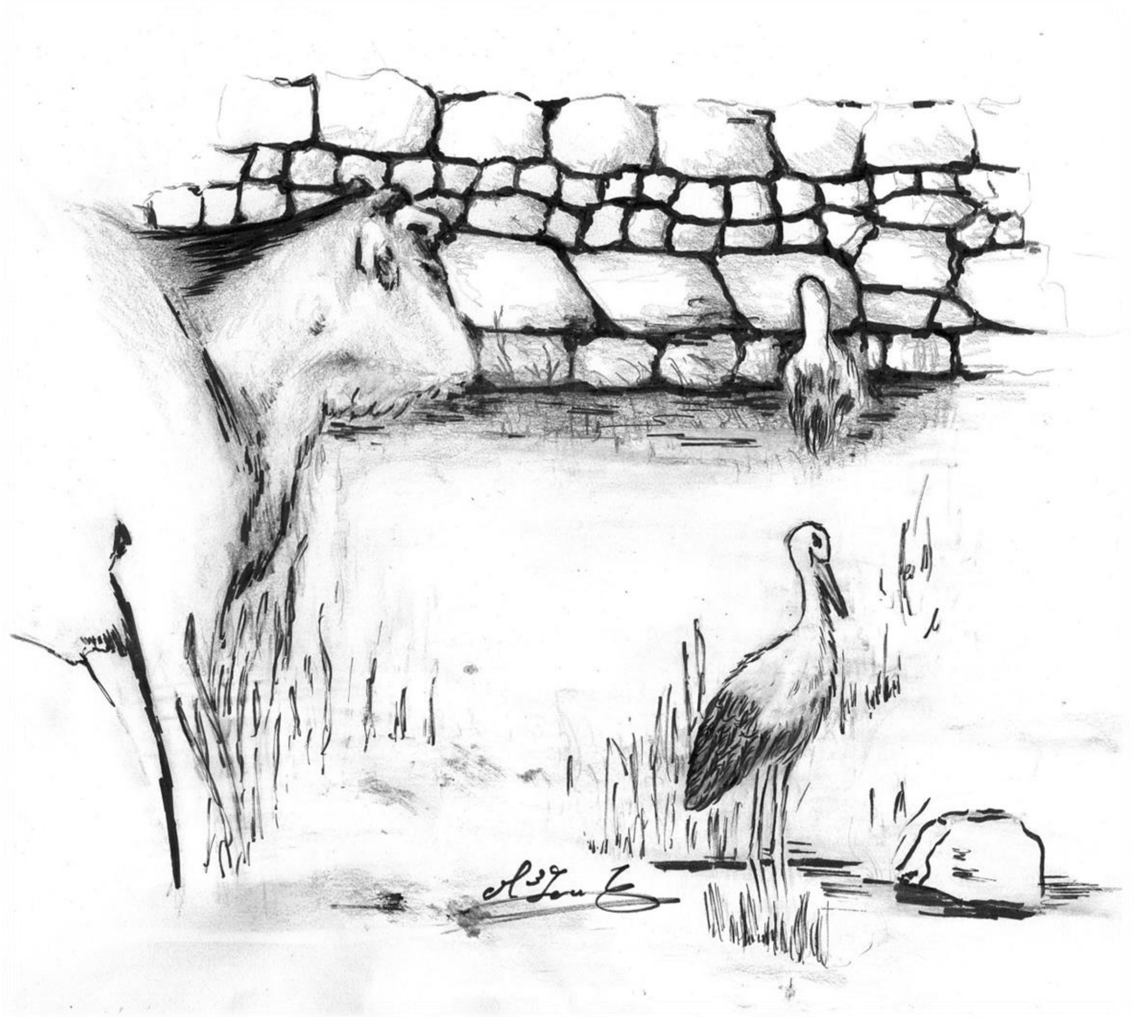
Model	**df**	**logLik**	**AICc**	**ΔAICc**	**w**
Probability of nest occupation 1984					
Agri+Arable+Forest+PD+PD^2+Urban+Water	**9**	**-210.52**	**439.47**	**0**	**0.295**

Agri+Forest+PD+PD^2+Water	**6**	**-213.79**	**439.79**	**0.316**	**0.252**
Agri+Arable+Forest+PD+PD^2+Water	7	-213.62	441.51	2.043	0.106
Agri+Forest+PD+PD^2+Urban+Water	8	-212.63	441.6	2.131	0.102
Agri+Arable+Forest+Urban+Water	7	-214.65	443.57	4.103	0.038
Agri+Arable+Forest+PD+Urban+Water	8	-214.3	444.93	5.462	0.019
Agri+Forest+Water	4	-218.44	444.98	5.509	0.019
Agri+Forest+PD+Water	5	-217.47	445.07	5.602	0.018
Agri+Forest+Urban+Water	6	-216.58	445.36	5.888	0.016
Agri+Arable+Forest+Water	5	-217.73	445.59	6.123	0.014
Nest density					
Agri+Urban+Water+ (Arable+PD+Landfill)*year+Landfill^2	**18**	**-1544.47**	**3126.21**	**0**	**0.178**
Agri+Urban+Water+Landfill+ (Arable+PD)*year+Landfill^2	**16**	**-1546.87**	**3126.76**	**0.549**	**0.135**
Agri+Forest+Urban+Water+ (Arable+PD+Landfill)*year+Landfill^2	**19**	**-1544.1**	**3127.61**	**1.402**	**0.088**
Agri+Forest+Urban+Water+Landfill+ (Arable+PD)*year+Landfill^2	**17**	**-1546.44**	**3128.02**	**1.811**	**0.072**
Agri+Urban+Water+Arable+ (PD+Landfill)*year+Landfill^2	**16**	**-1547.56**	**3128.12**	**1.915**	**0.068**
Agri+Urban+Water+Arable+Landfill+ PD*year+Landfill^2	**14**	**-1550.2**	**3129.17**	**2.962**	**0.041**
Agri+Forest+Urban+Water+Arable+ (PD+Landfill)*year+Landfill^2	17	-1547.11	3129.35	3.147	0.037
Agri+Urban+Water+Arable+Landfill +year+Landfill^2	11	-1553.81	3130.1	3.891	0.025
Urban+Water+ (Arable+PD+Landfill)*year+Landfill^2	17	-1547.5	3130.14	3.93	0.025
Agri+Forest+Urban+Water+Arable+Landfill+ PD*year+Landfill^2	15	-1549.66	3130.2	3.993	0.024
New nesting sites 2004					
Agri+Forest+Landfill+PD+Urban+Water	**8**	**-496.04**	**1008.2**	**0**	**0.594**
Agri+Arable+Forest+Landfill+PD+Urban+Water	**9**	**-495.84**	**1009.83**	**1.631**	**0.263**
Agri+Forest+Landfill+PD+Water	6	-501.09	1014.25	6.045	0.029
Forest+Landfill+PD+Urban+Water	7	-500.35	1014.79	6.591	0.022
Agri+Arable+Forest+Landfill+PD+Water	7	-500.56	1015.22	7.019	0.018
Agri+Arable+Forest+Landfill+Water	6	-501.84	1015.74	7.542	0.014
Agri+Arable+Forest+Landfill+Urban+Water	8	-499.84	1015.8	7.601	0.013
Agri+Forest+Landfill+Urban+Water	7	-501.27	1016.64	8.441	0.009
Arable+Forest+Landfill+PD+Urban+Water	8	-500.35	1016.82	8.617	0.008
Arable+Forest+Landfill	5	-504.01	1018.07	9.866	0.004

New nesting sites 2021					
Arable+Landfill+Urban+Water	**6**	**-1445.97**	**2903.97**	**0**	**0.317**
Agri+Arable+Landfill+Urban+Water	**7**	**-1445.52**	**2905.09**	**1.116**	**0.182**
Arable+Landfill+PD+Urban+Water	**7**	**-1445.88**	**2905.81**	**1.846**	**0.126**
Arable+Forest+Landfill+Urban+Water	**7**	**-1445.94**	**2905.93**	**1.961**	**0.119**
Agri+Arable+Landfill+PD+Urban+Water	8	-1445.37	2906.81	2.841	0.077
Agri+Arable+Forest+Landfill+Urban+Water	8	-1445.51	2907.08	3.108	0.067
Arable+Forest+Landfill+PD+Urban+Water	8	-1445.88	2907.82	3.852	0.046
Agri+Arable+Forest+Landfill+PD+Urban+Water	9	-1445.37	2908.82	4.853	0.028
Arable+Urban+Water	5	-1450.32	2910.67	6.705	0.011
Arable+PD+Urban+Water	6	-1449.66	2911.36	7.387	0.008
Abandoned nesting sites 1984					
Agri+Arable+Forest+Landfill	**5**	**-75.52**	**161.32**	0	0.074
Agri+Arable+Landfill	**4**	**-76.79**	**161.78**	**0.455**	**0.059**
Agri+Arable	**3**	**-77.92**	**161.95**	0.633	0.054
Agri+Arable+Forest	**4**	**-76.91**	**162**	0.681	0.053
Agri+Forest	**3**	**-78.02**	**162.16**	0.838	0.049
Agri+Forest+Landfill	**4**	**-77.4**	**163**	1.679	0.032
Agri+Arable+Forest+Landfill+PD	**6**	**-75.39**	**163.17**	1.854	0.029
Agri+Arable+Forest+PD	**5**	**-76.45**	**163.18**	1.859	0.029
Agri+Arable+PD	**4**	**-77.51**	**163.2**	1.88	0.029
Agri+Arable+Forest+Landfill+Water	**6**	**-75.42**	**163.25**	1.925	0.028
Agri+Arable+Landfill+Water	**5**	**-76.5**	**163.28**	1.962	0.028
Agri+Arable+Landfill+PD	5	-76.66	163.61	2.292	0.023
Agri+Forest+Urban	5	-76.7	163.69	2.368	0.023
Abandoned nesting sites 2004					
Agri+Arable+Forest+Landfill+PD+Urban	**8**	**-399.22**	**814.56**	**0**	**0.159**
Agri+Arable+Forest+Landfill+PD+Urban+Water	**9**	**-398.22**	**814.6**	**0.036**	**0.156**
Arable+Forest+Landfill+Urban+Water	**7**	**-400.32**	**814.74**	**0.175**	**0.145**
Agri+Arable+Forest+Landfill+Urban+Water	**8**	**-399.52**	**815.16**	**0.596**	**0.118**
Arable+Forest+Landfill+PD+Urban	**7**	**-400.68**	**815.46**	**0.898**	**0.101**
Arable+Forest+Landfill+PD+Urban+Water	**8**	**-399.74**	**815.61**	**1.044**	**0.094**
Arable+Forest+Landfill+Urban	**6**	**-402.09**	**816.26**	**1.698**	**0.068**
Agri+Arable+Forest+Landfill+Urban	7	-401.65	817.39	2.829	0.039
Agri+Arable+Forest+PD+Urban	7	-401.87	817.82	3.263	0.031
Agri+Arable+Forest+PD+Urban+Water	8	-400.89	817.91	3.346	0.03

Note: terms are **df,** degrees of freedom; **AICc,** Akaike information criterion corrected for small sample size; **ΔAICc,** the AICc difference between the current model and the one with the lowest AICc value; **w,** Akaike weights; **Agri,** percentage of other agricultural fields; **Arable,** percentage of arable fields; **Forest,** percentage of land cover by forest; **Landfill,** Landfill distance; **PD,** percentage of pastures & dehesas; **Urban,** degree of urbanization; **Water,** distance to the nearest body water.

CHAPTER 2 - INFLUENCE OF LANDFILL USE ON BREEDING NEST-SITE SELECTION AND BREEDING SUCCESS IN WHITE STORKS (*CICONIA CICONIA*)



This chapter is based on the manuscript: López-García, A., Martínez-Miranzo, B. & Aguirre J.I. *Under review*. Influence of landfill use on nest-site selection and breeding success in white storks (*Ciconia ciconia*).

Abstract

Landscape transformation by humans through habitat degradation, agriculture intensification, and urbanization results in the loss of natural feeding areas. However, populations of certain species have adapted to these changes and benefit from some anthropogenic food sources as well as an absence of predators and milder environmental conditions in urban landscapes. Although breeding near landfills secures food availability and saves energy and time on foraging activities, the poor food quality and risks associated with these facilities imply some fitness costs.

Our study shows the effects of landfills on nest-site selection and breeding success in breeding white storks. Birds breeding near landfills occupy more urban areas where many structures are available to nest, while breeding pairs 20-30 km from landfills select nest-sites with high quality food sources in the vicinity such as pastures, meadows, and agro-forestry areas. Furthermore, we demonstrate that habitat quality in the surrounding area is more relevant than landfills on breeding outcome, but both of them are crucial for brood size. The number of fledglings was higher at a medium distance to the landfill than at 20-30 km from the landfill, but not near the landfill (0-10km). This suggest that the best strategy includes food from landfills as a complementary or alternative food source. Future studies should further investigate if these shifts in habitat preference are permanent, and the potential impact of habitat degradation and landfill closures at a population level, for this species.

Resumen

La transformación del paisaje por los seres humanos mediante la degradación de los hábitats, la intensificación de la agricultura y la urbanización, ha dado lugar a la pérdida de las áreas naturales de alimentación de muchas especies. Sin embargo, las poblaciones de determinadas especies se han adaptado a estos cambios y se benefician de algunas fuentes de alimentación antrópicas, así como, de la ausencia de depredadores y el ambiente templado de los entornos urbanos. Aunque criar cerca de los vertederos asegura la disponibilidad de alimento y el ahorro de energía y tiempo en las actividades de forrajeo, la peor calidad del alimento y los riesgos asociados con el uso de estas instalaciones implica algunos costes en la eficacia biológica de los individuos.

Nuestro estudio muestra los efectos de los vertederos en la selección del lugar de nidificación y el éxito reproductor de las cigüeñas blancas. Las aves que nidifican cerca de los vertederos ocupan más áreas urbanas, dónde abundan las estructuras disponibles para nidificar, mientras que las parejas que crían a 20-30 km de los vertederos seleccionan lugares de nidificación con fuentes de alimentación de alta calidad en los alrededores, tales como pastos, prados húmedos y dehesas. Además, se ha mostrado que la calidad del hábitat en los alrededores del nido es más relevante en el éxito reproductor que la distancia a los vertederos, pero ambas variables son cruciales en el número de pollos. El número de volantones fue mayor a una distancia media de los vertederos que a 20-30 km de dichas instalaciones, pero no cerca de ellos (0-10 km). Esto sugiere que la mejor estrategia incluye alimentarse en los vertederos como fuente de alimentación complementaria o alternativa. Estudios futuros deberían investigar en profundidad si estos cambios en las preferencias de la cigüeña blanca son permanentes y el impacto potencial de la degradación del hábitat y el cierre de los vertederos a nivel poblacional.

Introduction

Since the origin of agriculture, over 10,000 years ago, human activities have transformed major parts of the Earth's surface (Lewis & Maslin, 2015; Venter *et al.*, 2016). The increasing and unstoppable human pressure has led to habitat degradation through agricultural intensification and urbanization with consequences for animal populations, such as loss of resources leading to loss of individual fitness, population declines or displacement, and species extinctions (Boivin *et al.*, 2016; Venter *et al.*, 2016; Ceballos *et al.*, 2017; Powers & Jetz, 2019).

However, new food subsidies from anthropogenic development have emerged in parallel with this reduction in the availability of natural food sources. Many species have adapted to these changes or have exploited the human landscape transformations to their benefit (Blair, 1996; Møller, 2009). In environments altered by humans, Predictable Anthropogenic Food Subsidies (PAFS) constitute a wide, extensive, abundant, and predictable food source with several benefits at population and individual levels (Oro *et al.*, 2013; Plaza & Lambertucci, 2017). With millions of tonnes of human refuses disposed each year (Stenmarck *et al.*, 2016), landfills are one of the most important PAFS in terrestrial ecosystems, and have been shown to increase populations of both pests and endangered species (Payo-Payo *et al.*, 2015; Tauler-Ametller *et al.*, 2017). In contrast to other animals, birds can easily access food refuse at landfills, avoiding perimetral fences, machinery, and workers. In fact, more than half of the animal species that have been reported feeding at these human facilities are birds (Plaza & Lambertucci, 2017).

The use of this food source implies saving time and energy costs associated with foraging activities (Gilbert *et al.*, 2016; Soriano-Redondo *et al.*, 2021). This is particularly important during the breeding season when nest construction and nestling rearing increases both the energetic costs and food intake of breeders (Ricklefs, 1973; Bryant, 1988; Tinbergen & Dietz, 1994). A reduction in foraging time allows more time for breeders to cover other offspring requirements (Moritzi *et al.*, 2001), increasing fertility and offspring survival during the nestling period (Kilpi & Ost, 1998; Annett & Pierotti, 1999; Auman *et al.*, 2008; Steigerwald *et al.*, 2015).

Although landfills secure food availability during the breeding season (Martin, 1987; Massemin-Challet *et al.*, 2006; Newsome *et al.*, 2015), the attractiveness of these facilities increases local population densities, resulting in higher intra-specific competition for nesting place availability (Gilchrist & Otali, 2002; Djerdali *et al.*, 2016b). As a consequence, the availability of nesting sites or predation avoidance, but not the availability of natural foraging areas in the proximities of the nest, may become the most relevant limiting factors in nest-site selection (Newton, 2007). Moreover, feeding on human refuse has been associated with poor food resource quality (Grémillet *et al.*, 2008; Murray *et al.*, 2018) and may imply an increase in feeding rates.

In accordance with the Optimal Foraging Theory (Schoener, 1971), the benefits of feeding at landfills during breeding season (*i.e.,* saving time and energy) decrease when distance to landfill increases because incubation and nestling rearing limit the daily movements of breeders. In addition, the availability and the quality of food sources in the surrounding area is crucial for nest-site selection and reproductive output when birds breed far away from landfills (Sergio & Newton, 2003; Lambrechts *et al.*, 2004; Orłowski *et al.*, 2019). In fact, foraging at landfills as well as the probability of attendance to landfills in white storks decrease when the distance to these facilities increases (Gilbert *et al.*, 2016; Soriano-Redondo *et al.*, 2021).

Therefore, birds breeding near landfills may prioritize nest-site locations with high availability of nesting structures or low predation, while breeding pairs far away from landfills need sites with high availability of natural foraging areas in the vicinity of the nest.

White storks represent an excellent animal model for exploring these issues. This generalist bird traditionally breeds in areas surrounded by mixed land cover with a high proportion of foraging habitats, such as pastures, meadows, and agro-forestry areas (also called “Dehesas” in the Mediterranena region), avoiding agricultural areas and crop fields (Alonso *et al.*, 1991; Carrascal *et al.*, 1993; Zurell *et al.*, 2018; Orłowski *et al.*, 2019). The intensification of agriculture and subsequent habitat degradation of these natural foraging habitats were thought to be the main cause of the sharp population decline in the western populations between the 1950’s to the 1980’s (Chozas, 1983; Barlein, 1991;

Carrascal *et al.*, 1993; Schulz, 1999). However, the increasing use of landfills and the legal protection measures enacted at the beginning of the 90s were responsible for the population growth in recent decades (Blanco, 1996; Tortosa *et al.*, 2002; Massemin-Challet *et al.*, 2006).

In addition, this species exhibits central-place foraging behaviour from the nest, so the habitat in the surrounding area can be considered representative of the availability of "natural" feeding areas during breeding season with a reasonable cost-benefit ratio in terms of feeding sources (Zurell *et al.*, 2015, 2018). The return rate to their nests to feed from "natural" food sources and take care of the brood has limited the average trip distance to less than 2 km (Alonso *et al.*, 1991; Johst *et al.*, 2001; Moritzi *et al.*, 2001; Nowakowski, 2003; Zurell *et al.*, 2018). Given the abundance of food in landfills, white storks have been shown to travel distances further than 20 km to reach landfills, though the probability of visits significantly decreased beyond a 10-15 km distance (Gilbert *et al.*, 2016; Soriano-Redondo *et al.*, 2021).

Understanding the impact of landfills on nest-site selection during the breeding season and among individuals gives us a chance to disentangle the underlying processes in the population dynamics of species linked to these facilities and reach conservation and management goals. Therefore, our aim was to determine the effects of landfills on nest-site selection and how this impacts breeding success. We hypothesized that breeding pairs will differ in habitat preference in the surrounding areas with different distances to landfills. Breeding pairs nesting close to landfills will be less dependent on "natural" feeding areas in the vicinity of their nest, while pairs breeding far away from landfills will have a higher preference for *natural* feeding habitats and a strong avoidance of poor quality habitats (areas which have neither nesting supports nor food sources).

Materials and methods

Study area

Our study area covers the potential area of influence of the landfills in the Comunidad de Madrid in the centre of the Iberian Peninsula (Fig. 1). In this region, the first landfill following EU regulations (75/442/CEE) was opened in the 1980s. Afterwards, the increasing human population and the implementation of new legislation allowed for the establishment of up to 6 landfills in this region. White storks were observed feeding on landfills for the first time in this region at the beginning of the 1990s (Blanco, 1996; Tortosa *et al.*, 2002). We included the 5 landfills where storks were observed feeding (Alcalá de Henares, Colmenar Viejo, Nueva Rendija, Pinto and Valdemingomez; Fig. 1).

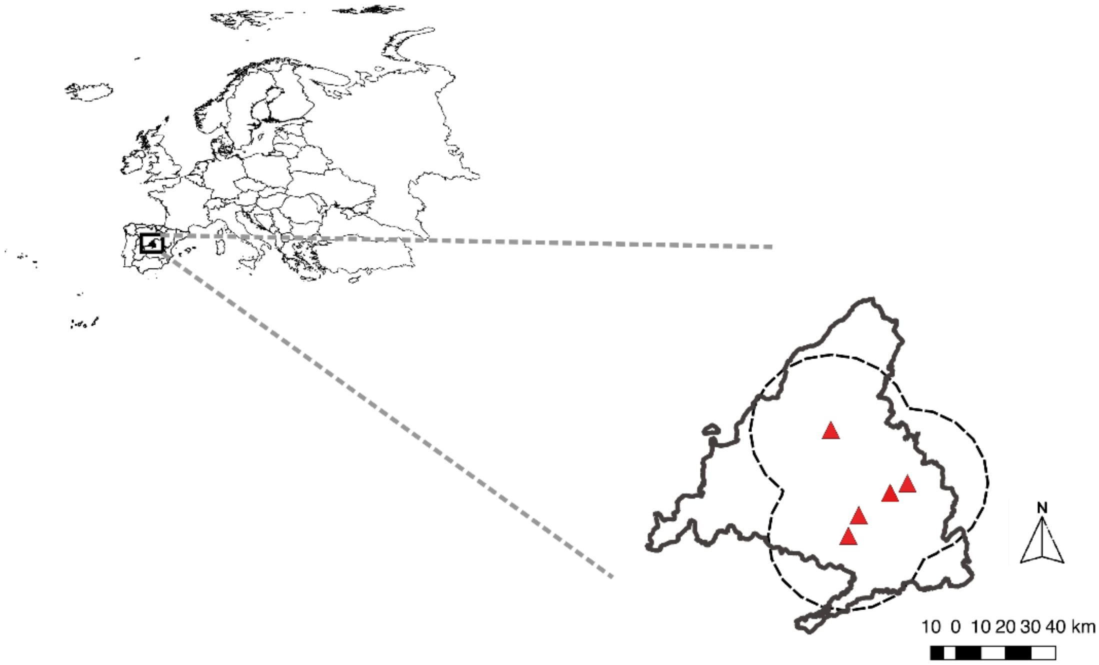


Fig. 1. The map representing the limits of the Madrid region (black solid line), the buffer of 30 km around each landfill (dashed line), and the locations of the landfills (triangles).

Data collection

Based on the distance that the stork population in the Iberian Peninsula is moving away from their nests (Bécares *et al.* 2019), we monitored a buffer of 30 km from each landfill in 2021 and compiled information on the locations of nests occupied during the 2001 (Aguirre & Atienza, 2002) and 2004 breeding censuses (Molina & Del Moral, 2005). A nest observed at least once with an adult white stork in March or later in the season, was considered as occupied (Aguirre & Atienza, 2002; Aguirre & Vergara, 2009). In 2001 and 2004, nests from the same colony (Schulz, 1999) were given the same coordinates due to the absence of wide use of Geographical Positioning System (GPS) before 2004. We recorded 4040 nest locations in total to estimate the Nest-site Ranking Index and Nest-site selection (To detached information see Table S2). A substantial number of these nests were monitored each year during the breeding season (n=2557), from March to June, with at least 1 visit at the end of May or the beginning of June to estimate the number of fledglings per nest.

Spatial analysis and nest-site selection index

We extracted habitat information from the Corine Land Cover database (https://centrodedescargas.cnig.es) for the area surrounding each monitored nest. We pooled the initial 33 CORINE Land Cover classes in our study area into 7 groups (Table S1), known to be potentially important for white storks, based on the literature on this species (Alonso *et al.*, 1991; Carrascal *et al.*, 1993; Radović *et al.*, 2015; Zurell *et al.*, 2018; Orłowski *et al.*, 2019; Bialas *et al.*, 2020, 2021): urban areas (also known as areas greatly altered by humans (Bialas *et al.*, 2021), Urban); arable land (Arable); other agricultural land (Agri); pastures, meadows & "dehesas" (Pastures & Dehesas); forestry areas (Forest); non-suitable habitat (Non-suitable); and inland waters (Inland water). We selected the Corine Land Cover layer temporally closest to the year of each census (*i.e.,* CLC 2000 to 2001 census, CLC 2006 to 2004 census, and 2018 CLC to 2021 census). To manage the changes in land cover as well as the number of landfills during the extended timeline of this study, we performed our spatial analysis for each year independently.

We calculated the percentage of each land cover group in a 2 km buffer around each nest as the potential habitat selected during breeding season by each breeding pair (Bialas *et al.*, 2020, 2021). In order to perform nest-site selection and nest-site preference analysis, we considered the proportion of each type of habitat in the entire study area defined by a 30 km buffer from each landfill as available habitat, following previous studies (Aebischer *et al.*, 1993; Holt *et al.*, 2010; Sharps *et al.*, 2015). Then, we obtained a selection index based on selection ratios. For a particular habitat type, the selection ratio corresponds to the ratio between the proportion of the selected habitat type and the proportion of the available habitat type. The index indicates the likelihood that a specific habitat type would be used given its relative availability in the region (Manly *et al.*, 2002). Therefore, index values higher than 1 mean positive selection of that habitat type and values between 0 and 1 mean negative selection of that habitat type. For cases where the habitat value is zero (not available or no use), we used a number an order of magnitude smaller than the values for used habitat, as recommended in Aebischer *et al.* (1993). Habitat types named as Inland Water and Non-suitable were excluded from the rest of the analysis because their values were zero or < 1% for available habitat in all cases, and compositional analysis is not possible under such conditions.

Distance to the nearest landfill from each nest was calculated using the *Distance matrix* tool. Landfill distance can be used as a proxy for access to these anthropogenic food sources (Jagiello *et al.*, 2020; Pineda-Pampliega *et al.*, 2021). According to the potential intensity use of the landfill, we classified each nest into one of three landfill distance categories: easy access to landfill (0-10 km), access to landfill but with energetic costs (10-20 km), and nearly null or unlikely access to landfill (20-30 km). These categories were defined based on previous knowledge of this species' landfill usage during the breeding season: white storks significantly decrease visits to landfills when breeding up to 10-13 km away from a landfill (Gilbert *et al.*, 2016; Soriano-Redondo *et al.*, 2021), and they rarely reach distances greater than 27.7 km daily (Bécares *et al.*, 2019).

For spatial analyses, we used the QGIS 3.16.11 open-source software (QGIS Development Team, 2022).

Statistical analysis

We performed compositional analysis using the "compana" function of the *adehabitatHS* package (Calenge, 2020) to examine hierarchical nest-site preferences (Aebischer *et al.*, 1993). We set data randomization to 1000 iterations and alpha equal to 0.05. For graphical interpretation of the compositional analysis, we calculated a nest-site preference index as a result of the sum of the differences in log-ratios between habitats produced from the compositional analysis following the methodology of Holt *et al.* (2010).

To analyse differences in the nest-site selection index associated with the distance to landfills, and controlling for year, we performed general linear mixed model (GLMM) using the *lme4* package (Bates *et al.*, 2022), with a Gaussian error structure and the identity link function for *Urban*, *Forest*, and *Pastures & Dehesas*. For *Agri* and *Arable,* we used the Gamma error structure and log link function due to the data having a left skew as a consequence of the high number of values near zero in these variables.

With respect to breeding output, we distinguished between: a) breeding outcome, as a binomial variable, for nest occupied without fledglings (0) and nests observed with at least one fledgling (1), and b) the number of fledglings in nests with confirmed breeding success. We performed a general linear mixed model (GLMM) to assess the effects of landfill distance and nest-site selection on breeding outcomes (binomial error structure and logit link function), and the number of fledglings (Poisson error structure and log link function). The nest-site selection index was included in the breeding output models as a measure of habitat quality in the surrounding area. The index was transformed into a binomial factor, with values higher than 1 representing a higher proportion of a certain habitat than the habitat availability in the study area (Higher Proportion, 1) and values lower than 1 representing a lower proportion of a certain habitat than the habitat availability in the study area (Lower Proportion, 0). We included only the habitats positively selected by white storks in at least one distance to landfill category (*Urban*" and "*Pastures & Dehesas*") (Martínez-Miranzo *et al.*, 2016).

Since all the nests of the same colony are exposed to the same variables and can be related, we included colony and year as random factors in all models to avoid pseudo-replication.

The distribution of nests is not continuous in space and is usually grouped in colonies. Because of that, we avoided the use of a continuum distribution of distances in our statistical analysis.

Model selection was based on the Akaike's Information Criterion corrected for sample size (AICc) (Burnham & Anderson, 2002). Models with a ΔAICc < 2 points were considered equivalent (Burnham and Anderson, 2002) and we applied model averaging among them. We used the *model.avg* function in the *MuMIn* package to obtain conditional averages of variable estimates, standard errors (SE), and confidence intervals (Barton, 2022).

Pairwise comparisons with Tukey post-hoc tests were used to explore differences in landfill distance categories and years. We report the results of post-hoc analysis only when confidence intervals were significant.

Multicollinearity in our analysis was not excessive (VIF < 2).

All statistical analyses were performed with R Statistical Software (version 4.1.2; R Core Team 2020).

Results

Mean colony size was 4.33 nests in 2001, 5.28 nests in 2004, and 12.05 nests in 2021.

Nest-site Ranking Index

Nest-site preference differed far from random in our study area, independently of the distance to landfill range or census year (P > 0.001 for all compositional analysis; see Table S2).

In general, white storks showed a preference for *Urban* and *Pastures & Dehesas* habitats, and avoidance for *Arable* and *Agri* (Table S2) in the surrounding area when establishing their nest. While nest-site preference was similar for up to a 10 km distance to the landfill, habitat ranking near landfills (0-10 km) was almost completely different amongst years (Table S2). The compositional analysis indicated that avoidance for *Arable* increased with landfill distance in 2001 and 2004 but not in 2021 (Table S2).

Nest-site selection models

In general, model selection indicated that nest-site selection varied with landfill distance (Table 1). The model including the interaction between landfill distance and year was the best model for *Arable* habitat after model selection (Table 1). Breeding white storks showed the highest preference and selection for *Arable* habitat at 0-10 km in 2001 and 2004 (Table S2, Fig. 2, all Tukey HSD tests p < 0.001). Unlike the previous census, there was avoidance of *Arable* habitat near landfills (0-10 km) in 2021 (Table 1, all Tukey HSD tests p<0.008). Nonetheless, in 2021 nesting pairs at 10-20 km from landfills utilized a significantly higher proportion of Arable habitat than at 20-30 km, but not closer to the landfill, at 0-10 km (Table S2, Fig. 2) (Tukey HSD test, Estimate=3.396 ± 0.867, Wald = 3.916, p = 0.003). None of the other differences of Arable habitat were significant (all Tukey HSD tests p > 0.09).

Forest increased in the surrounding area of nests (2 km buffer) with an increasing distance to landfills, independent of the census year (Table 1, Fig. 2). Post-hoc tests confirmed significant differences amongst each landfill distance category (all Tukey HSD test p < 0.006). White storks also showed avoidance of this type of habitat, even at 20-30 km from landfill (Fig. 2, Nest-site Selection Index < 1).

Opposite to *Forest*, the highest positive selection of *Urban* was at 0-10 km from a landfill (all Tukey HSD test, p < 0.001; Table 1, Fig. 2). Moreover, we found a positive selection for *Urban* (Nest-site selection index >1; Table 1). Model selection for *Urban* did not include the census (Table 1).

Model selection indicated that *Pastures & Dehesas* varied with landfill distance in relation to the census year (Table 1). Breeders selected *Pastures & Dehesas* at 0-10 km at half the rate in 2021 compared to the previous census, and more than double in 2001 and 2004 than in 2021, when landfill distance was greater than 10 km (Fig. 2, all post-hoc tests p < 0.02). Before 2021, *Pastures & Dehesas* was less frequently selected near landfills (0-10 km) than up to 10 km from a landfill (all Tukey HSD test, p < 0.001) (Table 1, Fig. 2). Accordingly, white storks nesting near a landfill (0-10 km) increased their preference for *Pastures & Dehesas* with time (Table S2). However, we did not find significant differences in *Pastures & Dehesas* in the surrounding area of nests between 2001 and 2004 for any landfill distance category (all Tukey HSD tests p > 0.63). Neither did we find significant differences in 2021 amongst distance to landfill categories (all Tukey HSD tests p > 0.14).

Furthermore, *Pastures & Dehesas* were positively selected in the surrounding area in 2021 independently of the landfill distance, but we found neither positive nor negative selection of this habitat near landfills (0-10 km) in 2004 and 2001 (Fig. 2).

Other agricultural habitat (*Agri*) was avoided (mean of Nest-site selection index = 0.161 ± 0.140) independently of the distance to landfill category or the census year. Model selection for this habitat type resulted in three top models ($\Delta AICc < 2$) with the null model as the best one, so we did not detect any significant influence of landfill distance nor year on *Agri* variation in surrounding areas (Table 1).

Table 1. Variation in the nest-site selection index in relation to the landfill distance and census year. We include additive (+) and interaction (∗) effects. In the case of Other agricultural areas (Agri), estimates and significance were obtained by model averaging of models with a ΔAIC < 2. The best models and variables with a significant effect (*i.e.,* the 95% confidence interval did not overlap with zero) are shown in bold.

Model selection						**Model averaging or best model**				
M	**Effects**	**K**	**AICc**	**ΔAICc**	**w**	**Variable**	**Est**	**SE**	**2.5%CI**	**97.5%CI**
Agri										
$M_{Ag}1$	**Null**	**4**	**-22013.15**	**0.00**	**0.438**	Landfill10-20	0.02	0.06	-0.15	0.29
$M_{Ag}2$	**Landfill**	**6**	**-22011.75**	**1.40**	**0.217**	Landfill20-30	-0.05	0.10	-0.41	0.04
$M_{Ag}3$	**Year**	**6**	**-22011.72**	**1.43**	**0.215**	Year04	-0.01	0.07	-0.28	0.24
$M_{Ag}4$	Landfill + year	8	-22010.56	2.59	0.120	Year21	-0.05	0.11	-0.48	0.07
$M_{Ag}5$	Landfill * year	12	-22005.8	7.35	0.011					
Arable										
$M_{Ar}1$	**Landfill * year**	**12**	**-40255.72**	**0.00**	**0.999**	**Landfill10-20**	**1.92**	**0.28**	**1.37**	**2.47**
$M_{Ar}2$	Landfill	6	-40239.46	16.27	0.001	**Landfill20-30**	**0.04**	**0.28**	**-0.51**	**0.60**
$M_{Ar}3$	Landfill + year	8	-40235.74	19.98	0.000	Year04	0.19	0.28	-0.37	0.75
$M_{Ar}4$	Null	4	-40178.00	77.72	0.000	Year21	-0.04	0.29	-0.60	0.52
$M_{Ar}5$	Year	6	-40176.09	79.63	0.000	**Landfill10-20: Year04**	**1.08**	**0.38**	**0.34**	**1.82**
						Landfill20-30: Year04	**-0.96**	**0.40**	**-1.73**	**-0.19**
						Landfill10-20: Year21	**0.99**	**0.38**	**0.24**	**1.73**
						Landfill20-30:Year21	-0.77	0.40	-1.54	0.01
Urban										
$M_{ur}1$	**Landfill**	**6**	**-2295.09**	**0**	**0.948**	**Landfill10-20**	**0.86**	**0.09**	**0.68**	**1.04**
$M_{ur}2$	Landfill + year	8	-2289.28	5.81	0.052	**Landfill20-30**	**-0.32**	**0.10**	**-0.50**	**-0.13**
$M_{ur}3$	Landfill * year	12	-2278.65	16.44	0.000					
$M_{ur}4$	Null	4	-2221.69	73.40	0.000					
$M_{ur}5$	Year	6	-2217.63	77.46	0.000					
Forest										
Mfo1	**Landfill**	**6**	**-5823.93**	**0**	**0.999**	**Landfill10-20**	**-0.19**	**0.03**	**-0.25**	**-0.12**
$M_{fo}2$	Landfill + year	8	-5810.05	13.88	0.001	Landfill20-30	-0.02	0.03	-0.09	0.05
$M_{fo}3$	Null	4	-5797.41	26.53	0.000					
$M_{fo}4$	Landfill * year	12	-5793.07	30.86	0.000					
$M_{fo}5$	Year	6	-5783.72	40.21	0.000					

Pasture & Dehesas										
$M_{PD}1$	**Landfill * year**	**12**	**915.97**	**0**	**0.999**	**Landfill10-20**	**-1.13**	**0.04**	**-1.21**	**-1.04**
$M_{PD}2$	Landfill	6	955.43	39.47	0.001	**Landfill20-30**	**0.68**	**0.04**	**0.60**	**0.76**
$M_{PD}3$	Landfill + year	8	964.20	48.24	0.000	Year04	0.21	0.23	-0.20	0.62
$M_{PD}4$	Null	4	985.14	69.17	0.000	Year21	0.15	0.21	-0.26	0.56
$M_{PD}5$	Year	6	994.19	78.22	0.000	**Landfill10-20: Year04**	**-0.64**	**0.07**	**-0.77**	**-0.51**
						Landfill20-30: Year04	**0.23**	**0.07**	**0.10**	**0.36**
						Landfill10-20: Year21	**-0.53**	**0.06**	**-0.65**	**-0.40**
						Landfill20-30: Year21	**0.31**	**0.06**	**0.19**	**0.43**

Note: Terms are **M**, model abbreviation; **k,** number of parameters; **AICc,** Akaike information criterion corrected for small sample size; **ΔAICc,** the AICc difference between the current model and the one with the lowest AICc value; **w,** Akaike weights; **Est,** estimate; **SE,** Standard error; **Landfill,** Landfill distance category.

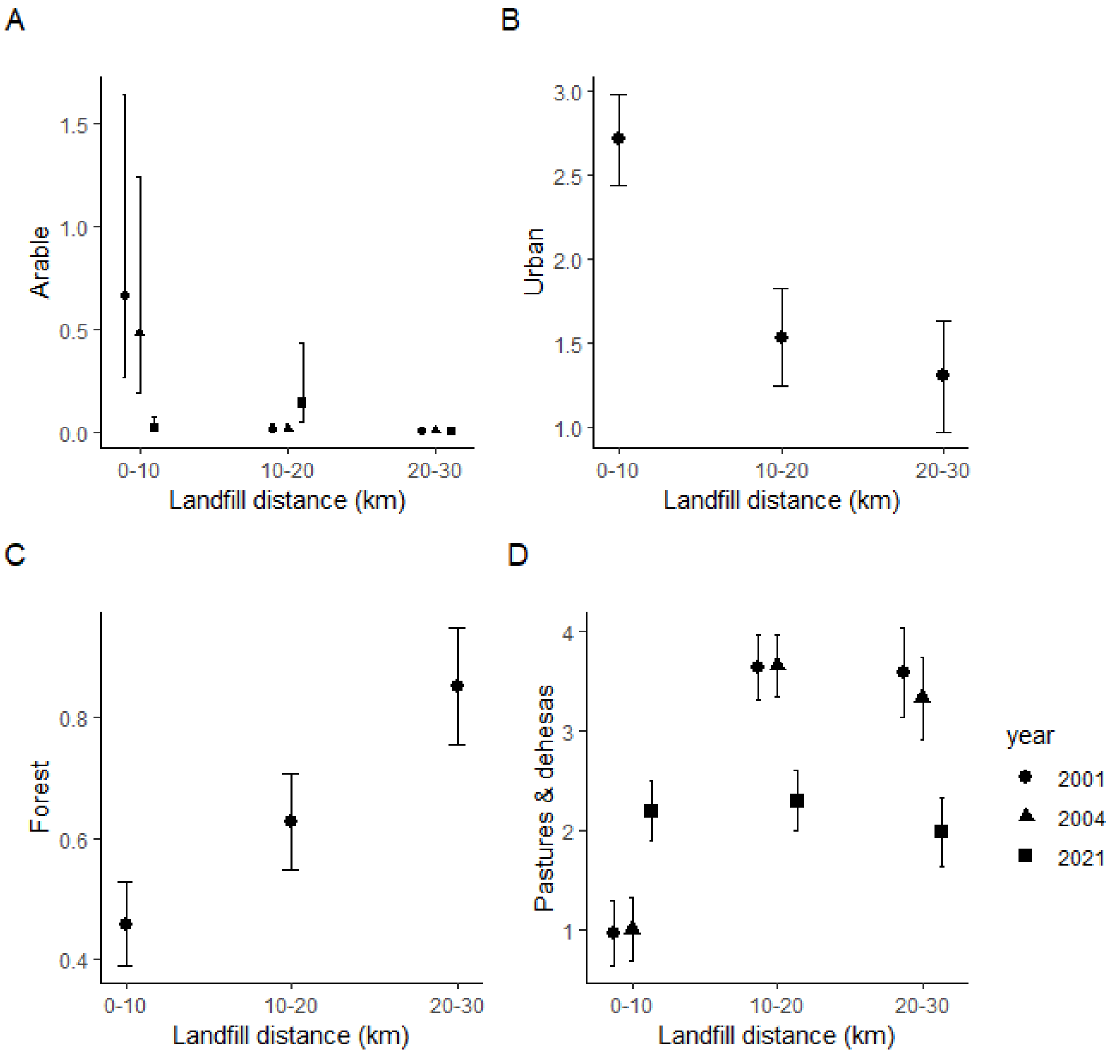


Fig. 2. Effect of the landfill distance category on the nest-site selection index: **a)** Arable lands, **b)** Urban area, **c)** Forest habitat, and **d)** Pasture & dehesas area. Whiskers represent the 95% confidence intervals. Census years are marked

with symbols: dots (2001), triangles (2004), and squares (2021). Note that non-significant differences between years are not plotted in **a)** Arable lands and **b)** Urban areas.

Breeding success

The number of fledglings varied in our study area from 0 to 5, with an average of 1.93 ± 1.25 fledglings per pair. The 82% of the 2556 monitored pairs in the three censuses registered a successful breeding outcome.

Breeding outcome depends on the proportion of habitat selected in the nest-site, and the year (Table 2; Fig. 3). Pairs with Higher Proportion of *Pastures & Dehesas* in the nest-site area showed a higher breeding outcome in 2004 (Tukey HSD test, Estimate =-1.6176 ± 0.373, Wald = -4.333, p < 0.001) and 2021 (Tukey HSD test, Estimate = -0.803 ± 0.274, Wald = -2.930, p = 0.040) but not in 2001 (Tukey HSD test, p = 0.999; Fig. 3).

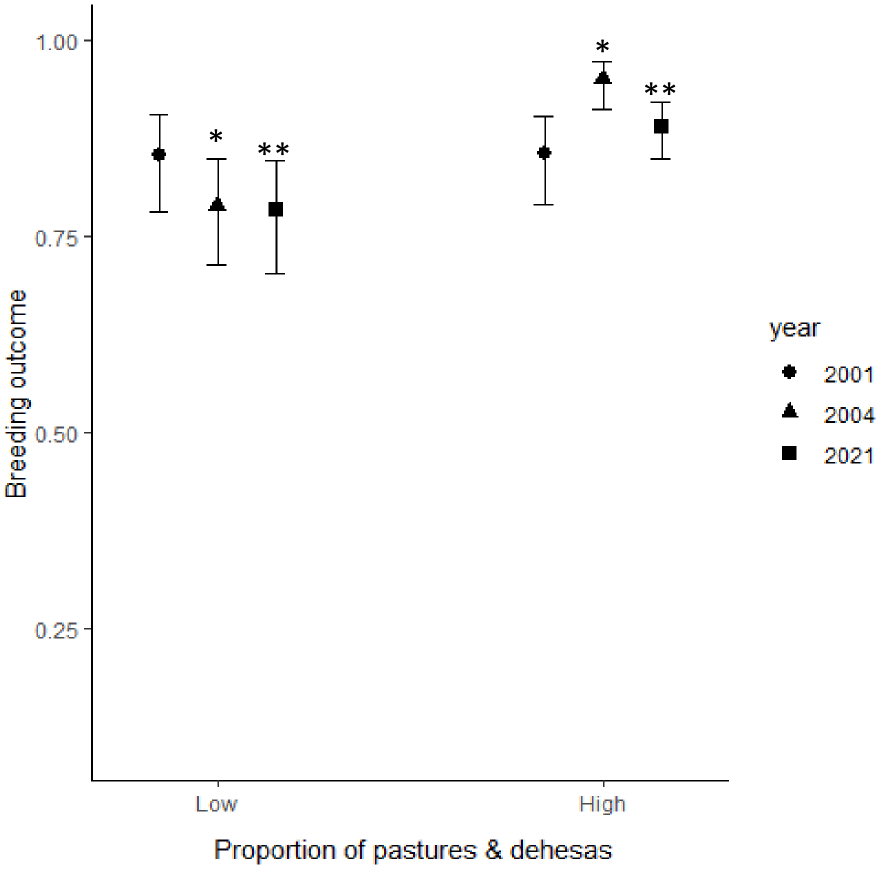


Fig. 3. Probability of producing at least one nestling (breeding outcome) related to the proportion of *Pastures & Dehesas* surrounding the nests. Whiskers represent the 95% confidence intervals. Census years are marked with symbols: dots (2001), triangles (2004), and squares (2021). Significant differences are marked with asterisks (*; **).

Although *Urban* habitat was included in the model average and showed a negative tendency in breeding outcome, this difference was not significant (Table 2).

We found an increase of 13% in the number of fledglings per pair when white storks bred 10-20 km from the landfill when compared to a 20-30 km distance (Tukey HSD test, Estimate = 0.130 ± 0.052, Wald = 2.593, p = 0.026). There were no significant differences amongst the other landfill distance categories (Tukey HSD test, p > 0.180; Fig. 4). In 2001, the number of fledglings was 17% higher than in 2004, independently of the landfill distance or habitat type in the surrounding area (Tukey HSD test, Estimate = 0.157 ± 0.041, Wald = 3.866, p < 0.001).

Similar results were found in breeding outcomes, where a Higher Proportion of *Pastures & Dehesas* in the nest-site area has a positive effect on the number of fledglings per pair, while a Higher Proportion of *Urban* may have a negative effect on the number of fledglings per pair (Table 2).

Table 2. Effects of landfill distance and the proportion of habitat selected on breeding success. We include additive (+) and interaction (∗) effects. Estimates and 95% confidence intervals (CI) were calculated by model averaging of models with a ΔAIC < 2. The best models and variables with significant effects (*i.e.,* the 95% confidence interval did not overlap with zero) are shown in bold.

Model selection						**Model averaging or best model**				
M	**Effects**	**K**	**AICc**	**ΔAICc**	**w**	**Variable**	**Est**	**SE**	**2.5%CI**	**97.5%CI**
Breeding outcome										
$M_{bo}1$	**Year* P.PD + Urban**	**9**	**2252.65**	**0.00**	**0.409**	Urban	0.162	0.092	-0.02	0.34
$M_{bo}2$	**Year* P.PD**	**8**	**2253.74**	**1.09**	**0.237**	**P.PD**	**-0.41**	**0.10**	**-0.60**	**-0.23**
$M_{bo}3$	P.PD*year + Landfill + Urban	11	2256.28	3.63	0.067	Year2004	-0.08	0.14	-0.35	0.19
$M_{bo}4$	Landfill*year + P.PD*year + Urban	15	2256.3	3.65	0.066	Year2021	0.26	0.15	-0.03	0.54
$M_{bo}5$	P.PD + Urban	5	2256.44	3.79	0.061	**P.PD:Year2004**	**0.40**	**0.13**	**0.14**	**0.67**
$M_{bo}6$	P.PD*year + Landfill	10	2257.64	4.99	0.034	**P.PD:Year2021**	**-0.41**	**0.14**	**-0.69**	**-0.13**
$M_{bo}7$	P.PD	4	2257.96	5.31	0.029					
$M_{bo}8$	Landfill*year + P.PD*year	14	2258.18	5.53	0.026					
$M_{bo}9$	Year + P.PD + Urban	7	2258.93	6.28	0.018					
$M_{bo}10$	Landfill*P.PD + year* P.PD + Urban	13	2259.5	6.85	0.013					
Number of fledglings										
$M_{f}1$	**Landfill + P.PD + Urban + year**	**9**	**6445.60**	**0.00**	**0.238**	Landfill10-20	0.00	0.03	-0.05	0.05

$M_{f}2$	**Landfill + P.PD + year**	**8**	**6447.21**	**1.61**	**0.107**	**Landfill20-30**	**0.06**	**0.03**	**0.01**	**0.11**
$M_{f}3$	**Landfill* P.PD + year + Urban**	**11**	**6447.33**	**1.74**	**0.100**	**P.PD**	**0.05**	**0.02**	**-0.09**	**-0.02**
$M_{f}4$	Landfill*year + P.PD + Urban	13	6447.61	2.01	0.087	Urban	0.03	0.02	-0.001	0.06*
$M_{f}5$	Landfill*year + Landfill* P.PD + Urban	15	6448.44	2.84	0.057	**Year2004**	**0.08**	**0.02**	**0.04**	**0.13**
$M_{f}6$	Landfill*year+ Landfill* P.PD + Urban	15	6448.44	2.84	0.057	**Year2021**	**-0.08**	**0.02**	**-0.13**	**-0.04**
$M_{f}7$	Landfill*year + P.PD	12	6448.71	3.12	0.050	Landfill10-20:PD	0.01	0.03	-0.05	0.06
$M_{f}8$	Landfill* P.PD + year	10	6448.86	3.26	0.047	Landfill20-30:PD	-0.04	0.03	-0.09	0.01
$M_{f}9$	Year + P.PD + Urban	7	6449.23	3.63	0.039					
$M_{f}10$	P.PD*year + Landfill + Urban	11	6449.40	3.80	0.036					

Note: Terms are **M**, model abbreviation; **k,** number of parameters; **AICc,** Akaike information criterion corrected for small sample size; **ΔAICc,** the AICc difference between the current model and the one with the lowest AICc value; **w,** Akaike weights; **Est,** estimate; **SE,** Standard error; **Landfill,** Landfill distance category; **P.PD,** proportion of Pastures & Dehesas habitat in relation to the availability in the region; **Urban,** proportion of urban habitat in relation to availability in the region.

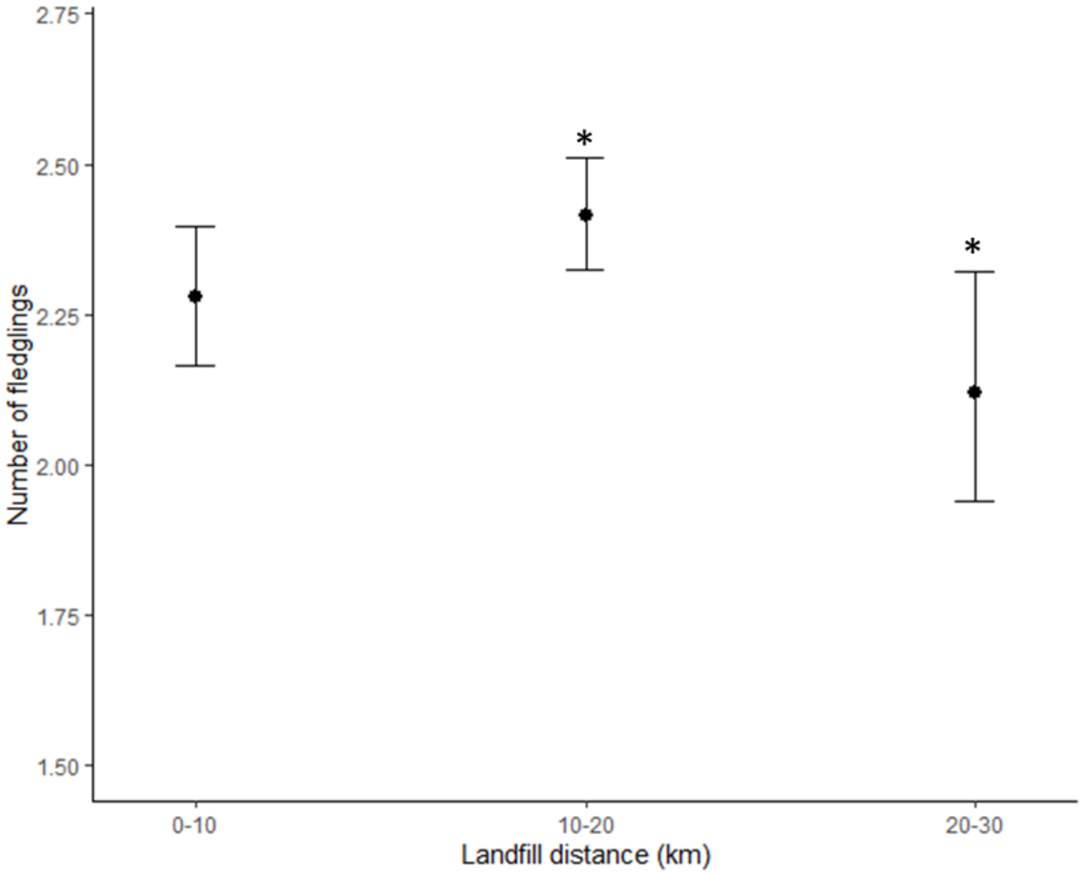


Fig. 4. Number of fledglings in relation to landfill distance. Whiskers represent the 95% confidence intervals. Significant differences are marked with asterisks (*).

Discussion

In this study, we focused on the differences in nest-site selection in relation to nesting distance to landfills and its consequences on reproductive parameters. As we hypothesized, breeding near a landfill affects nest-site selection by reducing the pressure for quality foraging areas in the surrounding area, and may also increases tolerance to humans. Additionally, we showed a reduction of this effect over the last 20 years, perhaps related to the plastic ability of this species to switch diet and explore new food sources (Sanz-Aguilar *et al.*, 2015; Carlson *et al.*, 2021), and the large population growth registered in the last census in the province of Madrid (López-García & Aguirre, *In press*). Although breeding near landfills usually increases breeding output and body condition of nestlings (Tortosa *et al.*, 2002; Archaux *et al.*, 2008; Steigerwald *et al.*, 2015), our results highlight the important role of high quality foraging areas in breeding success.

Consistent with previous findings in the Iberian Peninsula before landfills were extensively used by this species (Alonso *et al.*, 1991; Carrascal *et al.*, 1993), breeding pairs at distances greater than 10 km from a landfill in our region show the highest preference for pastures and dehesas and avoidance of poor feeding habitats in the Mediterranean context (*i.e.,* arable lands and other agricultural areas). Pastures and meadows represent an optimal foraging habitat for white storks, with high success in foraging attempts and prey abundances (Zurell *et al.*, 2018; Orłowski *et al.*, 2019; Bialas *et al.*, 2021). The strong selection and the preference for pastures and dehesas in this and other studies across Europe, highlights the relevance of this habitat in nest-site selection of white storks (Alonso *et al.*, 1991; Carrascal *et al.*, 1993; Radović *et al.*, 2015; Zurell *et al.*, 2018; Bialas *et al.*, 2020, 2021; Carlson *et al.*, 2021).

The degradation of wetlands and pastures has triggered the use of arable land as a suboptimal foraging area in some countries (Orłowski *et al.*, 2019; Bialas *et al.*, 2021). Contrary to these findings, white storks in our study area avoid arable land, except for breeders near landfills in 2001, which did not show preference or avoidance (Fig. 2). The lower prey abundance of these habitat types and its negative effect on reproductive success may explain why white storks avoid this type of habitat in the Mediterranean region (Orłowski *et al.*, 2019). We found a marked increase of high-quality feeding areas and avoidance of suboptimal foraging habitats up to 10 km from a landfill, which is particularly relevant during the energetically demanding breeding season (Ricklefs, 1973; Bryant, 1988).

However, our findings indicate that landfills alter this compromise. This alternative, abundant, and predictable anthropogenic resource guarantees food availability, mitigating the percentage and quality of foraging habitats required in the close vicinity of the nest during the breeding season (Oro *et al.*, 2013; Tauler-Ametlller *et al.*, 2019; Evans & Gawlik, 2020). Although arable areas are suboptimal foraging areas, human activities, such as mowing or ploughing, enhance their value for storks (Golawski & Kasprzykowski, 2021; Wikelski *et al.*, 2021). Similar to the use of landfills, individuals feeding during these agricultural activities may display a broader tolerance to human presence (Bonier *et al.*, 2007). Furthermore, breeders with a wider nest-site selection seem to be the best candidates to explore new food sources (*i.e.,* landfills), as we see in 2001 and 2004 (Bonier *et al.*, 2007; Evans *et al.*, 2010). The near absence of a landfill effect on habitat preference and nest-site selection in 2021 has several non-exclusive explanations (Fig. 2 and Fig. S1). It is possible that individuals were segregated by distance in the past, based on their ability to use landfills, whereas individuals that complemented natural foraging areas with food from anthropogenic sources proliferated throughout the entire region in recent years (Gilbert *et al.*, 2016; Mendes *et al.*, 2018; O'Hanlon *et al.*, 2022). Another explanation might be that the population growth observed over the last 20 years (López-García & Aguirre, *In press*) boosted intra-specific competition, maximizing the use of natural foraging areas in the vicinity of the nest (Real *et al.*, 2017). Similar to previous studies, the impact of landfills on bird diet decreases with distance (Zorrozua *et al.*, 2020b; Pineda-Pampliega *et al.*, 2021). This supports the conclusion that the intensity of landfill use decreases with distance because of the trade-off between flight costs and benefits (Gilbert *et al.*, 2016; Soriano-Redondo *et al.*, 2021).

In addition to anthropogenic food sources (*i.e.,* landfills, zoos, and rivers surrounding areas in an urban landscape), the urban habitat provides other nesting benefits. Urban dwellers commonly experienced a reduction of predation risks and a longer breeding season with higher breeding success due to the attemperate climate conditions of cities

(Tella *et al.*, 1996; Møller, 2009). However, the disadvantages of nesting in an urban environment may overcome the benefits, as urban habitat decreases with landfill distance. Urban birds usually must cope with physiological stress, a widespread variety of pollutants, and human perturbations (Lowry *et al.*, 2013; Mainwaring, 2015; James Reynolds *et al.*, 2019). What is more, the proliferation of birds in human-made structures often increases conflicts with humans and might even change the perception of the species to that of a pest species (Belant, 1997; Belant *et al.*, 1998).

In contrast, forestry areas are the ancestral nest-site habitats, which provide far fewer food sources to white storks but also less human pressure and conflicts (Donázar *et al.*, 2002; Tryjanowski *et al.*, 2018; Zurell *et al.*, 2018). Urban structures provide nesting sites in areas where nest sites are otherwise limited (Mainwaring, 2015). The increment of the forest habitat in nest-site selection with distance from a landfill, in contrast with the strongest nest site selection surrounded by urban habitat near a landfill, may signal the buffer effect of anthropogenic food sources on nest-site selection.

Overall, breeding in an urban landscape requires resilience to anthropogenic nuisances. Both the use of landfills and breeding in new structures imply the species may be less sensitive to human pressure. White storks breeding near landfills may develop a plastic response or behavioural changes to cope with human perturbance (Bonier *et al.*, 2007; Lowry *et al.*, 2013). Therefore, birds nesting in the same landfill distance category may share some traits with a genetic basis that ultimately could lead to population divergence (Garroway & Sheldon, 2013; Johnson & Munshi-South, 2017).

Several studies have demonstrated the advantages of feeding on anthropogenic food sources on different reproductive traits, including the increase in clutch size, viability of the last egg, body condition, number of nestlings, and fledgling rate (Tortosa *et al.*, 2002; Djerdali *et al.*, 2008b, 2016a; Pineda-Pampliega *et al.*, 2021). Despite such benefits, habitat quality (*i.e.,* high proportion of Pastures & Dehesas) but not breeding near landfill seems to be the key for breeding outcome in our study area, as well as in other regions (Orłowski *et al.*, 2019; Bialas *et al.*, 2021). The high quality of prey in natural foraging areas (Kosicki *et al.*, 2006; Vrezec, 2009) may contrast with the low quality of rubbish food sources (Annett & Pierotti, 1999; Murray *et al.*, 2018). Beyond poor food quality, birds feeding on landfills are exposed to a variety of pollutants and potential physical damage caused by plastic ingestion or entanglement in ropes (Peris, 2003; Henry *et al.*, 2011; Plaza & Lambertucci, 2018). These threats are extensive to individuals breeding in urban habitats, where contamination and physiological stress by human pressure causes physical or genetic alterations which may affect the survival or reproductive output of breeders (Seress & Liker, 2015; Capilla-Lasheras *et al.*, 2017). The use of landfills together with urban habitats increases aircraft and power line collisions (Pfeiffer *et al.*, 2020; Marcelino *et al.*, 2021). This may explain why we found a positive effect of landfill distance on the number of fledglings in nests at a medium distance from landfills (10-20 km) compared to nests far away from a landfill (20-30 km), but not with nesting pairs near a landfill (0-10 km). It is in line with our previous findings that moderate use of landfills enhance breeding success, supporting the idea of a trade-off between risks and benefits (López-García *et al.*, 2021).

Further research may confirm that the wide range in the number of nestling breeding success in our study is due to differences in the intensity of landfill use. Although the costs of foraging on landfills increases with distance (Gilbert *et al.*, 2016; Soriano-Redondo *et al.*, 2021), we cannot omit that it may benefit younger or poor quality breeders (Genovart *et al.*, 2010; Steigerwald *et al.*, 2015). In addition, birds breeding in poor quality habitats may have no other choice for foraging than at a landfill, independently of the distance covered to reach it (Evans & Gawlik, 2020; Langley *et al.*, 2021). Therefore, individuals may show diet specialization even within each landfill distance category (Sanz-Aguilar *et al.*, 2015; López-García *et al.*, 2021; O'Hanlon *et al.*, 2022).

To summarize, landfills provide a constant, predictable, and almost unlimited food source which allows birds nesting near landfills to be less selective about habitat quality during the breeding season. It is possible that individual differences in the intensity of use, at the same landfill distance, are underlaid by nest-site selection. Habitat quality in the surrounding area of the nest is a determinant in nest site selection and breeding success, independent of landfill use. Because of this, we need to preserve pastures and meadows. This is even more important as European legislation allows for restricting birds from landfills (Directive 1999/31/EC and 2018/850/EC). This knowledge about shifts of habitat preference related to landfill use may allow us to predict the recovery or expansion of a population and the future of the species in the absence of these facilities.

Supplementary material

Table S1. Corine Land Cover classes into seven Habitats categories, as functional Land cover groups, and percentage of cover of each habitat category in our study area in 2001, 2004, and 2021. Habitat categories were defined based on previous knowledge on white storks (Alonso *et al.*, 1991; Carrascal *et al.*, 1993; Radović *et al.*, 2015; Zurell *et al.*, 2018; Orłowski *et al.*, 2019; Bialas *et al.*, 2020, 2021). Corine Land Cover Code and Corine Land Cover classes were extracted from European Environment Agency (https://www.eea.europa.eu/data-and-maps/data/corine-land-cover-2/corine-land-cover-classes-and/clc_legend.csv).

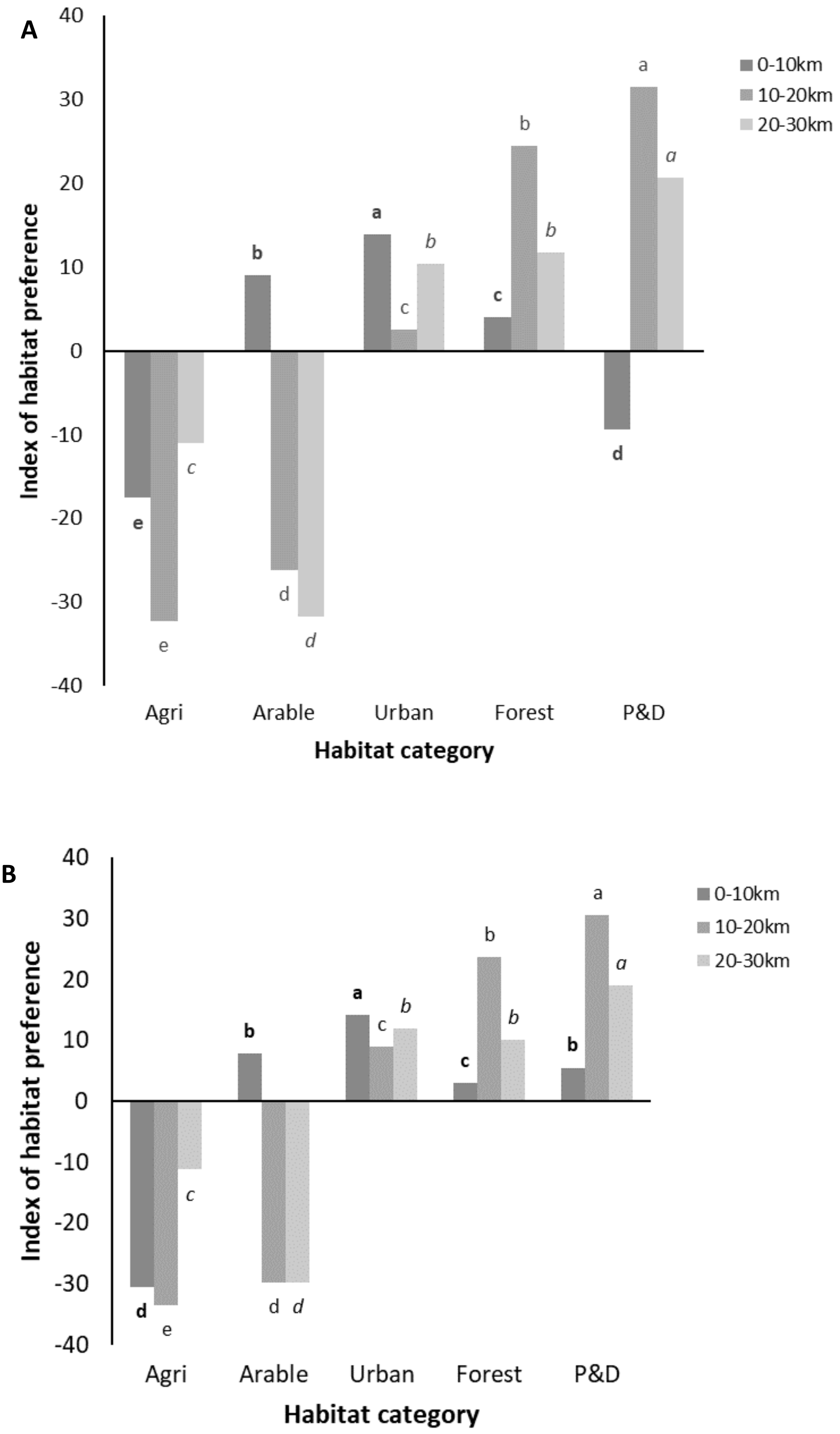
CLC Code	Corine Land Cover Class	Habitat category	2001	2004	2021
111	Continuous urban fabric	Urban	12.82	15.97	17.52
112	Discontinuous urban fabric				
121	Industrial or commercial units				
122	Road and rail networks				
124	Airports				
131	Mineral extraction sites				
132	Dump sites				
133	Construction sites				
141	Green urban areas				
142	Sport and leisure facilities				
211	Non-irrigated arable land	Arable	33.62	31.34	31.70
212	Permanently irrigated land				
221	Vineyards	Agri	13.12	12.68	8.06
222	Fruit trees and berry plantations				
223	Olive groves				
242	Complex cultivation patterns				
243	Land principally occupied by agriculture, with significant areas of natural vegetation				
231	Pastures	P&D	12.97	12.72	14.53
244	Agro-forestry areas				
321	Natural grasslands				

311	Broad-leaved forest	Forest	26.52	26.30	26.68
312	Coniferous forest				
313	Mixed forest				
322	Moors and heathland				
323	Sclerophyllous vegetation				
324	Transitional woodland-shrub				
331	Beaches, dunes, sands	Non-Suitable	0.42	0.37	0.94
332	Bare rocks				
333	Sparsely vegetated areas				
334	Burnt areas				
411	Inland marshes	Water	0.53	0.62	0.57
511	Water courses				
512	Water bodies				

Table S2. Nest-site Ranking Index by breeding white storks in relation to distance to landfill along several years. Habitat types are ranked from the most preferred to the least preferred. A significant difference in habitat preference between two habitats is indicated by ‘‘>>’’ while a non significant difference was indicated by ‘‘>’’. P values were obtained by random permutations.

Landfill Distance	Year	n	Wilk’s λ	p	Habitat ranking
0-10km	2021	885	0.216	0.001	P&D>Urban>>Forest>>Arable>>Agri
	2004	424	0.291	0.001	Urban>>Arable>P&D>>Forest>>Agri
	2001	374	0.472	0.001	Urban>>Arable>>Forest>>P&D>>Agri
10-20km	2021	952	0.178	0.001	P&D>Urban>>Forest>>Arable>>Agri
	2004	527	0.103	0.001	P&D>>Forest>>Urban>>Arable>>Agri
	2001	389	0.072	0.001	P&D>>Forest>>Urban>>Arable>>Agri
20-30km	2021	272	0.288	0.001	P&D>>Urban>Forest>>Arable>>Agri
	2004	121	0.443	0.001	P&D>>Urban>Forest>>Agri>>Arable
	2001	96	0.419	0.001	P&D>>Forest>Urban>>Agri>>Arable

Note: terms are **Landfill Distance,** range of distance to the nearest Landfill; **Year,** census year; **n,** number of nests; **Wilk’s λ**, statistic estimate value of Wilk’s λ; **p,** significance value; **Urban**, cover of artificial urban areas; **P&D,** cover of pastures and dehesas; **Forest,** cover of forest areas without grassland; **Arable,** cover of arable lands; **Agri,** cover of other agricultural lands.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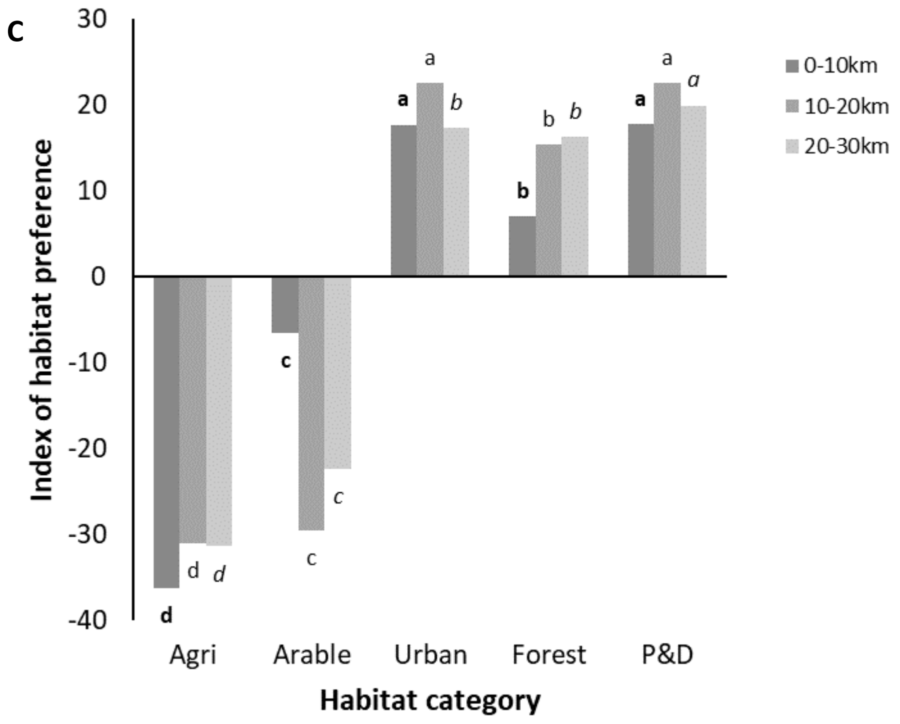


Fig. S1. Nest-site Ranking Index by white storks in Madrid region for 2001 **(A)**, 2004 **(B)** and 2021 **(C)**. Habitat categories: **Agri**, cover of other agricultural lands; **Arable,** cover of arable lands; **Urban,** cover of artificial urban areas; **Forest,** cover of forest areas without grassland; **P&D,** cover of pastures and dehesas. Significant differences between habitat categories are marked with: bold letters in the proximities of landfills (0-10km), normal letters in breeding pairs with access to the landfill but with extra cost (10-20km), and italic letters in breeding pairs with difficult or non-access to the landfill (20-30km).

CHAPTER 3 - THE TRADE-OFFS OF FORAGING AT LANDFILLS: LANDFILL USE ENHANCES HATCHING SUCCESS BUT DECREASE THE JUVENILE SURVIVAL OF THEIR OFFSPRING ON WHITE STORKS (*CICONIA CICONIA*)

Based on manuscript: López-García, A., Sanz-Aguilar, A. & Aguirre, J.I. 2021. The trade-offs of foraging at landfills: Landfill use enhances hatching success but decrease the juvenile survival of their offspring on white storks (*Ciconia ciconia*). *Science of the Total Environment*, 778: 146217. doi.org/10.1016/j.scitotenv.2021.146217

Abstract

During the last decades, landfills have become a valuable food source for wildlife, being in some cases determinants of large avian population increases. Superabundant food resources at landfills can increase reproductive and/or survival parameters; however, negative effects such as intoxication, plastic ingestion, skeletal deformities, unbalanced oxidative stress, and other health problems have also been reported. White stork (*Ciconia ciconia*) commonly benefits from landfill resources. Here, we evaluate potential landfill effects on demographic parameters (reproduction and offspring survival) at the individual level in a single population.

Our results show that a more intense use of landfills by breeders has a positive effect on hatching success but a negative effect on juvenile survival probability after emancipation, at least during the first year of life. High amount of food and proximity to landfill may explain their beneficial effect on reproductive parameters. On the other hand, poor quality food, pollutants, and pathogens acquired during early development from a diet based on refuse may be responsible for reduced future survival probability. Consequently, both positive and negative effects were detected, being foraging at landfills at low to medium levels the better strategy. Although our study shows that intense foraging on rubbish can imply both costs and benefits at an individual level, the benefits of superabundant food provisioning observed at population level by other studies cannot be ignored. Management actions should be designed to improve natural food resources, reduce non-natural mortality and/or human disturbances to guarantee the species viability under current European Union regulations designed to ban open-air landfills in a near future.

Resumen

Durante las últimas décadas, los vertederos se han convertido en una fuente de alimentación valiosa para la fauna, siendo en algunos casos determinantes del gran incremento de las poblaciones de aves. Los recursos tróficos superabundantes en los vertederos pueden influir positivamente sobre los parámetros reproductores y/o de supervivencia; sin embargo, también se han observado efectos negativos, como la intoxicación, la ingestión de plásticos, las deformidades esqueléticas, algunos desequilibrios del estrés oxidativo y otros problemas de salud. La Cigüeña blanca (*Ciconia ciconia*) es una de las especies que se ha beneficiado en las últimas décadas de los recursos tróficos que ofrecen los vertederos. En este trabajo se evalúan los efectos potenciales del vertedero en los parámetros demográficos (reproducción y supervivencia de la descendencia) a nivel individual dentro de una misma población.

Nuestros resultados muestran que un uso más intenso de los vertederos por parte de los reproductores tiene un efecto positivo en el éxito de eclosión, pero un efecto negativo en la probabilidad de supervivencia juvenil tras la emancipación del nido, al menos durante el primer año de vida. La elevada cantidad de alimento y la cercanía al vertedero podrían explicar los efectos beneficiosos en los parámetros reproductores. Por otro lado, la mala calidad de los alimentos, los contaminantes y los patógenos adquiridos durante las etapas tempranas del desarrollo, debido a una dieta basada en desperdicios, quizá sean responsables de la reducción en la supervivencia futura. En consecuencia, fueron detectados ambos efectos, negativos y positivos, siendo la mejor estrategia hacer un uso bajo o medio de los vertederos. Aunque nuestro estudio muestra que un uso intenso de los vertederos puede implicar costes y beneficios a nivel individual, los beneficios de la inmensa cantidad de alimento proporcionado por estas instalaciones en otros estudios no pueden ser ignorados. Las medidas de gestión deberían estar dirigidas a la mejora de las fuentes de alimentación naturales, a la reducción de la mortalidad no natural y/o de las perturbaciones del ser humano, con el fin de garantizar la viabilidad de las especies bajo las actuales regulaciones de la Unión Europea diseñadas para prohibir los vertederos de cielo abierto en un futuro próximo.

Introduction

Around one third of total worldwide food production is wasted or lost in supply chains (FAO, 2011). In the European Union, this translates to 88 million tonnes of food are wasted yearly and mainly end up in landfills (Stenmarck *et al.*, 2016), transforming these facilities in a worldwide, predictable, abundant, and anthropogenic food resource for wildlife. Since feeding on landfill facilitates food accessibility, reduces energetic costs typically associated with food foraging on wilderness, and diminishes competition for feeding resources, it implies several benefits to landfill scavengers at both population and individual levels (Oro *et al.*, 2013; Plaza & Lambertucci, 2017). As a consequence, a large diversity of animal species have changed their behaviour to take advantage of anthropogenic food refuse (Oro *et al.*, 2013; Plaza & Lambertucci, 2017). The superabundant food provided by landfills can improve the body condition as well as the reproductive parameters of species exploiting these resources (Eley *et al.*, 1989; Tortosa *et al.*, 2003; Auman *et al.*, 2008; Steigerwald *et al.*, 2015; Djerdali *et al.*, 2016b). Furthermore, easy food access and predictability associated with landfills imply an increase on individual survival (Eley *et al.*, 1989; Rotics *et al.*, 2017). Thus, for some species, landfill use promotes population growth and plays a key role for the recovery of vulnerable or endangered species (Tauler-Ametller *et al.*, 2017).

Anthropogenic leftovers are, however, usually poor-quality food (Grémillet *et al.*, 2008; Murray *et al.*, 2018) and, in landfills, organic waste is mixed with other non-beneficial items (e.g., metals, plastics, glasses, wires, and different toxic products or pollutants) that may cause amputation, suffocation, and/or intoxication (Matejczyk *et al.*, 2011). Nutrition is a key component of animal health, particularly on early development, with important consequences on future phenotypic traits (Lindström, 1999). Lack of essential micronutrients linked to low quality food available at dumps could modify morphological or behavioural traits and jeopardize future individual survival (Catoni *et al.*, 2008; Noguera *et al.*, 2015; Richardson *et al.*, 2019). In the same vein, dense aggregations of individuals in predictable rubbish subsides may also involve a wide range of health issues on immune system including higher probabilities of pathogen transmission which potentially effects reproduction and/or survival (Becker *et al.*, 2015; Plaza & Lambertucci, 2018). Finally, landfills favour aggregation of breeding individuals in their periphery (Bialas *et al.*, 2020), reducing the time that breeders spend away from their breeding sites (Moritzi *et al.*, 2001). These animals are attracted by food abundance, which would lead to an increase of competence (*i.e.,* nesting places) (Gilchrist & Otali, 2002; Djerdali *et al.*, 2016b). In summary, both positive and negative effects on life history traits that play a key role on population dynamics can occur as a consequence of landfill use by wildlife (Oro *et al.*, 2013).

The white stork (*Ciconia ciconia*) is a good model species to study landfill use effects on wildlife. The white stork is a species for which numerous positive effects of increased food availability at landfills have been described (Tortosa *et al.*, 2002; Djerdali *et al.*, 2016a; Rotics *et al.*, 2017; Cheng *et al.*, 2019). European populations experienced dramatic declines and became endangered after the 50s, but most of them have evidenced dramatic population recoveries over the last 30 years associated to an increasing use of landfills, among other factors (Barlein, 1991; Blanco, 1996; Schulz, 1999; Tortosa *et al.*, 2002; Massemin-Challet *et al.*, 2006). Specifically, storks nesting near landfills show bigger clutch sizes, increased egg viability, nestling survival, and number of fledglings (Tortosa *et al.*, 2002; Djerdali *et al.*, 2008b, 2016a,b). Moreover, the appearance and increased availability of predictable feeding resources during the last decades across Europe resulted in shortened migration distances, new wintering areas closer to breeding areas, shifts on passage at Strait of Gibraltar, or complete migration suppression resulting in enhanced survival probabilities of juveniles and adults at a population level (Ciach & Kruszyk, 2010; Sanz-Aguilar *et al.*, 2015; Rotics *et al.*, 2016, 2017; Zurell *et al.*, 2018; Cheng *et al.*, 2019). On the other hand, negative effects of anthropogenic food resources have also been shown for white storks such as plastic or rubber band ingestion, leg deformation by strings or wires (Peris, 2003; Kwieciński *et al.*, 2006), accumulation of metals, metaloids, PCBs, brominated flame retardants, and a variety of pathologies linked to this pollutants (Smits *et al.*, 2005, 2007; Saez *et al.*, 2009; Muñoz-Arnanz *et al.*, 2011; De la Casa-Resino *et al.*, 2015a; Pérez-López *et al.*, 2016; Martín-Maldonado *et al.*, 2020).

Overall, despite the inconveniences of foraging at landfills, positive effects of landfill use on demographic parameters (*i.e.,* reproduction and survival) at the population level are well known and have been documented for several stork populations. However, populations are typically composed of individuals with different behaviours (Bolnick *et al.*, 2003; Araújo *et al.*, 2011). Variability in individual experience, competitive abilities, optimization criteria, or physiological requirements could tip the scale to exploiting landfills instead of agricultural or other "natural" feeding areas (Sanz-Aguilar *et al.*, 2015). Consequently, individuals within a single population could experience differential reproductive or survival prospects associated to their use of foraging resources available at landfills.

In this study, we evaluate the demographic effects of individual foraging on landfills on reproductive parameters and offspring survival. We expected that higher use of predictable and abundant food resources at landfills may positively associate to demographic parameters as a consequence of extra-food supply (Oro *et al.*, 2013). However, foraging on junk food could also generate impacts on future survival of nestlings. The effects may be positive if the potential quantity of food is more important than the quality of food, or negative otherwise.

Materials and methods

Study area

We monitored the biggest white stork breeding colony in the Madrid region (Prado Herrero, 40.44 N, 3.49 W; Spain) from 1999 to 2019 (Aguirre & Atienza, 2002). The colony is located within a private cattle farm inside a protected area (Cuenca Alta del Manzanares Regional Park) and has experienced substantial growth during the last 20 years (55 breeding pairs in 1999; 163 in 2002; 179 in 2019). It is located 12 kilometres from the second largest landfill in the Madrid region (Colmenar Viejo, 40. 39N 3.44W).

From 1999 to 2002, the colony was surveyed at least twice a month in February and March to identify breeders previously marked with PVC rings using telescopes (20 × 30–60). We considered as breeders the individuals that were observed constructing, defending, or perching on nests. Moreover, from the end March to July the colony was visited to collect data on clutch size, number of hatched nestlings and fledglings, and to mark nestlings (see Aguirre and Vergara 2009 for further details). For each nest, we calculated hatching success (the ratio between number of nestlings and number of eggs laid) and fledgling success (the ratio between the number of fledglings and nestlings). Chicks with less and more than 40 days from hatching were considered nestlings and fledglings, respectively (Jovani & Tella, 2004). During the study period, one or the two breeders from 36 breeding pairs were individually identified at the breeding colony based on alphanumeric PVC rings. Age of marked breeders, ranging from 2 (3rd calendar year) to 11 years old (12th calendar year), was determined by their year of ringing because all individuals were marked as nestlings (More detailed information in Appendix A, Table A.1). For 30 pairs only one member was ringed but for 6 pairs both members were ringed: in 5 pairs both members had the same age and in one case they differ by one year-old.

Landfill use and Individual data

Colmenar Viejo Landfill was visited regularly (at least once a week) between 1999 and 2002, always from noon to sunset (during two to three hours of observation), when storks were concentrated foraging in largest numbers and to avoid peaks of activity by rubbish trucks (Blanco, 1996).

Using observations of marked storks at the landfill during the breeding season (March to June), we created an individual *Landfill Use Index* as a proxy of the use of landfill and the importance of rubbish in the diet of marked breeders and nestlings. The index was calculated as the number of observations of one particular bird within the total number of visits to landfill per year.

Reproductive parameters modelling

To evaluate the effect of Landfill Use Index by breeders on breeding success, we performed General Linear Mixed Models (GLMM) with *clutch size*, *number of nestlings*, *number of fledglings* (with a Poisson error distribution and a logit link function), and *hatching success and fledging success* (with binomial distribution and logit function) as response variables. The factors analysed were Landfill Use Index, parental age and year, and their interactions. For those pairs in which both members were marked (N=36), we used the average age and Landfill Use Index of both parents to avoid pseudo-replication in the data. Parental age was included in all analyses to control for a potential effect of individual breeding experience, as age in the white stork has been previously related with breeding performance(Schaub *et al.*, 2005; Vergara & Aguirre, 2006; Vergara *et al.*, 2007b; Nevoux *et al.*, 2008). Year of breeding was also included as a factor. Individual identity was considered as a random factor to avoid pseudo-replication (Hurlbert, 1984). Models were performed with package "lme4" in R 3.2.3 (https://www.r-project.org/). Sample sizes varied among models (see Results) because we were unable to collect clutch size or number of fledglings in all nests.

Model selection was based on the Akaike's Information Criterion corrected by sample size (AICc) (Burnham & Anderson, 2002). Models were ranked based on the differences between the AICc of a given model and the AICc of the model with the lowest AICc (ΔAICc) and models with a ΔAICc <2 points were considered equivalent (Burnham & Anderson, 2002). To minimise the potential for overfitting and significance-by-chance effects, models included those single variables and interactions that ranked lowest AICc.

Survival Modelling

We studied future survival of nestlings marked between 1999 and 2002 in our study area by means of capture-recapture models (Lebreton *et al.*, 1992). As we were interested in evaluating the potential effects of parental food provisioning on future individual survival, we only considered for our analyses those individuals marked at nests in which at least one parent was identified by PVC rings, resulting in 130 birds that were descendants from 26 breeders (*i.e.,* 26 families of storks). To build life histories, we considered seeing "1" or not seeing "0" each year, independently of the number of individual resighting per year. We also calculated a field effort covariate based on the standardised number of visits to Colmenar Viejo per year as a potential predictor of resighting probabilities.

We used the software U-CARE 2.2.2 (Choquet *et al.*, 2009a) to assess goodness-of-fit of the Cormack Jolly Seber model (CJS) with two age classes ($\phi_{2a.t}$, p_t) to our data (Pradel *et al.*, 2005). As individuals were marked as nestlings, all the models considered at least two age classes in survival (Pradel *et al.*, 2005). Goodness of fit tests indicated a good fit of the data ($\chi 2= 25.857$, df=24, p=0.360) and an overdispersion inflation factor (c-hat = 1.077) was applied to all models (Burnham and Anderson 2002).

We started from a general model including age dependent variation in survival (3 biologically meaningful age classes, see below) and resighting probabilities (2 biologically meaningful age classes, see below), and temporal variation for resighting parameters. Since juveniles (in their first year of life) return late to breeding areas or even remain their first year in Africa while adults could even not migrate, we considered that first year resighting probability may differ from that of older individuals (Doligez *et al.*, 2004; Nevoux *et al.*, 2008; Shephard *et al.*, 2015; Cuadrado *et al.*, 2016; Martín *et al.*, 2016). Regarding survival, previous studies have demonstrated that: Survival differs between first year and older individuals (Barbraud *et al.*, 1999; Schaub *et al.*, 2005; Nevoux *et al.*, 2008); survival differs among first year (juveniles), second year (subadults or young breeders), and individuals from 3 years old onwards (breeding individuals)(Kanyamibwa *et al.*, 1993); and survival differs among first year (juveniles), individuals from 2 to 5 year-old (young-breeders starting reproduction, here after young breeders) and individuals from 6 years old onward (experienced breeding adults)(Doligez *et al.*, 2004; Nevoux *et al.*, 2008). In our case, and given that storks start breeding at age 2 in our study area (Vergara & Aguirre, 2006), we used this last structure to start model selection.

We started model selection by testing the effect of time, the potential effect of effort devoted to identify marked birds, and the effect of age on resighting probabilities. Once we identified the best structure for resighting parameters, we tested alternative age structures for survival. Using the retained basic structure of resighting and survival parameters, we tested our hypothesis of interest on survival probabilities. For that purpose, we included three variables that could affect offspring survival: a) Landfill Use Index: “poor” quality food provided by parents during early development could affect survival on different life-stages, b) Parental age: older parents have more experience and may have a better performance in parental care improving offspring survival (Vergara & Aguirre, 2006; Vergara *et al.*, 2007b), and c) Family effect: since individuals from the same pair share similar genes, parental care, and parental diet, their survival probability could be more similar (Choquet *et al.*, 2013). The effects of parental age and Landfill Use Index were modelled as individual covariates affecting future survival. The potential effect of familiarity (parents-offspring quality on the survival probabilities of siblings) was assessed and included the family identity as a random cluster effect (Choquet *et al.*, 2013). The factors potentially affecting survival were first tested on juvenile survival. As the immediate effects in early life are the more plausible and if significant, we tested for long-lasting effects when young breeders and adults. Models were built and fitted to the data using E-SURGE 2.1.4 software (Choquet *et al.*, 2009b). Model selection was based on Akaike's information criterion by overdispersion and sample size (QAICc) (Burnham & Anderson, 2002). The effect of covariates was statistically significant when beta estimate corresponding to the linear slope did not include zero. To determine significance of cluster effect we performed an LRT test for the random effect(s) on the best fixed-effect model at the 5% significance level (Choquet *et al.*, 2013).

Combined effects of landfill use on reproductive parameters and future offspring survival

In order to better assess the effects of foraging at landfills on parental fitness we estimated the probabilities of having that a laid egg becomes a one-year-old stork by multiplying the estimated values of hatching success, fledging success and juvenile survival from best models in model selection (Table 1 and Table 2 in Results section).

Results

During the course of the study (1999–2002), Landfill Use Index ranged between 0 and 0.36, with no differences with breeders age (β = -0.019 ± 0.112, Z=-0.16, p =0.873) or between years (χ^2=1.339, df=3, p=0.720).

Reproductive parameters

Reproductive parameters in the studied colony were: clutch size (mean=3.44 ± 0.40 eggs, range 0–5, n = 45 nests), number of hatchlings (mean=2.48 ± 0.34, range 0–5, n=56 nests), and number of fledglings (mean=2.14 ± 0.37, range 0–4, n=49 nests).

In absolute numbers, landfill use did not influence clutch size, number of nestlings, or number of fledglings (Table 1). In fact, for clutch size and number of nestlings, the null model was the best one (Table 1), while for number of fledglings the best model included age and landfill effects (Table 1). However, the 95% CI of the model averaged estimates indicate that only the age effect was significant.

On the contrary, when the relative success of the nest was modelled, model selection indicated that individuals using landfill with more intensity had a higher hatching success of their clutches (Model $M_{hs}1$ and $M_{hs}2$, Table 1) (Fig.1). Regarding fledgling success, we detected a significant increase with breeder´s age (Model $M_{fs}1$, Table 1) but not an effect of landfill use (Table 1).

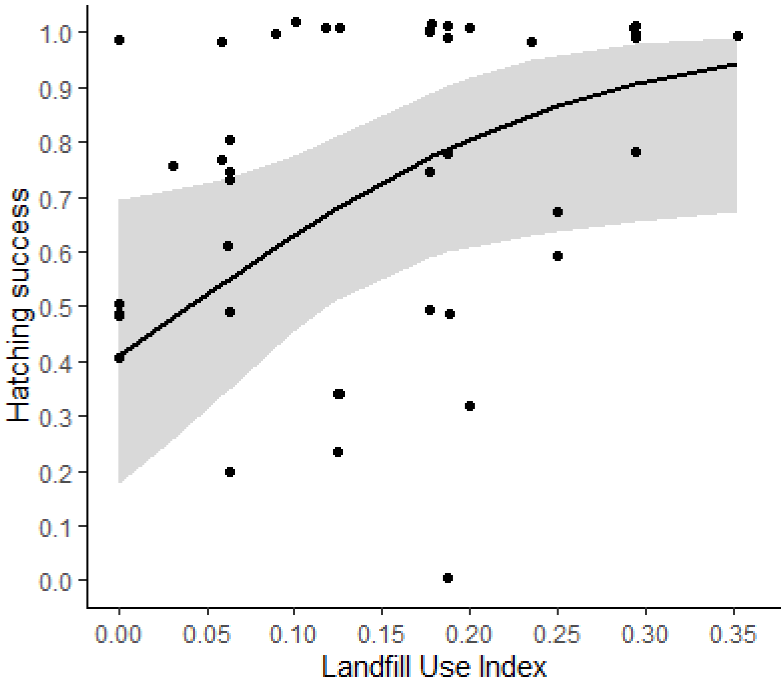


Fig. 1. Hatching success in relation to Landfill Use Index (and 95% CI given by shaded area) (Model $M_{hs}2$, Table 1).

Table 1. Reproductive parameters in relation to Landfill Use Index by breeders, age of breeder, and year of breeding. We include additive (+) and interaction (*) effects. Estimates and 95% confidence intervals (CI) were calculated by model averaging of models with a ΔAIC <2. In bold best models and variables receiving strong support (*i.e.,* the 95% confidence interval did not overlap with zero).

Model selection						**Model averaging**			
M	**Effects**	**K**	**AICc**	**ΔAICc**	**w**	**Variable**	**Est**	**2.5%CI**	**97.5%CI**
Clutch size									
M_c1	**Null**	**2**	**169.27**	**0.00**	**0.34**	Age	0.04	-0.02	0.09
M_c2	**Age**	**3**	**169.93**	**0.66**	**0.25**	Landfill	-0.78	-2.54	0.97
M_c3	**Landfill**	**3**	**170.76**	**1.49**	**0.16**				
M_c4	Landfill + Age	4	171.61	2.34	0.11				
M_c5	Year	4	172.99	3.72	0.05				

$M_{c}6$	Landfill * Age	5	173.73	4.46	0.04				
$M_{c}7$	Age + Year	5	173.84	4.57	0.04				
$M_{c}8$	Landfill + Year	5	175.02	5.75	0.02				
$M_{c}9$	Landfill * Age + Year	7	178.53	9.26	0.00				
Number of nestlings									
$M_{n}1$	**Null**	**2**	**195.88**	**0.00**	**0.24**	Age	0.04	-0.02	0.10
$M_{n}2$	**Age**	**3**	**196**	**0.12**	**0.22**	Year2000	0.29	-0.49	1.07
$M_{n}3$	**Year**	**5**	**197**	**1.12**	**0.14**	Year2001	0.16	-0.53	0.86
$M_{n}4$	**Landfill + Age**	**4**	**197.16**	**1.28**	**0.12**	Year2002	-0.22	-0.91	0.47
$M_{n}5$	**Landfill**	**3**	**197.19**	**1.31**	**0.12**	Landfill	0.83	-0.78	2.44
$M_{n}6$	Age + Year	6	198.58	2.70	0.06				
$M_{n}7$	Landfill + Year	6	198.76	2.88	0.06				
$M_{n}8$	Landfill * Age	5	199.56	3.68	0.04				
$M_{n}9$	Landfill * Age + Year	8	203	7.12	0.01				
Number of fledglings									
$M_{f}1$	**Age**	**3**	**165.55**	**0.00**	**0.49**	**Age**	**0.08**	**0.01**	**0.15**
$M_{f}2$	**Landfill + Age**	**4**	**167.15**	**1.60**	**0.22**	Landfill	0.93	-1.19	3.04
$M_{f}3$	Landfill * Age	5	168.36	2.80	0.12				
$M_{f}4$	Null	2	168.86	3.30	0.09				
$M_{f}5$	Landfill	3	170.71	5.16	0.04				
$M_{f}6$	Age + Year	6	171.4	5.85	0.03				
$M_{f}7$	Year	5	173.23	7.68	0.01				
$M_{f}8$	Landfill + Year	6	175.58	10.03	0.00				
$M_{f}9$	Landfill * Age + Year	8	175.6	10.04	0.00				
Hatching success									
$M_{hs}1$	**Landfill + Age**	**4**	**52.36**	**0.00**	**0.32**	**Landfill**	**9.45**	**0.50**	**18.39**
$M_{hs}2$	**Landfill**	**3**	**53.26**	**0.91**	**0.20**	Age	0.25	-0.05	0.55
$M_{hs}3$	Landfill + Year	5	54.76	2.40	0.10				
$M_{hs}4$	Landfill * Age	5	54.88	2.52	0.09				
$M_{hs}5$	Age + Year	5	55.28	2.92	0.08				
$M_{hs}6$	Year	4	55.32	2.96	0.07				
$M_{hs}7$	Age	3	55.65	3.29	0.06				
$M_{hs}8$	Null	2	56.28	3.92	0.05				
$M_{hs}9$	Landfill * Age + Year	7	56.95	4.59	0.03				
Fledging success									
$M_{fs}1$	**Age**	**3**	**27.45**	**0.00**	**0.61**	**Age**	**0.91**	**0.13**	**2.63**
$M_{fs}2$	Landfill + Age	4	29.71	2.26	0.20				

$M_{fs}3$	Null	2	31.28	3.83	0.09
$M_{fs}4$	Landfill * Age	5	31.91	4.46	0.07
$M_{fs}5$	Landfill	3	33.43	5.98	0.03
$M_{fs}6$	Year	5	36.98	9.53	0.01
$M_{fs}7$	Landfill + Year	6	39.62	12.17	0.00
$M_{fs}8$	Age + Year	5	55.28	27.82	0.00
$M_{fs}9$	Landfill * Age + Year	7	56.95	29.50	0.00

Note: Terms are **M,** model abbreviation; **np**, number of parameters; **AICc**, Akaike information criterion corrected for small sample size; **ΔAICc**, the AICc difference between the current model and the one with the lowest AICc value; **w**, Akaike weights; **Est**, estimate; **Landfill,** Landfill Use Index; **Age,** breeder´s age.

Offspring survival

We started model selection by testing different hypotheses on resighting probabilities (Table A.2, Appendix A). Our results indicated that resighting probabilities varied with the effort invested in field monitoring over the study period and were lower for first year individuals (Model 1p, Table A.2, Fig. A.1 in Appendix A).

For survival, different age structures (*i.e.,* no differences in survival between young breeders (age 2 to 5) and adults (age >5) or differences in survival among youngest breeders (age 2) and older adults (age > 2)) did not reduce the QAICc of the general model (Table 2). Consequently, we used the general and biologically meaningful age structure (Model 5s, Table 2) for testing the hypotheses of interest. Mean survival probabilities for juveniles, young breeders (age 2 to 5), and older adults (age >5) were 0.34 (95%CI=0.25-0.44), 0.75 (95%CI= 0.64-0.83), and 0.92 (95%CI=0.83-0.96), respectively (Model 5s, Table 2).

Parental age did not affect survival probabilities of offspring (Model 10s, Table 2), as the model did not reduce the QAICc value of model 5s and the 95%CI intervals of the beta estimate accounting for parental age largely overlapped zero (95%CI β: -1.29, +1.78). Similarly, family identity did not have a significant effect on offspring survival (χ^2=0.19, p=0.33; Models 9s, Table 2). On the contrary, models including an effect of the Landfill Use Index by parents on survival probabilities of offspring greatly reduced the QAIC (Table 2). Four models were equivalent in terms of QAICc, having less than 2 points of $QAIC_C$ difference among them (Models 1s-4s, Table 2). All of them included an effect on juveniles, but some of them also included an effect on young breeders (Model 3s, 4s) or older adults (Model 1s). The effect on juveniles was clear (Model 2s; 95%CI β: -9.64 – -

0.58) but the effect on adults was only significant (*i.e.,* the 95%CI of beta estimate did not include zero) when considered as additive (Models 1s, 3s) but not when a single slope (*i.e.,* interaction) was applied to young and old adult survival (e.g., Model 3s; Young breeders: 95%CI β = -1.15, 95%CI: -6.88 – +4.58.; Older adults: 95%CI β = -4.66, 95%CI: -15.93 – +6.61). The effect of parental landfill use on offspring survival was negative: individuals whose parents showed a lower use of the landfill had higher survival probabilities (Fig.2).

Table 2. Modelling of White Stork survival probabilities depending on age structure (up to 3 age classes with biological meaning, See Material and Methods), age of parents, Landfill Use Index by parents, and family resemblance (cluster effect). Recapture probabilities were modelled as a function of 2 age classes (first year vs. older birds) and temporal variations in field effort equally affecting both age classes (see Table A.2 Appendix A).

Model	Survival hypothesis	np	Deviance	$QAIC_c$	$\Delta QAIC_c$
1s	**$(\phi_{j}, \phi_{yb}, \phi_{ad})_{+landfill}$**	**7**	**510.53**	**488.45**	**0.00**
2s	**$\phi_{(j.\ landfill)}, \phi_{yb}, \phi_{ad)}$**	**7**	**510.59**	**488.51**	**0.06**
3s	**$(\phi_{j}, \phi_{yb})_{+landfill}, \phi_{ad}$**	**7**	**488.32**	**488.74**	**0.29**
4s	**$\phi_{(j.\ landfill)}, \phi_{yb.\ landfill)}, \phi_{ad}$**	**8**	**510.07**	**490.15**	**1.70**
5s	$\phi_{j}, \phi_{yb}, \phi_{ad}$	6	516.25	491.66	3.20
6s	$\phi_{(j.\ parental\ age)}, \phi_{(yb.\ landfill)}, \phi_{(ad.landfill)}$	9	509.46	491.73	3.27
7s	$\phi_{j}, \phi_{(yb.\ landfill)}, \phi_{ad}$	7	515.10	492.70	4.25
8s	$\phi_{j}, \phi_{yb}, \phi_{(ad.landfill)}$	7	515.29	492.88	4.42
9s	$\phi_{(j.\ familyl)}, \phi_{yb}, \phi_{ad)}$	7	516.04	493.57	5.12
10s	$\phi_{(j.\ parental\ age)}, \phi_{yb}, \phi_{ad}$	7	516.13	493.66	5.20
11s	$\phi_{j=yb}, \phi_{ad}$	5	523.11	495.94	7.49
12s	$\phi_{j}, \phi_{yb2}, \phi_{yb3-5=ad}$	6	523.09	498.01	9.56

Note: Terms are **np**, number of parameters; **QAICc**, Akaike information criterion corrected for overdispersion and small sample size; **ΔQAICc**, the QAICc difference between the current model and the one with the lowest QAICc value; **j**, juvenile (first year individuals); **yb**, young breeder including 2 to 5 years individuals; **yb2**, second year individuals; **yb3-5**, young breeder from 3 to 5 years old; **ad**, adult birds with 6 or more years old; **landfill**, parental Landfill Use Index effect; **parental age**, parental age effect on offspring survival; **family**, cluster effect regarding to sibling resemblance.

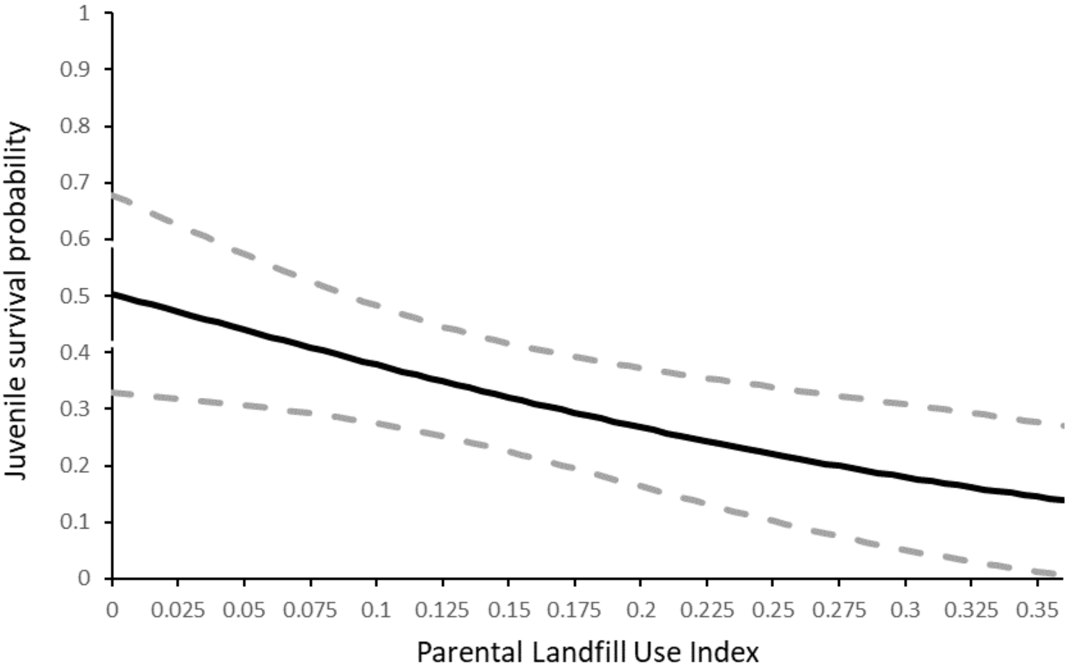


Fig. 2. Effect of Landfill Use by parents on future juvenile survival probability (and 95% CI given by dashed lines). Survival values change as a function of the Landfill Use Index described by Logit (ϕ_j)= 3.1519+ (-3.35287*Landfill Use Index)

Combined effects of landfill use on reproductive parameters and future offspring survival

Probability that a laid egg becomes a one-year-old stork was higher for older breeders with a low to moderate landfill use (Fig.3). An intensive landfill use index, although beneficial for hatching success implied high juvenile mortality, resulting in costs for parental fitness (Fig.3).

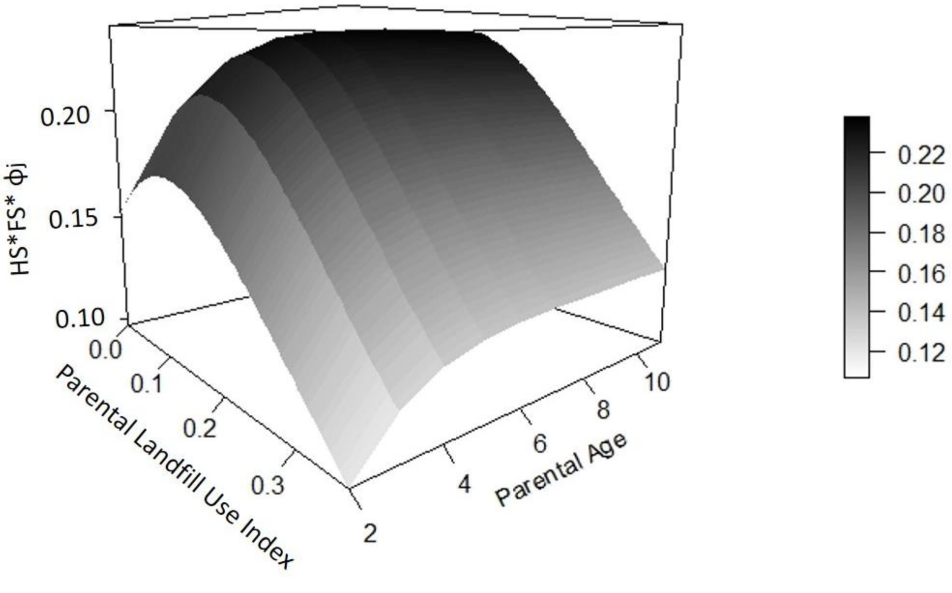


Fig. 3. Representation of the annual probability that a laid egg becomes a one-year-old stork (HS*FS*ϕ_j) next year as a function of parental age and landfill use.

Discussion

Despite the typical benefits of foraging at abundant and highly predictable food supplies provided by landfills described in the literature (See references in Oro *et al.*, 2013; Plaza and Lambertucci, 2017), our study highlights the existence of higher costs at individual level. In agreement with previous studies, our study reveals that a more intense landfill use by breeders can increase hatching success (Tortosa *et al.*, 2003; Djerdali *et al.*, 2008b, 2016b) but not fledgling success, clutch size of the total number of fledglings produced. However, breeders foraging intensively at landfills pay fitness costs in terms of future offspring survival. Our study highlights the existence of trade-offs at individual level: the absence of landfill foraging or an intense landfill use implied lower probabilities of having a one year-old descendent. On the contrary, storks with low to medium Landfill use index were the more successful.

Several studies provide evidence that white storks breeding near landfills have higher reproductive parameters and survival prospects than individuals from populations located at more “natural” areas (Tortosa *et al.*, 2002, 2003; Massemin-Challet *et al.*, 2006; Djerdali *et al.*, 2008b, 2016b; Rotics *et al.*, 2017). Here, at Prado Herrero colony, productivity parameters (See Results) were relatively lower than in other European populations breeding near landfills (clutch size=4.1 to 4.6, nestlings=3.6, fledglings=2.4 to 3.2)(Barbraud *et al.*, 1999; Massemin-Challet *et al.*, 2006).The number of fledglings was quite similar to those documented for eastern European populations not foraging intensively at landfills (fledglings=2.1 to 2.3)(Tobolka *et al.*, 2013, 2015; Fasolă-Mătăsaru *et al.*, 2018). Similarly, hatching success (0.73 ± 0.09) and fledging success (0.88 ± 0.08) were similar to other of European “natural” populations not foraging at landfills (hs= 0.79, fs=0.67 to 0.95)(Barbraud *et al.*, 1999; Massemin-Challet *et al.*, 2006). Mean survival of juveniles (0.34, 95%CI=0.25-0.44) and young breeders (0.75, 95%CI=0.64-0.83) was similar, while adult survival (0.91, 95%CI=0.83-0.96) was slightly higher to that estimated for other European populations (j=0.33 to 0.60; ad= 0.78 to 0.86, (Barbraud *et al.*, 1999; Schaub *et al.*, 2005; Nevoux *et al.*, 2008)). Therefore, mean demographic parameters of the Prado Herrero colony were more similar at the population level to those exhibited by European storks foraging at natural areas, even with being located near a landfill. These results seem to indicate that this population may partially feed on landfill but not depend totally on landfill refuses.

Population level variation among individuals exists: not all individuals exploit foraging resources equally (Bolnick *et al.*, 2003; Araújo *et al.*, 2011). The Landfill Use Index used here was calculated during the breeding season when storks barely spend time outside their nests on other activities that foraging (Moritzi *et al.*, 2001; Gilbert *et al.*, 2016). Consequently, it may reflect quite accurately the main source of individual diet, with individuals infrequently identified in the rubbish dump probably foraging in more "natural" areas and individuals seen multiple times and considered "landfill specialists". In this sense, previous studies on wintering areas indicate that foraging at landfills could be related to individual age, with old and experienced individuals with foraging preferences for more natural areas and potentially higher quality resources than young birds (Sanz-Aguilar *et al.*, 2015) and lower neophobia in juvenile individuals (Cambefort, 1981). Individual variation in landfill exploitation found here differs from this previous study since age was not correlated with Landfill Use Index and other factors like morphological, physiological, or behavioural traits (*i.e.,* personality) may influence individual diet especially during the breeding season.

The use of anthropogenic food subsides during breeding season allows breeders to reduce the energetic costs of foraging, resulting in an increased parental investment in nest attendance (Moritzi *et al.*, 2001; Gilbert *et al.*, 2016). In birds, warmth and care provided by parents are crucial at incubation and during the first days of life when hatchlings are particularly sensitive to adverse weather conditions (Jovani & Tella, 2004; Tobolka *et al.*, 2015). Therefore, higher parental care may explain incremental hatching success associated to higher landfill foraging use observed in this and previous studies (Djerdali *et al.*, 2008b). Furthermore, breeders not only would obtain food from landfills, but nest materials such as plastics, cardboard, and other items that would provide insecticide or thermoregulation benefits contribute to hatching success (Jagiello *et al.*, 2018). As well as landfills, parental age influenced positively the productivity of the nest, particularly for fledging success, as has been previously reported for several bird species (Newton, 1989). Older individuals are more efficient obtaining food (Marchetti & Price, 1989; Rotics *et al.*, 2016), provide a better parental care (Vergara & Aguirre, 2006;

Vergara *et al.*, 2006, 2010) return earlier from wintering areas and occupy best nests (Vergara & Aguirre, 2006; Vergara *et al.*, 2007b, 2010; Belabed *et al.*, 2019), explaining their better performance.

Beyond positive effects of foraging at landfills on hatching success probabilities at an individual level, we did not find any landfill use effects on productivity parameters. The number of eggs, nestlings, or fledgling success was similar among breeders differing in their landfill use. Other studies conclude that landfill use can increase productivity in white storks (Tortosa *et al.*, 2002, 2003; Djerdali *et al.*, 2008b, 2016b). However, these studies were conducted at the population level comparing different populations. Differences in foraging habits of storks breeding in different populations are probably much higher than the among individual differences within populations. Moreover, in our study area all individuals have access to a close landfill being able to increase landfill exploitation when other food resources are scarce. It may mask landfill benefit on other no critical reproductive traits (Steigerwald *et al.*, 2015).

While a higher use of landfills by breeders of the Prado Herrero colony resulted in short-term benefits in terms of hatching success, our results show a clearly negative impact on future offspring survival. The effect was especially noticeable during the first year of life when birds become independent of their parents. This first year of life is a critical period in which individuals need to grow and rapidly acquire the skills necessary to survive and early life conditions can make some individuals more prone to dying (Briga *et al.*, 2017). Parents foraging at landfills can provide better parental care and almost unlimited food but anthropogenic food remains are often poor quality food (Annett & Pierotti, 1999; Murray *et al.*, 2018). Thus, offspring from parents using landfills with high intensity could be receiving low quality food during early life or even later in live if they inherit the parental foraging preferences (Annett & Pierotti, 1999). Nutrition during early development can have important consequences in adulthood (Lindström, 1999; Krause *et al.*, 2009). For example, an antioxidant-rich diet contributes to reducing telomere attrition, and a lack of antioxidants on landfill diet may shorten life span (Tauler-Ametlller *et al.*, 2019). In addition, rooting food and health conditions in landfills stimulate presence of several pathogens (Matejczyk *et al.*, 2011). Storks transport these pathogens from landfill to nest on their body, nesting material or food and modifying the microbiota of the nest. In fact, several harmful and multidrug-resistant microorganism have been detected in stork populations near landfills (Höfle *et al.*, 2003, 2020; Camacho *et al.*, 2016; Martín-Maldonado *et al.*, 2020). Beyond this, toxic products as PCBs, flame retardants, metals, and metalloids are incorporated from a landfill diet (Sáez *et al.*, 2008; Saez *et al.*, 2009; De la Casa-Resino *et al.*, 2015a,b; Pérez-López *et al.*, 2016). These pollutants are accumulated on different bird tissues and can cause health problems in white storks (e.g. skeletal deformation)(Smits *et al.*, 2005, 2007). Therefore, the negative relationship between parental landfill use and juvenile survival may be a consequence of several potential risks associated to rubbish food resources (Peris, 2003; Henry *et al.*, 2011). Probably the first year of life acts as a filter in which mortality processes operate more intensively and because of this the effect of a poor-quality diet during early life is not evident on survival probabilities of older storks (Payo-Payo *et al.*, 2016).

Overall, our results show a compromise for parental fitness in using anthropogenic food resources: enhances hatching success but reduces juvenile survival probabilities. Although high levels of feeding on landfill lead to fitness costs, a low or moderate use of landfill improve the probabilities of having a surviving descendant by each egg laid. Thus, complementing natural diet by food resources available at landfills seems the optimal strategy based on results (Zorrozua *et al.*, 2020a). Therefore, breeders may increase food provisioning from landfill during adverse environmental conditions and/or when offspring necessities are higher.

In conclusion, our study shows the existence of trade-offs associated with parental landfill use in the white stork that influence parental fitness: low to medium levels of use are beneficial but very low or medium to high levels of use counterproductive. However, we think that, at population level in some populations, the current benefits that anthropogenic food supplies at landfills provide to white storks may exceed the costs in terms of offspring survival reduction. Superabundant food provisioning at landfills contributed to the growth of the Spanish white stork population (Blanco, 1996; Tortosa *et al.*, 2002). Although this population could be composed of suboptimal individuals (Genovart *et al.*, 2010) reproducing thanks to the extremely high food abundance (Steigerwald *et al.*, 2015), the population recovered and greatly reduced their extinction probabilities (Tortosa *et al.*, 2002; Molina & Del Moral, 2005). Traditional foraging areas of storks are degraded due to agricultural intensification and urbanisation (Barlein, 1991; Vitousek *et al.*, 1997). Under this scenario, and given that open air landfills, as an abundant and predictable food resource, are condemned to disappear soon due to European Union regulations (Landfill Waste Council European Directive 1999/31/CE and Directive 2018/850/EC) stork populations are expected to be heavily influenced. Management actions designed to compensate a potential future decline of stork populations could include the recovery and protection of natural traditional foraging areas , the reduction of non-natural mortality and temporal supplementary feedings areas to a soft transition between actual open landfills to new landfill management (Garrido & Fernandéz-Cruz, 2003; Molina & Del Moral, 2005; Tobolka, 2014; Martín *et al.*, 2018). Otherwise, avian population that feed in landfills, such as white storks, may have an uncertain future.

Appendix A

Table A.1: Number of pairs classified by age and year of breeding. We represent number of pairs of each age during our four breeding seasons. (*): In one of those 11 pairs, members of the pair differ by one year old; male was 3 years (4th calendar year) old and female was 2 years old (3rd calendar year). We retain the age of the older individual to do the analysis.

Age of pair	1999	2000	2001	2002
2 (3rd calendar year)			6	10
3 (4th calendar year)	1			11*
4 (5th calendar year)		3		
5 (6th calendar year)		2	4	
6 (7th calendar year)			2	2
7 (8th calendar year)	2		1	1
8 (9th calendar year)	1	1		1
9 (10th calendar year)			3	
10 (11th calendar year)			2	2
11 (12th calendar year)				1

Table A.2: Modelling of White Stork resighting probabilities (p). We tested if resighting probabilities varied as a function of age (first year migration different to older ages), field effort and time. We tested for additive (+) and interaction (.) effects between age classes. Survival probabilities were modelled as general as possible including differential parameters for juveniles (first year), young breeders (2-5 years old), and adults (6 years and onwards).

Model	Recapture hypothesis	np	Deviance	QAIC$_c$	ΔQAIC$_c$
1p	**$p_{2a+effort}$**	**6**	**516.25**	**491.66**	**0.00***
2p	$p_{2a.effort}$	7	516.23	493.75	2.09
3p	p_{effort}	5	535.25	507.21	15.55
4p	p_{2a+t}	22	508.76	520.45	28.79
5p	$p_{2a.t}$	24	507.62	524.18	32.52
6p	p_t	21	518.23	526.87	35.21
7p	p_{2a}	5	556.61	527.04	35.38
8p	p_c	5	556.88	527.29	35.63

Note: Terms are **np**, number of parameters; **QAICc**, Akaike information criterion corrected for overdispersion and small sample size; **ΔQAICc**, the QAICc difference between the current model and the one with the lowest QAICc value; **2a**, two age classes structure (first year vs two years onwards); **t**, time effect; **effort**, field effort effect; **c**, constant.

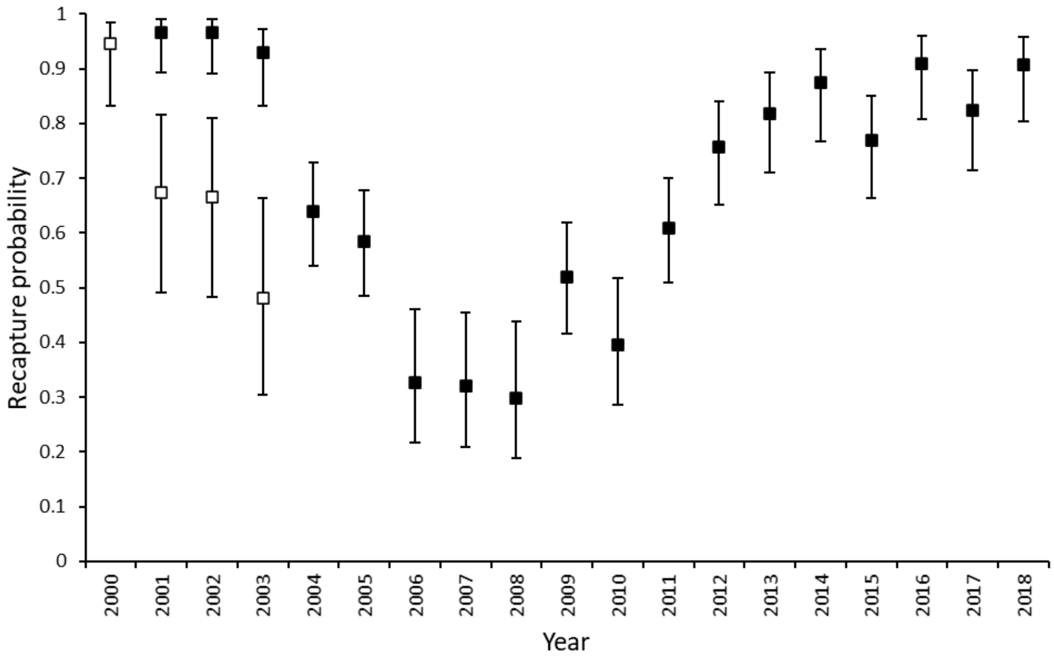
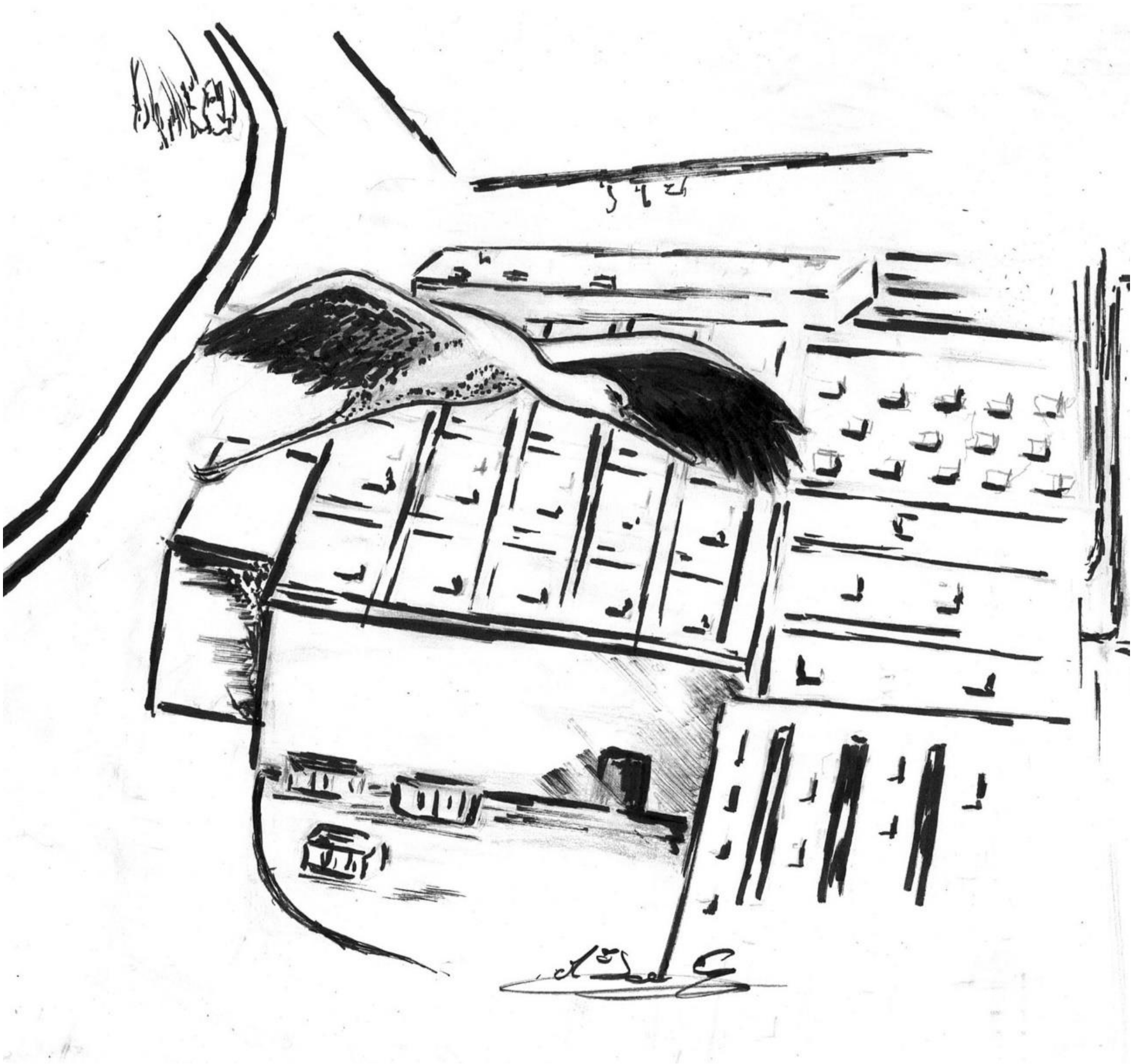


Fig. A.1: Variation in resighting probabilities depending on field effort along time and age (juveniles differ from adults on migration patterns). Black squares represent adult resighting; open squares represent juvenile resighting. Juveniles only include four years as we followed nestlings ringed in 1999 to 2002.

CHAPTER 4 - BACK TO THE FUTURE: LONG TERM EFFECTS OF LANDFILL CLOSURE ON BREEDING WHITE STORK (*CICONIA CICONIA*) POPULATIONS.



Based on: López-García, A., Gil-Tapetado, D. & Aguirre, J.I. *Under review*. Back to the future: long term effects of landfill closure on breeding white stork (*Ciconia ciconia*) populations.

Abstract

Organic waste production peaked following the urbanization and human development sprawl in the recent decades. This abundant and reliable anthropogenic food source has favoured several species, some of which consequently became overabundant. Landfills also present hazards to wildlife, which may suffocate on plastic materials, tangle on cords, and get exposed to pollutants and pathogens. In response to environmental and public health concerns over the maintenance of landfills, the European Commission proposed to close the landfills.

Our objective was to determine the impact of Landfill European Directive (Directive 2018/850/EC) on a bird species whose population recovery and growth was linked to landfill exploitation. We implemented species model distribution to project future population distribution in the absence of landfills. We included human population and emissions in low and high shared socioeconomic pathways. Given that protection measures were associated to population recovery, we also evaluated the overlapping degree between protected areas and projected distribution range.

Our models predicted a sharp decline in breeding population distribution, reaching values similar to the 1984 breeding census when the species was categorized as threatened. Our results also suggest a decrease in maximum nest density, possibly a consequence of reduced food availability. A more intense reduction in population distribution and nest density was predicted in the highest human development scenarios (ssp5), remarking the potential impact of climate change in this species.

However, measures such as gradual change in landfill management, continuous monitoring of breeding populations as well as evaluation of the conservation status of natural feeding areas before and after landfill closure should be considered.

Resumen

La producción de residuos orgánicos ha alcanzado su máximo tras el aumento de la urbanización y del desarrollo humano de las últimas décadas. Esta fuente de alimentación antrópica abundante y predecible ha favorecido a varias especies, algunas de las cuáles se han convertido en superabundantes. Además, los vertederos presentan peligros para la fauna, como: el ahogamiento por ingesta de plásticos, el enredo con cuerdas, y la exposición a contaminantes y patógenos. En respuesta a la preocupación ambiental, de salud pública y de sanidad animal que suponen los vertederos, la Comisión Europea ha propuesto la clausura de los vertederos.

Nuestro objetivo era determinar el impacto de la Directiva Europea de Vertederos (Directiva 2018/850/EC) en una especie de ave cuya recuperación y crecimiento poblacional está vinculado a la explotación de los vertederos. Se han implementado modelos de distribución de especies para proyectar la distribución futura de la población en ausencia de los vertederos. Se ha incluido el crecimiento de la población humana y de las emisiones tanto en escenarios de bajo como de alto desarrollo de trayectorias socioeconómicas compartidas. Dado que las medidas de conservación estuvieron asociadas con la recuperación de la especie, también se ha evaluado el grado de solapamiento entre las áreas protegidas y el rango de distribución proyectado.

Nuestros modelos predicen un descenso agudo en la distribución de la población reproductora, alcanzando valores similares a los de 1984, momento en que la especie fue catalogada como amenazada. Además, nuestros resultados sugieren una disminución en la densidad máxima de nidos, posiblemente debido a una reducción del alimento disponible. Se ha predicho una reducción más intensa en la distribución de la población y la densidad de nidos en los escenarios de mayor desarrollo (ssp5), remarcando el potencial impacto del cambio climático en esta especie.

No obstante, deberían ser consideradas medidas tales como el cambio gradual en la gestión de los vertederos, el seguimiento continuo de la población reproductora, así como la evaluación del estado de conservación de las áreas de alimentación naturales antes y después del cierre del vertedero.

Introduction

Human-induced changes have deeply transformed ecosystems and wildlife communities (Vitousek *et al.*, 1997). In addition to direct actions, such as landscape transformation or species translation, anthropogenic food sources are one of the most globally important factors impacting on animal biodiversity and distribution (Oro *et al.*, 2013; Newsome *et al.*, 2015). In developed countries, millions of tonnes of food remains generated by humans end up in landfills annually (Stenmarck *et al.*, 2016). Many animal species, particularly birds, have developed strategies to take advantage of this abundant and predictable in time and space food resource (Plaza & Lambertucci, 2017). Anthropogenic food provides numerous benefits to wildlife, including reduced energetic costs (Gilbert *et al.*, 2016; Soriano-Redondo *et al.*, 2021) and increased body size, reproductive parameters, and offspring survival (Tortosa *et al.*, 2002; Steigerwald *et al.*, 2015; López-García *et al.*, 2021), potentially resulting in a rapid population growth.

The competence process between and within species decreases in parallel with the abundance and reliability of resources (Restani *et al.*, 2001; Corman *et al.*, 2016). In addition to an increased reproductive success, wildlife is attracted to landfills and adjacent areas for both breeding and foraging, explaining alterations in distribution and resource selection of some populations (Duhem *et al.*, 2008). The continuous abundance of anthropogenic resources has also been correlated with alterations in migration patterns and seasonality. For instance, some migratory birds use anthropogenic sites, such as landfills, as important stop-overs, while others may shorten or suppress migratory behaviour (Flack *et al.*, 2016; Rotics *et al.*, 2017; Arizaga *et al.*, 2018). The increase in density of a particular species at these sites can promote competitive displacement of species, resulting in an ecological impact on trophic webs and homogenisation of local community composition (Malekian *et al.*, 2021).

All of these changes in avian distribution can translate to the aggregation of birds reaching abnormally high densities that usually produce conflicts with humans such as nuisance, damage on buildings or even health issues (Hatch, 1996; Belant, 1997; Vergara *et al.*, 2007a). Additionally, the risk of collision with power-lines and airplanes increase in the proximities of landfills (Garrido & Fernandéz-Cruz, 2003; Moreira *et al.*, 2018; Pfeiffer *et al.*, 2020; Marcelino *et al.*, 2021). The conflicts human-wildlife generally change the human perception of animals becoming them in the so-called pest species (Belant, 1997; Payo-Payo *et al.*, 2015).

Landfills are generally associated with several environmental risks, including soil contamination, greenhouse gases, hazardous emissions, and water pollution from runoffs (Butt *et al.*, 2008; Vaverková, 2019). Landfills also pose important challenges to wildlife. For instance, decomposing organic waste provides the optimal environment for a number of pathogens to proliferate (Plaza & Lambertucci, 2018; Tauler-Ametlller *et al.*, 2019; Martín-Maldonado *et al.*, 2020). Additionally, ingestion of plastics pose as a choke and injury hazard, while heavy metals and other pollutants may intoxicate birds (Peris, 2003; Henry *et al.*, 2011; Muñoz-Arnanz *et al.*, 2011).

As a consequence, the European Union passed environmental policies to reduce the production of human refuse, diminish the percentage of biodegradable waste at landfills, and transform the organic waste in compost or biofuel by 2030 (Landfill Waste Council European Directive 1999/31/CE and Directive 2018/850/CE). The new waste facilities will prevent wildlife access to refuse while the "old" open air landfills should be closed or substantially modified (Directive 2008/98/CE and Directive 2018/850/CE).

The recent landfill legislation may come at cost to wildlife. Landfills have been associated with recovery of multiple populations of threatened and endangered species (Tortosa *et al.*, 2002; Rumbold *et al.*, 2009; Tauler-Ametller *et al.*, 2017; Arnold *et al.*, 2021). For example, the Western European population of white storks (*Ciconia ciconia*) was suffering a generalised sharp decline between the 1950s and 1980s (Barlein, 1991). A total of only 6,753 white stork breeding pairs was reported by the National Census of 1984 in Spain, the lowest number of nesting pairs ever recorded (Lázaro *et al.*, 1986). The conservation campaign in the following years and the white stork's exploitation of landfills resulted in the rapid recovery of their breeding population in Western Europe (Schulz, 1999; Tortosa *et al.*, 2002; Massemin-Challet *et al.*, 2006). Although many studies discuss the potential effects of change in waste management on species that rely on landfills (See references in Plaza and Lambertucci 2017), only a few have explored the effects of landfill closure on yellow-legged gulls, *Larus michahellis* (Payo-Payo *et al.*, 2015; Delgado *et al.*, 2021; Pinto *et al.*, 2021). In this species, the absence of landfill forced them to shift diet preferences but the scarcity in alternative feeding sources negatively impacted their population dynamics (Steigerwald *et al.*, 2015; Zorrozua *et al.*, 2020a).

The use of species distribution models (SDM) allow scientists to identify environmental conditions which determine the spatial distribution of bird species (Thuiller, 2004). Climate variables and continued rapid urbanization are associated with habitat fragmentation, habitat loss, and altered prey distribution (Ancillotto *et al.*, 2016; Bradsworth *et al.*, 2017; Ashrafzadeh *et al.*, 2019).

We focused on the breeding population of white stork of Community of Madrid, Spain where number and location of main landfills were nearly constant over the last 35 years. Our main objective was to evaluate the potential future impact of landfill closure on a breeding population distribution and density in a species that heavily rely on these facilities under different climate change scenarios. We categorized both areas of high stork suitability and areas with potential habitat loss or gain in 30- (2050) and 50-year (2070) projections. We also evaluated the relevance of the studied anthropo-ecological variables (e.g., land cover class, distance to water bodies, human density, etc.) in our models. The combined use of both climate and habitat variables improves the predictive accuracy of SDMs (Barbet-Massin *et al.*, 2012). Finally, we predicted the effectiveness of protected areas to serve as breeding habitat.

Materials and Methods

Study area and data collection

The Autonomous Community of Madrid (central Spain) was surveyed between the first week of March to the end of June in 2021 (Fig 1). Nest were monitored based on methodology from previous National and European censuses (Schulz, 1999; Molina & Del Moral, 2005). We recorded nest location and occupancy during each visit. Most nests were visited on multiple occasions, but only one visit was possible in some locations. For the latter, we conducted these single visits during the peak of the breeding season (*i.e.,* end of April to middle of May) and considered nests to be occupied if breeding adults or nestlings were sighted. This method allows us to detect approximately 90% of breeding pairs within the nesting season (Aguirre and Vergara 2009). We assumed that our sampling effort is representative of the total white stork breeding population in this territory.

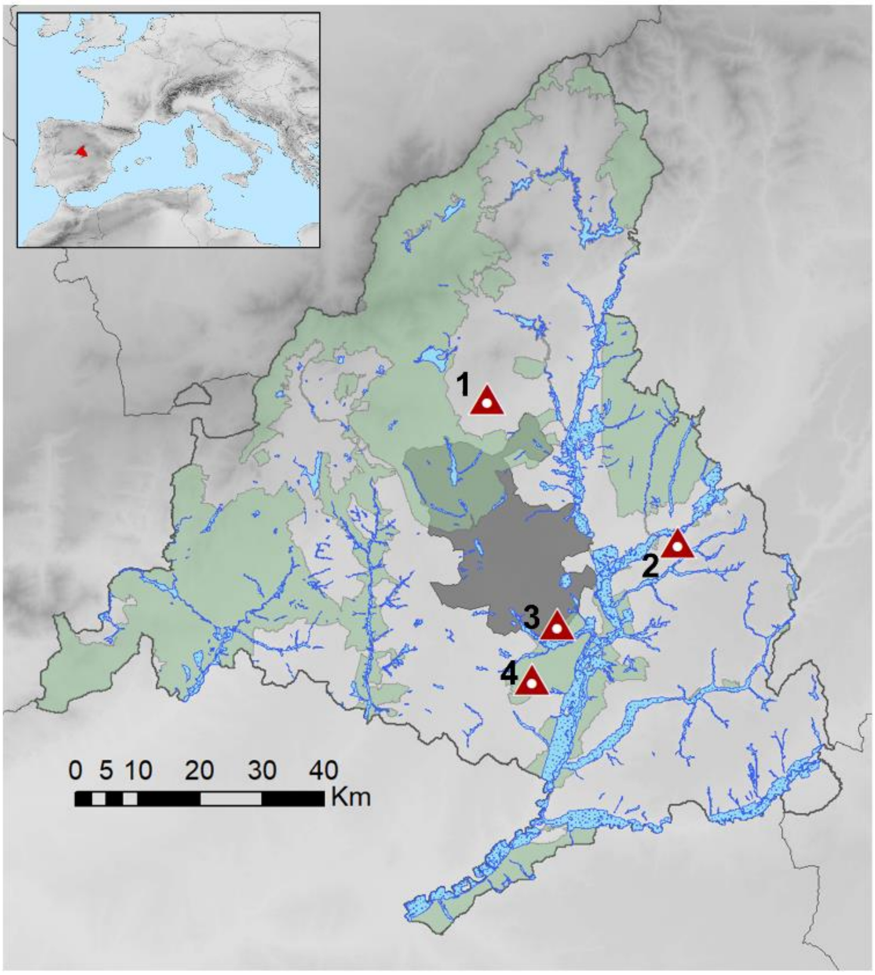


Fig. 1. Map of our study area, the Community of Madrid. Rivers and water bodies are represented in blue and the Municipality of Madrid by the dark grey area in the centre of the map. The protected natural areas of this region are marked in green. Landfills are represented by triangles and numbered: (1) Colmenar Viejo, (2) Alcalá de Henares, (3) Las Dehesas, and (4) Pinto.

Landfill locations were obtained from PRTR-España, the Spanish Registry of Emissions and Pollutant Sources (Table S1). During the breeding season of 2021, there were four active landfills in the study area (Alcalá de Henares, Colmenar Viejo, Pinto y Las Dehesas) (Fig. 1). White storks were frequently observed feeding in all of them. The first landfill was opened in 1978 within our study area, but white storks were not observed to forage at that location until the mid-1980s (Chozas, 1983; Blanco, 1996).

In response to our recent finding that landfill use altered nest-site selection in white storks (López-García *et al.* 2022 in review), we gathered locations of breeding population from the national census of 1984 (Lázaro *et al.*, 1986) to project the future distribution of white storks in absence of landfills.

Anthropo-ecological variables

Based on the previously acquired knowledge of this species on nest-site selection during the breeding season (Carrascal *et al.*, 1993; Radović *et al.*, 2015; Orłowski *et al.*, 2019; Hmamouchi *et al.*, 2020b; Bialas *et al.*, 2021), we selected a set of 9 environmental variables to estimate the potential distribution of breeding pairs (Fig S1). Corine Land Cover layers were reclassified into 7 land use classes (López-García et al 2022 in-press): urban areas (CLC class 1); arable fields (CLC class 21); other agricultural land (CLC class 22 and 24, except 244); pastures, meadows & agro-forestry areas (CLC class 231, 321 and 244); forests (CLC class 31 and 32, except 321); non-suitable habitat (CLC class 33); and inland waters (CLC class 41 and 51). We discarded forests, inland waters, and non-suitable areas to avoid redundance in our dataset. Urban areas were defined based on a human population density layer, *i.e.,* urban extents (UE) with a population density of ≥1000 persons/km^2 (Table S1). Distances were measured with a Distance matrix tool. Furthermore, we incorporated information of the current climate from WorldClim database version 2.1 (Table S1). We used a resolution of 30 arc-sec (grid cell of 1x1km size) for all the ecological and land use variables.

To ensure the independency of variables, we performing a hierarchical cluster analysis showing the similarity among all variables in a dendrogram, following the methodology presented by Polidori *et al.* 2021 (Dormann *et al.*, 2013). We used one of the most commonly suggested methods based on the correlation matrix, the Ward-clustering (Harrell, 2001) (Fig. S1). The distance-threshold to form the clusters was stablished at 0.3 (*i.e.,* less than 70% correlation). Among the 15 variables that passed this threshold, we chose the most derived variable in each cluster. Finally, we calculated a variance inflation factor (VIF) (Lin *et al.*, 2011) and eliminated redundant variables that overestimated the variance (VIF > 5) (Stine, 1995) (Fig S1). The final set of selected variables included 12 variables (Table S1, Fig S1).

We used QGIS 3.16.11 “Hannover” open-source software (QGIS Development Team, 2022).

Future variables

We gathered bioclimatic variables from WorldClim (CanESM5; https://www.worldclim.org/data/cmip6/cmip6climate.html) with 30 arc-sec resolution to explore changes in population dynamics in response to different climate change predictions. We selected a conservative and an extreme emission pathway (Shared Socioeconomic Pathways SSP1-2.6 and SSP5-8.5, respectively) for each scenario. Projections were calculated for two periods 2041–2060 (2050) and 2061–2080 (2070). Predictions are referred herein as 2050–2.6, 2050–8.5, 2070–2.6 and 2070–8.5 for simplicity. We also obtained population density and measured distance to urban areas from Global 1-km Downscaled Population Base Year and Projection Grids Based on the SSPs, v1.01 (Table S1). We used Corine Land Cover 2018 to assess potential distribution of breeding pairs in our projections.

Potential distribution procedure

We estimated the potential present distribution of white storks through a set of SDMs using the occurrence data of nests surveyed in 2021, where storks were known to use landfills. Also, to force the models to give importance to nest density in each of the 1x1 km grids, we considered the spatial contagion of occurrences. Because our sampling includes the total number of stork nests, this spatial contagion is not due to external variables (e.g., proximity to cities in surveys with photographic records) but to variables intrinsic to these birds and their nest-site selection. Thusly, the results of the SDMs provide suitability as a proxy for nest density. We performed six different algorithms using the *biomod2* library (Thuiller *et al.*, 2019): generalized linear model (GLM), generalized additive model (GAM), artificial neural network (ANN), Classification Tree Analysis (CTA), maximum entropy (MaxEnt) and Random Forest (RF).

The average ensemble model based on 34 iterations of these six algorithms (204 individual models) was used to predict the potential distribution of the species. The construction of background and pseudoabsences were based on a 1 km buffer from each nest based on the previous acquired knowledge of the species (Olsson & Bolin, 2014; Zurell *et al.*, 2018; Orłowski *et al.*, 2019).

Presence and pseudoabsence data were split in 75/25% to generate an external Area Under the receiver operating characteristic Curve (AUC) evaluation for the final models, independently of the internal AUC evaluation of each individual model generated by *biomod2*. The total of 204 individual models were tested and only models with AUC >0.7 were chosen (*i.e.,* good to excellent performance of the model following the scale of Thuiller (2003)). A final ensemble average model was then obtained. Finally, the final average model was evaluated though the external AUC test with 25% of the data. AUC value of the final model was 0.98, indicating an excellent discrimination capability. A cut-off value of the final model was also calculated (0.532) to establish the areas of presence (>0.532) and absence (<0.532) of white storks.

We projected stork distribution considering scenarios in which there were no landfills as a source of food under EU regulations. As the 2021 occurrences are *a priori* strongly influenced by landfills, we cannot use them without biasing resource selection. To emulate the conditions that storks would be subject to in the absence of landfills, we have used nest survey data of 1984, when these birds did not yet forage at landfills. We performed another ensemble SDM with these occurrences and with the same methodology commented above. Finally, we projected this model into four future scenarios with the variables concerning 2050 and 2070 (2050–2.6, 2050–8.5, 2070–2.6 and 2070–8.5) (see the Future variables section). AUC of this model was 0.98 and cut-off value was 0.547.

Statistical analysis

We use the function "BIOMOD_RangeSize" to evaluate change in our distribution. This function determines percent potential changes in habitat and range. We also considered the overlap between present and future distributions and the protected areas in Madrid. We considered the Peñalara National Park and Natura 2000 areas as protected areas (Fig. 1).

We performed a Generalized Linear Model (GLM) with binomial distribution to determine differences in nest site selection (presence vs absence) in relation to environmental variables in the predicted model of 2021. To evaluate significant differences in suitability as a response to environmental variables, we performed a Linear Model with Gaussian distribution.

All statistical and spatial analysis were performed on R v 3.5.0 program (R Development Core Team, 2015) using RStudio Software v 1.1.453 (RStudio Team, 2015).

Results

Changes in Suitability

Our methodology included all nests observed in each grid cell, allowing us to stablish a relationship between suitability and nest density in our projections. Suitability (in this case, a proxy of nest density) decreased in areas far from water bodies but close to urban sites and high human population density in all our projections (Table 1). Areas with high presence of power lines, higher percentage of agriculture fields and arable lands, higher forest coverage, lower percentage of pastures & agro-forestry, lower isothermality, and higher precipitation were also associated with reduced suitability (Table 1). In addition, average temperature of the warmest annual quarter had a significant negative impact on suitability in the extreme emission pathway, ssp5 (Table 1). As a result, projected higher suitability areas were located in floodplain of main rivers (Jarama, Henares, Manzanares and Tajo) and valleys of the Northern region and the West municipalities of Madrid, which are associated with livestock (Fig. 2).

Areas with higher habitat suitability/nest density in 2021 were estimated to show the highest decline in nest density based on our projections: 2050ssp1 (Est = -0.013 ± 0.0004, t = -31.82, $p < 0.0001$), 2050 ssp5 (Est = -0.013 ± 0.0004, t = -34.05, $p < 0.0001$), 2070ssp1 (Est = -0.013 ± 0.0004, t = -32.23, $p < 0.0001$), 2070ssp5 (Est = -0.013 ± 0.0004, t = -34.27, $p < 0.0001$). Therefore, our projection points out that nest density also declines in future (Fig. 2).

Table 1. Predicted suitability variation considering different environmental variables under each respective climate-change scenario. In our projections, suitability is correlated with nest density.

	Estimate	SE	t value	p	
2050SSP1					
Water distance	-0.001	0.001	-48.775	< 0.0001	***
Power lines	0.002	0.001	9.644	< 0.0001	***
Agri	-0.090	0.004	-21.205	< 0.0001	***
Arable	-0.123	0.003	-45.552	< 0.0001	***
Index forest	-0.151	0.006	-23.697	< 0.0001	***
Pasture & AF	0.028	0.003	9.496	< 0.0001	***
Urban distance	-0.001	0.001	-33.216	< 0.0001	***
Human dens	-0.002	0.001	-29.331	< 0.0001	***
bio3	1.002	0.096	10.398	< 0.0001	***
bio5	-0.023	0.088	-0.261	0.794	
bio19	-0.017	0.002	-7.886	< 0.0001	***
2050SSP5					
Water distance	-0.001	0.001	-43.818	< 0.0001	***
Power lines	0.001	0.001	8.465	< 0.0001	***
Agri	-0.056	0.004	-14.451	< 0.0001	***
Arable	-0.093	0.002	-37.476	< 0.0001	***
Index forest	-0.118	0.006	-19.988	< 0.0001	***
Pasture & AF	0.033	0.003	12.103	< 0.0001	***
Urban distance	-0.001	0.001	-37.163	< 0.0001	***
Human dens	-0.002	0.005	-33.776	< 0.0001	***
bio3	1.113	0.098	11.408	< 0.0001	***
bio5	-0.277	0.083	-3.326	< 0.001	***
bio19	-0.018	0.002	-8.711	< 0.0001	***
2070SSP1					
Water distance	-0.001	0.001	-46.248	< 0.0001	***
Power lines	0.001	0.001	9.152	< 0.0001	***
Agri	-0.075	0.004	-18.27	< 0.0001	***
Arable	-0.116	0.003	-44.126	< 0.0001	***
Index forest	-0.143	0.006	-22.944	< 0.0001	***

Pasture & AF	0.031	0.003	10.808	< 0.0001	***
Urban distance	-0.001	0.001	-33.194	< 0.0001	***
Human dens	-0.002	0.001	-31.616	< 0.0001	***
bio3	0.719	0.096	7.459	< 0.0001	***
bio5	0.096	0.085	1.120	0.263	
bio19	-0.019	0.002	-9.474	< 0.0001	***
2070SSP5					
Water distance	-0.001	0.001	-38.505	< 0.0001	***
Power lines	0.001	0.001	9.671	< 0.0001	***
Agri	-0.052	0.004	-13.945	< 0.0001	***
Arable	-0.073	0.002	-30.837	< 0.0001	***
Index forest	-0.098	0.007	-17.294	< 0.0001	***
Pasture & AF	0.035	0.003	13.355	< 0.0001	***
Urban distance	-0.003	0.001	-38.42	< 0.0001	***
Human dens	-0.002	0.001	-39.513	< 0.0001	***
bio3	1.395	0.104	13.41	< 0.0001	***
bio5	-0.602	0.086	-7.013	< 0.0001	***
bio19	-0.013	0.002	-6.461	< 0.0001	***

Note: Terms are **Water dist,** distance to water bodies; **Power lines,** length of the power lines; **Agri,** percentage of agricultural crops, **Arable,** percentage of arable lands; **Index forest,** coverage of forest and density of trees; **Pasture & AF,** percentage of pastures and agro-forestry areas; **Urban distance,** distance to urban areas; **Human dens,** density of human population; **bio3,** Isothermality; **bio5,** Average temperature of the warmest annual quarter; **bio19,** precipitation of the coldest annual quarter**.**

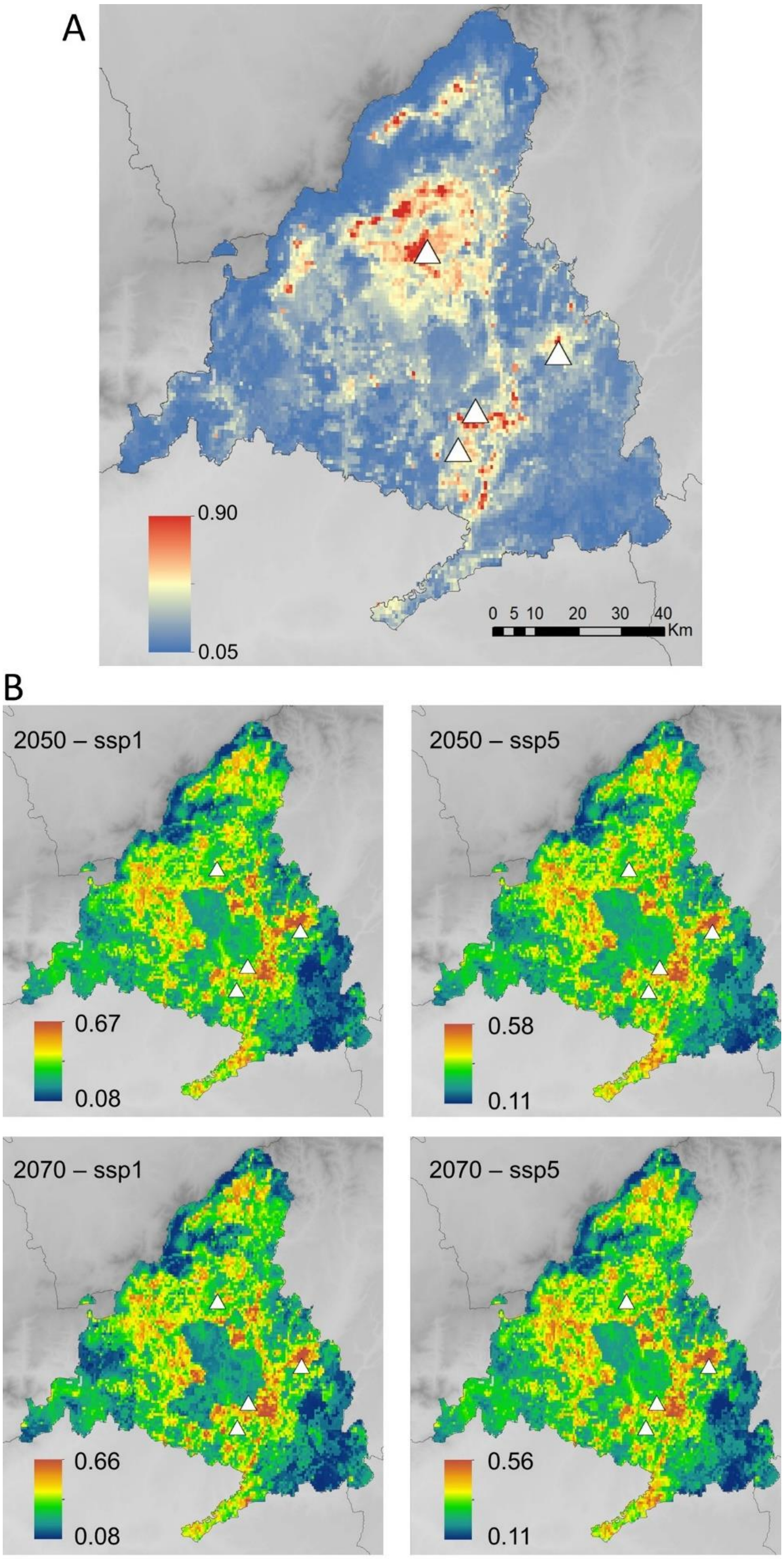


Fig. 2. Habitat suitability in 2021 **(A)** and in future scenarios **(B)**. In our projected models, suitability is a proxy of nest density. Therefore, maps **(B)** show a decrease in the maximum nest density values in future, which are even more pronounced in extreme emission pathway scenarios (ssp5). Landfills are represented by white triangles. High density is represented in red and low density in blue.

Nest-site selection in 2021

The ensemble model for 2021 distribution clearly revealed that the most important variable is landfill distance (Fig. 3). Other relevant variables influencing nest-site selection are distance to urban areas, distance to water bodies, percentage of forest cover, and climatic variables (lower isothermality, average temperature of the warmest annual quarter, and higher precipitation) (Fig. 3).

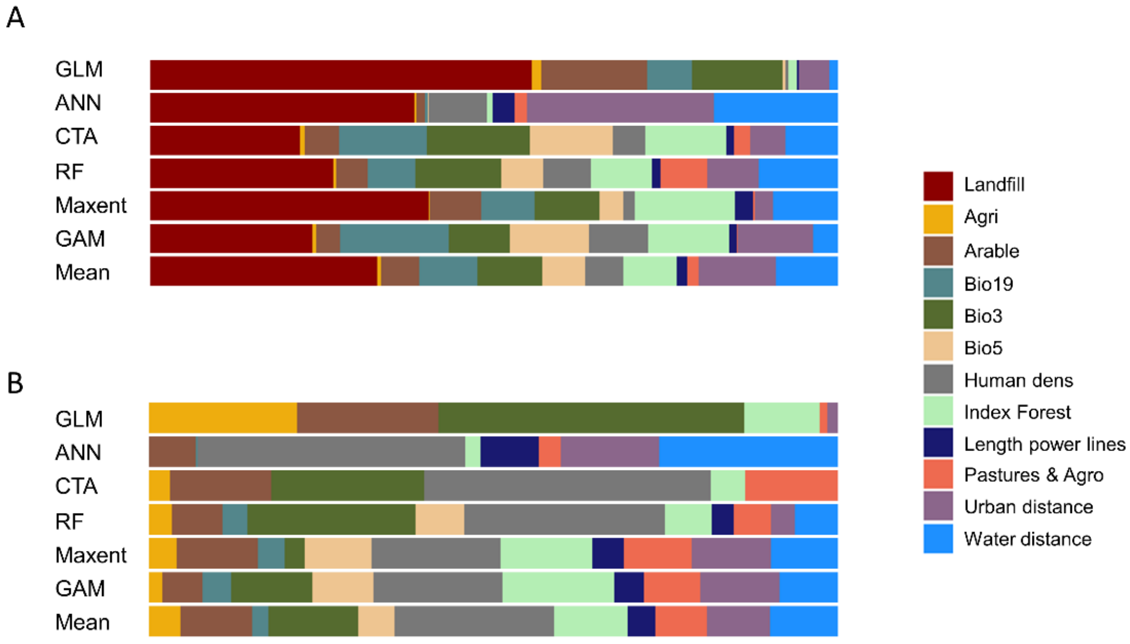


Fig. 3. Importance of the variables considered in our models: in 2021 (A) and in 1984 (B) with different algorithms (GLM, general linear model; artificial neural network (ANN), classification tree analysis (CTA), Random Forest (RF), maximum entropy (MaxEnt), generalized additive model (GAM), and the ensemble model (Mean)). Landfill is the most important variable independently of the algorithm that we use. Terms are **Landfill,** distance to the nearest landfill; **Water distance,** distance to water bodies; **Length power lines,** length of the power lines; **Agri,** percentage of agricultural crops, **Arable,** percentage of arable lands; **Index Forest,** coverage of forest and density of trees; **Pasture & AF,** percentage of pastures and agro-forestry areas; **Urban distance,** distance to urban areas; **Human dens,** density of human population; **bio3,** Isothermality; **bio5,** Average temperature of the warmest annual quarter; **bio19,** precipitation of the coldest annual quarter**.**

White storks significantly prefer areas near landfills (Est = -0.002 ± 0.0001, Wald = -4.877, $p < 0.0001$), near water bodies (Est = -0.001 ± 0.0001, Wald = -3.958, $p < 0.0001$), with high percentage of pastures & agro-forestry (Est = 0.014 ± 0.004, Wald = 3.223, $p < 0.0001$), lower percentage of arable lands (Est = -0.029 ± 0.005, Wald = -5.52, $p < 0.0001$), higher isothermality (*i.e.,* day-to-night temperatures oscillations relative to the summer-to-winter annual oscillations; bio3, Est = 1.178 ± 0.244, Wald = 7.026, $p < 0.0001$), lower maximum temperature (bio5, Est = -0.227 ± 0.101, Wald = -2.257, $p = 0.024$) and lower precipitation in the coldest quarter (bio19, Est = -0.046 ± 0.017, Wald

= -2.655, p < 0.0001) (Fig 4). Moreover, our ensemble model seems to show that white storks avoided urban areas (Est = 0.0001 ± 0.001, Wald = 3.88, p = 0.008) in 2021 (Fig 4). None of the other environmental variables were significant (p>0.05).

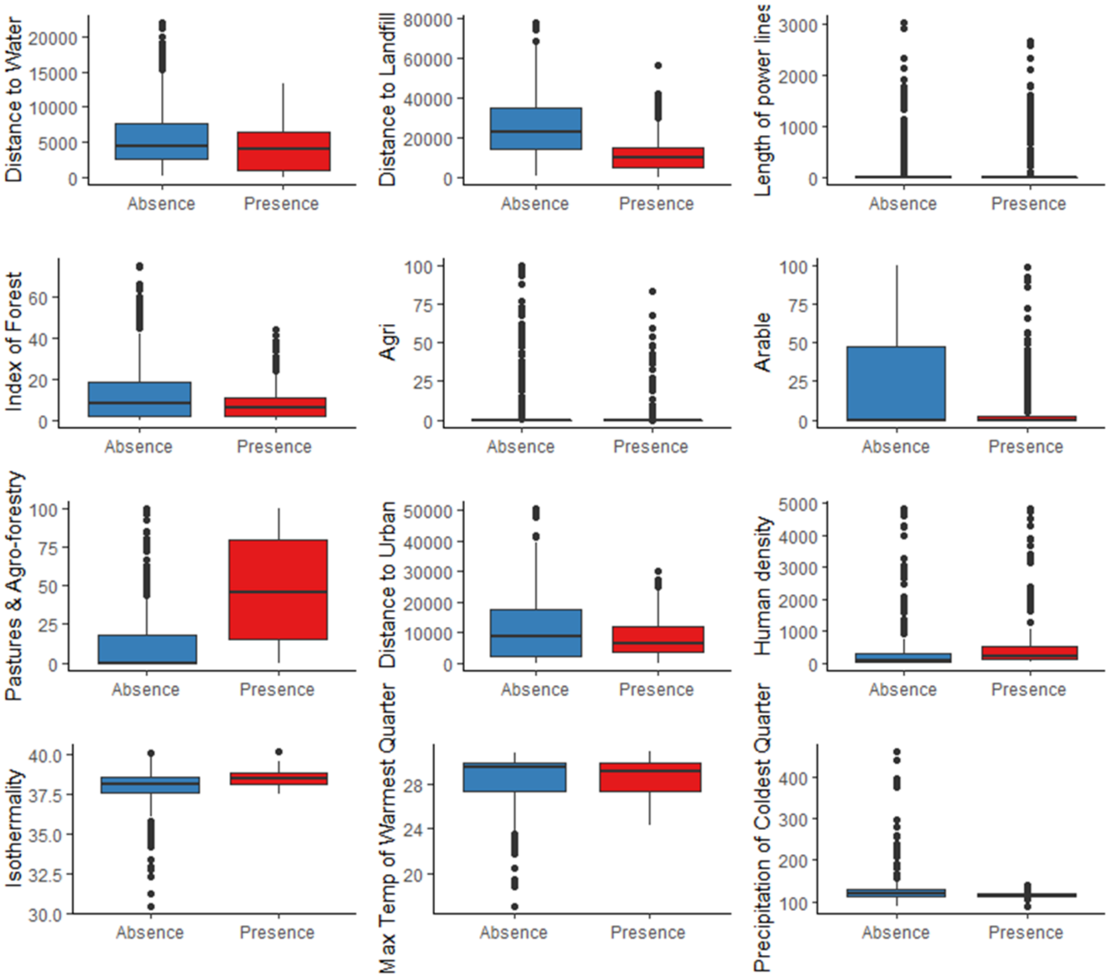


Fig. 4. Differences in the environmental variables between the presence points and absence points. Outliers with a value more than 1.5 times the interquartile range are shown as circles.

However, the most relevant variables in 1984 were human population density (39.84%), isothermality (22.35%), forest (18.27%), arable lands (17.78%), distance to water bodies (17.04%) and pastures & agro-forestry (12.73%).

Range distribution loss

The actual breeding population distribution of white storks in 2021 is 321 grid cell (321 km^2) and the model prediction estimation is 289 grid cell (289 km^2) in 2021. The range of distribution in our models predicted a large reduction in the breeding population in all future scenarios. For instance, the distribution range diminish by 56.40% in 2050ssp1 and 70.16% in 2070ssp1 (Fig. 5). Pronounced breeding populations declines were predicted in the extreme emission pathway scenario, declining by 77.16% in 2050ssp5 and 92.73% in 2070ssp5 when compared to distribution ranges from 2021 (Fig. 5). Both 2050ssp5 and 2070ssp5 highlight the high impact of climate change on white stork breeding populations and are even lower than the smallest distribution range recorded for this species (in 1984), when breeding area was 73.36% smaller than in 2021.

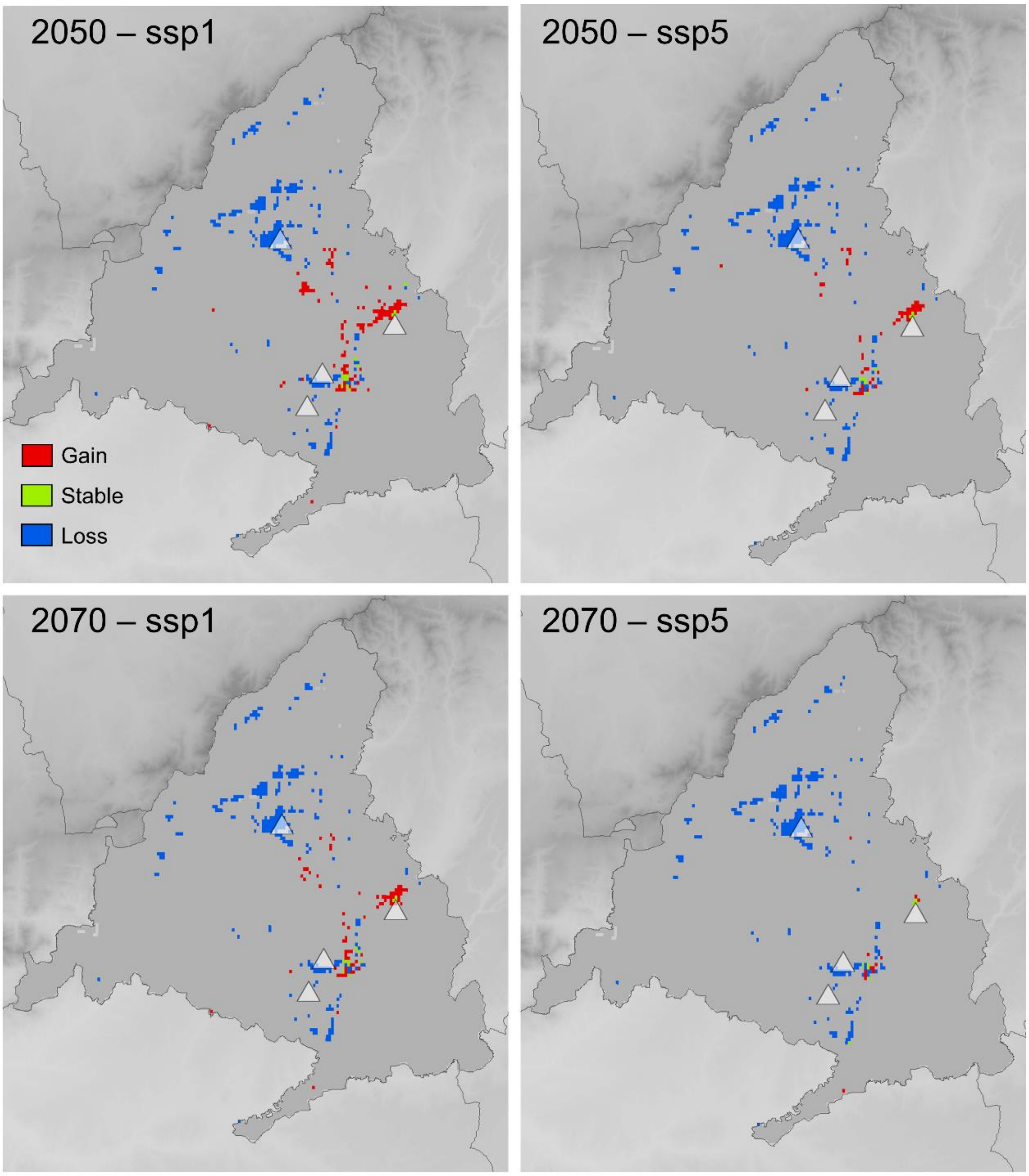


Fig. 5. Changes on potential breeding population distribution of White stork in Community of Madrid in absence of food availability from landfills and different climate change future scenarios (ssp1 and ssp5). Landfills are represented by white triangles.

The model projections also showed an interesting spatial pattern. Loss of potential distribution range mainly occurred in the nearby area of three out of the four major landfills of Community of Madrid (Colmenar Viejo, Pinto and Las Dehesas). In fact, the potential distribution range in the nearby area of Alcala de Henares (East landfill) increased in 2050ssp1, 2050ssp5 and 2070ssp1 when compared to Henares River (Fig. 5). Madrid municipality remained unoccupied.

Protected areas

In 2021, more than a half of the predicted breeding areas overlapped with protected areas (62.28%). While the potential nest distribution area decreased in our projections (see above), the percentage of potential distribution of breeding areas overlapping with protected areas in 2050ssp1 (69.05%), 2050ssp5 (65.15%) and 2070ssp1 (63.95%) remained similar to population distribution in 2021. However, the percentage of overlapping with protected areas decrease to 42.86% in the extreme emission pathway scenario in 2070 (*i.e.,* 2070ssp5).

Discussion

In this study, we explored potential consequences of landfill closure on the breeding population distribution of an avian species that heavily relies on landfills under different climate change scenarios and accounted for progressive urbanization process. Our projections showed a significant reduction in population distribution and a decline in the nest density in absence of landfills.

Our models resonated with predictions from previous studies (Duhem *et al.*, 2008; Tauler-Ametller *et al.*, 2017; Bialas *et al.*, 2021), supporting the importance of distance to landfill in nest-site selection above every other variables. This remarks the role of landfills as reliable and abundant food resources during breeding season (López-García et al 2021).

Several processes potentially underly the predicted reduction in stork population distribution and nest density in the absence of landfills. As a consequence of landfill closure, breeding pairs may increase range size to cope with the decrease in food abundance (Zurell *et al.*, 2018) while food uncertainty reduces feeding success rate (Cowie, 1977; Olsson & Bolin, 2014). Larger home ranges along with food uncertainty imply an increase in energetic demands and less time devoted to parental investment

(Schoener, 1971). Lower feeding rates may have negative effect on body condition and worse nourished of both adults and nestlings (Tauler-Ametlller *et al.*, 2019; Pineda-Pampliega *et al.*, 2021), impacting adults survival and breeding success (Steigerwald *et al.*, 2015; Delgado *et al.*, 2021). Unsuccessful breeding pairs would likely progressively replace original nesting areas with areas with higher availability of food resources (Vergara *et al.*, 2006). Secondly, the absence of extra food-supply promotes intra-specific competition and territoriality (Hixon, 1980), restricting breeding opportunities for young breeders and poor quality individuals (Vergara *et al.*, 2007b). In fact, we assessed strongest declines of future suitability in areas near landfills with higher nest density in 2021. Thirdly, this sudden change in anthropogenic food resources was described to directly lead to diet shifts in gulls (Zorrozua *et al.*, 2020a; Spelt *et al.*, 2021). The increase in pressure over natural prey may produce temporary unbalance of the system with potential ecological consequences such as local extinctions of some prey species.

In addition to the effect of landfill closure, our projections show that climate change can have dramatic consequences in future breeding population distribution. Future climate predictions indicate extreme temperatures, lower precipitation, marked seasonality, and extreme climatic phenomena (e.g., droughts and frost events) which may relocate European birds to more temperate areas (Barbet-Massin *et al.*, 2012). According to this, our study region, which is characterized by a Mediterranean climate, would likely encourage white storks to shift their distribution towards areas with higher isothermality (*i.e.,* showing the thermal 'stability' of a region relative to annual variations in temperature), such as floodplains or valleys as our results signalled. These geographic regions overlap with areas typically preferred by white storks, and they should be considered to be protected to preserve this species (Carrascal *et al.*, 1993; Nowakowski, 2003; Radović *et al.*, 2015). Higher maximum temperatures (*i.e.,* bio5) and short periods of overabundant precipitation (*i.e.,* bio19) have previously demonstrated to have a direct negative impact on body condition and survival of nestlings (Carrascal *et al.*, 1993; Jovani & Tella, 2004; Fasolă-Mătăsaru *et al.*, 2018).

Climate change also indirectly impacts population dynamics through variation in abundance and distribution of food resources whereas land-use changes may decrease most suitable feeding areas. While our study population showed consistent preference for breeding areas with pasture and agro-forestry and near water bodies similar to other European populations (Radović *et al.*, 2015; Zurell *et al.*, 2018; Bialas *et al.*, 2020), several studies forecasted a decrease in pastures in the future mainly due to the scarcity of water and land-use changes (Riahi *et al.*, 2017; Fitton *et al.*, 2019). Furthermore, scarce precipitation may reduce several typical prey species of white storks (e.g., amphibians, earthworms or insects) (Markovic *et al.*, 2014; Wessely *et al.*, 2017; Fourcade & Vercauteren, 2022). However, the ability of generalist species to exploit new food resources, as invasive species, may mitigate landfill closure and climate change effects in some areas of the breeding distribution (Negro *et al.* 2014).

Our models showed that increasing urbanization restricted the already limited white stork population distribution in future scenarios. Even when white storks show some tolerance to human presence, as demonstrated by their habit of constructing nests on buildings, this species is not typically found in high populated urban areas (Hmamouchi *et al.*, 2020b). High human densities induce physiological stress in storks, and the high percentage of impervious cover soil surface in massive cities difficult their access to food resources (Garroway & Sheldon, 2013; Lowry *et al.*, 2013; Blas *et al.*, 2018).

Conservation measures effectivity have been diluted as white stork population have increased in the last decades (Unpublished data). Deterrent devices as well as nest removal have been promoted or, at least, treated with permission by administration entities (Garrido & Fernandéz-Cruz, 2003; Vergara *et al.*, 2007a; Moreira *et al.*, 2018). It is particularly noticeable that current protected areas overlap with a significant percentage of the population distribution but not in 2070ssp5 scenario, when projected population distribution is more restricted. This highlights the potential risk of white storks to become endangered again.

Future research should monitor diet shifts in this species after landfill closure. At a population level, one consideration should be taken into account. We considered the entire population as a whole but several studies have demonstrated diet specialization at an individual level in generalist species (Araújo *et al.*, 2011; Sanz-Aguilar *et al.*, 2015).

Thus, not all breeders may suffer the consequences of landfill closure at the same level with potential effects in population structure and population genetics.

This study provides relevant information that can be use in strategic planning for the management of wildlife which rely on landfills. Landscape-scale planning for restoration and conservation of wetlands as well as pastures and meadows should be a priority in areas adjacent to landfills in addition to a gradual landfill closure process. Furthermore, periodical monitoring programs of breeding populations along with GPS tracking of fledglings and breeding individuals are excellent tools to evaluate population dynamics and determine changes on nest-site selection (Rodríguez *et al.*, 2012; Bouten *et al.*, 2013).

Conclusions

In summary, our model projections of white stork breeding population showed a strong reduction in range of distribution and density of this species in all future scenarios in the absence of landfills. This reduction become a dramatic forecast when we consider a full contamination scenario and human population expansion. However, landfill closure does not necessarily imply the collapse of white stork population. Gradual changes in landfill management, protection of natural feeding areas, and a reduction in environmental pollution and greenhouse gas emissions contribute to white stork conservation and would likely allow their population to adjust to landfill closures with no severe declines.

Supplementary Material

Table S1. Description of variables included in our models, including the year and source of the dataset used.

Variable	Name	Source/Reference	Date of datasets
Distance to nearest landfill	Landfill	https://en.prtr-es.es/Informes/InventarioInstalacionesIPPC.aspx	-
Distance to nearest water body	Water distance	https://land.copernicus.eu/pan-european/corine-land-cover	1990 & 2018
Percentage of other agricultural fields	Agri	https://land.copernicus.eu/pan-european/corine-land-cover	1990 and 2018
Percentage of Arable fields	Arable	https://land.copernicus.eu/pan-european/corine-land-cover	1990 and 2018
Percentage of Pastures, grassland and agro-forestry areas	Pastures & Agro	https://land.copernicus.eu/pan-european/corine-land-cover	1990 and 2018
Tree cover density	Index forest	https://land.copernicus.eu/pan-european/high-resolution-layers/forests/tree-cover-density/status-maps/tree-cover-density-2018	2018
Human population density	Human dens	Gridded Population of the World (GPW), v3 and v4 (https://sedac.ciesin.columbia.edu/data/collection/gpw-v4)	1990 and 2020
Distance to nearest urban area	Urban distance	Calculated based on human density layer	1990 and 2020
Length of total power lines in the grid cell	Length power lines	National Topographic Layer 1:100.000 (http://centrodedescargas.cnig.es/CentroDescargas/catalogo.do?Serie=MAUT)	2015
Isothermality, *i.e.* day-to-night temperatures oscillations relative to the summer-to-winter annual oscillations	Bio3	http://www.worldclim. org	1970-2000
Maximum temperature of warmest month	Bio5	http://www.worldclim. org	1970-2000
Precipitation of coldest quarter	Bio19	http://www.worldclim. org	1970-2000

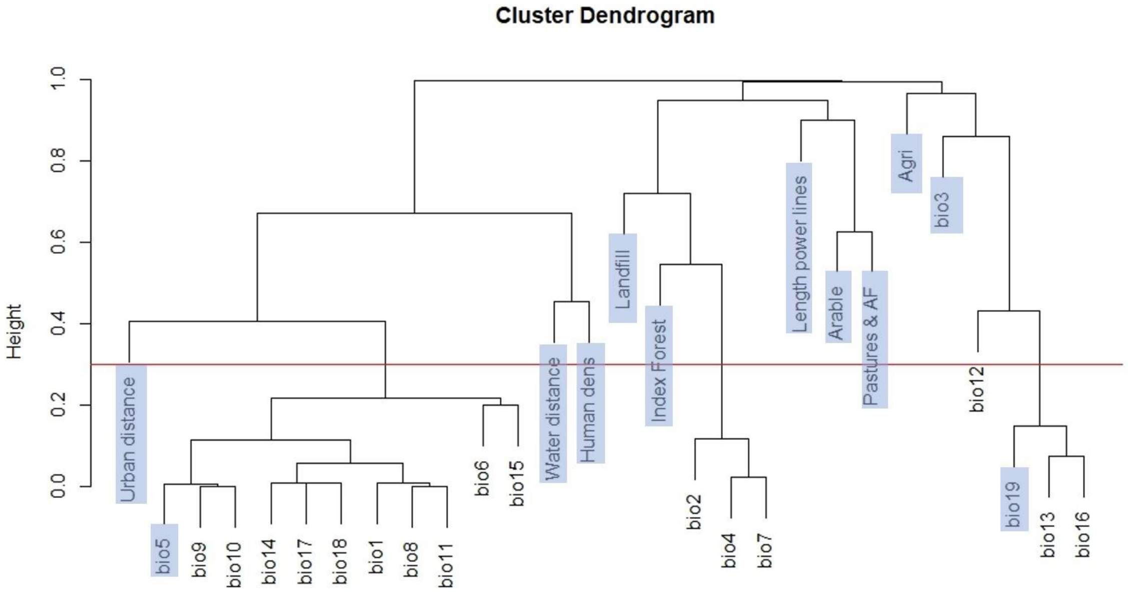


Fig. S1. Dendrogram obtained from the cluster analysis that was employed to select the relevant variables (highlighted in blue); the red horizontal line indicates the distance-threshold to form the clusters (0.3, *i.e.,* less than 70% correlation). **Note:** Terms are **Landfill dist,** distance to landfill; **Distance water,** distance to water bodies; **Length power lines,** length of the power lines; **Agri,** percentage of agricultural crops, **Arable,** percentage of arable lands; **Index forest,** coverage of forest and density of trees; **Pasture & AF,** percentage of pastures and agro-forestry areas; **Urban dist,** distance to urban areas; **Human dens,** density of human population; **bio3,** Isothermality; **bio5,** Average temperature of the warmest annual quarter; **bio19,** precipitation of the coldest annual quarter**.**

DISCUSIÓN GENERAL

En esta tesis se ha recopilado información de datos históricos de las poblaciones de cigüeña blanca en la Comunidad de Madrid, combinados con datos actuales con el fin de desentrañar los efectos de los vertederos sobre diversos aspectos relacionados con la población reproductora y sus parámetros de reproducción. En concreto: la distribución espacial de dicha población, los cambios en la selección del lugar de nidificación, los efectos a corto y largo plazo en la descendencia y la perspectiva futura de la población en ausencia de vertederos bajo diferentes escenarios de cambio climático. Para ello, se ha partido del nivel poblacional, determinando el impacto de los vertederos en la distribución de la población en tres momentos separados por un intervalo similar: cuando la población aún no se alimentaba en los vertederos (1984), primer censo en el que se considera que la población se ha recuperado (2004), y la situación actual (2021).

Los resultados de esta tesis confirman el impacto positivo de los vertederos en la población reproductora occidental de cigüeña blanca, en línea con estudios anteriores (Tortosa *et al.*, 2002; Massemin-Challet *et al.*, 2006; Djerdali *et al.*, 2016a; Soriano-Redondo *et al.*, 2021). El número de parejas reproductoras en la región de Madrid prácticamente se ha multiplicado por diez desde que su población empezó a utilizar los vertederos, tal como señalan los distintos censos realizados en esta especie (López-García & Aguirre; Chozas, 1983; Lázaro *et al.*, 1986; Aguirre & Atienza, 2002; Molina & Del Moral, 2005). Este hecho, por sí sólo, no demuestra que los vertederos estén relacionados con el incremento de la población a lo largo de estas casi cuatro décadas, puesto que existen otras circunstancias coetáneas, como las medidas de conservación aplicadas y los cambios en el uso del suelo (Schulz, 1999; Molina & Del Moral, 2005), que han podido afectar al crecimiento de la población. No obstante, este incremento no ha sido uniforme. La ingente cantidad de desperdicios orgánicos depositada diariamente en los vertederos atrae a un gran número de individuos de esta especie, entre otras (Plaza & Lambertucci, 2017; Arizaga *et al.*, 2018; Arnold *et al.*, 2021). Además, los resultados muestran que no sólo las parejas reproductoras crían cada vez más cerca de los vertederos, sino que la probabilidad de encontrar un nido nuevo es mayor cuanto más cerca del vertedero se sitúe. Al nidificar más cerca de los vertederos, se reduce, aún más si cabe, el bajo coste energético asociado con el forrajeo (Soriano-Redondo *et al.*, 2021), lo que permite incrementar el tiempo que estos individuos dedican a otras tareas, como el cuidado de los pollos o reforzar el vínculo con la pareja (Moritzi *et al.*, 2001) (Capítulo 1).

La agregación de nidos, en términos de densidad, también es mayor en las proximidades de estas instalaciones. De acuerdo con estudios previos, la densidad de nidos en esta especie parece estar directamente relacionada con la abundancia de alimento disponible (Carrascal *et al.*, 1993; Hilgartner *et al.*, 2014; Djerdali *et al.*, 2016b). Este efecto se ha observado en múltiples especies, en las que los individuos presentan menor competencia inter e intra-específica y, por tanto, mayor tolerancia a individuos conespecíficos en su área de campeo (Restani *et al.*, 2001; Gilchrist & Otali, 2002; Bino *et al.*, 2010). De este modo, disminuye la territorialidad y defensa del nido que realiza la pareja durante el periodo reproductor, (Redondo *et al.*, 1995; Bocheński & Jerzak, 2006) permitiendo que se establezcan otras parejas en las proximidades, lo que aumenta la agregación de parejas entorno a los vertederos. En consecuencia, se favorece la formación de colonias cerca de los vertederos mientras que las cigüeñas situadas en zonas alejadas suelen criar de forma solitaria al estar limitadas por las fuentes de alimentación naturales (Rolland *et al.*, 1998).

La obtención de alimento en los vertederos también ha tenido un impacto indirecto en la selección de los lugares de nidificación. Se ha observado que la selección del lugar de nidificación ha variado desde que las cigüeñas comenzaron a usar los vertederos (Capitulo 1, Capítulo 2, Capítulo 4). Puesto que es crucial para el desarrollo y la supervivencia de los pollos, los recursos tróficos disponibles en el área circundante son una señal de la calidad del hábitat. Sin embargo, este efecto se ve atenuado por la proximidad de los vertederos, que proporcionan una fuente de alimento abundante y predecible en el tiempo y en el espacio. A medida que la población de cigüeña blanca ha ido creciendo y se ha habituado al uso del vertedero, los nuevos lugares de nidificación se sitúan con mayor probabilidad en zonas próximas al vertedero pero, habitualmente, a expensas de la calidad del hábitat circundante al área de nidificación (Capítulo 1, Capítulo 2). Este cambio no sólo se observa a lo largo del tiempo, sino que se encuentran diferencias en la tipología del hábitat circundante al nido en función de la distancia al vertedero (Capítulo 2). Las parejas que nidifican más próximas al vertedero muestran mayor tolerancia hacia un mayor porcentaje de áreas con escasa o nula disponibilidad trófica (áreas urbanas y tierras de labor) en las zonas colindantes a su nido, lo cual podrían compensar con una mayor intensidad del uso de los vertederos (Gilbert *et al.*, 2016; Soriano-Redondo *et al.*, 2021).

Por el contrario, las cigüeñas que crían lejos del vertedero muestran una escasa presencia de estos hábitats pobres en recursos tróficos y lugares de nidificación (Zurell *et al.*, 2018; Orłowski *et al.*, 2019; Hmamouchi *et al.*, 2020b) mientras que muestran una fuerte selección a favor de los pastos y dehesas en las zonas próximas a su nido (Capítulo 2). Aun así, en general, se mantiene la preferencia observada típicamente en esta especie por zonas de nidificación próximas a masas de agua y zonas de pastos, praderas inundables y dehesas a lo largo de los años (Alonso *et al.*, 1991; Carrascal *et al.*, 1993; Radović *et al.*, 2015; Zurell *et al.*, 2018; Bialas *et al.*, 2020, 2021; Carlson *et al.*, 2021) destacando la importancia de estos hábitats para esta especie.

Éxito reproductor

Precisamente el porcentaje de pastos, praderas inundables y dehesas, en el área circundante a los nidos parece tener un efecto importante en la productividad de esta especie y, por lo tanto, en su dinámica poblacional (Nowakowski, 2003; Bialas *et al.*, 2021). Nuestros datos señalan que, independientemente de la distancia al vertedero, la calidad del hábitat circundante al nido es determinante en el éxito o fracaso reproductor, así como en el número de volantones (Capítulo 2). Estos hábitats presentan una mayor diversidad y abundancia de presas (Alonso *et al.*, 1991; Vrezec, 2009; Orłowski *et al.*, 2019), y una mayor tasa de éxito de forrajeo especialmente en zonas de pasto de ganado vacuno (Tryjanowski *et al.*, 2005; Zbyryt *et al.*, 2020) que podrían ser claves para la supervivencia de los pollos. Estos últimos son especialmente sensibles a los periodos prolongados de falta de ingesta o bajo aporte calórico durante sus primeras semanas de vida (Lázaro, 1982; Tortosa & Villafuerte, 1999; Chenchouni *et al.*, 2015).

Además del hábitat circundante, el uso de vertederos tiene un impacto directo en el éxito reproductor, como demuestran múltiples estudios previos (Kilpi & Ost, 1998; Tortosa *et al.*, 2002; Steigerwald *et al.*, 2015; Djerdali *et al.*, 2016a). Sin embargo, existe un equilibrio o "trade-off" en los efectos de los vertederos sobre la supervivencia de la prole (Capítulo 3). A corto plazo, es decir en la etapa en la que los pollos están en el nido, el uso del vertedero tiene un efecto positivo en la supervivencia de los mismos. Esto seguramente es debido a que el uso del vertedero permite asegurar el aprovisionamiento de comida para las crías, mejorando su condición corporal (Pineda-Pampliega *et al.*, 2021) y suprimiendo los periodos prolongados de inanición que tienen un efecto severo en el desarrollo y supervivencia de los pollos (Martin, 1987; Tortosa & Redondo, 1992; Djerdali *et al.*, 2008a). Además, el uso del vertedero permite a los progenitores invertir menos tiempo en la búsqueda de alimento y pasar más tiempo en el nido, lo cual es particularmente importante ante las diferentes inclemencias climáticas durante la incubación y el desarrollo temprano de los pollos (Jovani & Tella, 2004; Tobolka *et al.*, 2015; Djerdali *et al.*, 2016a). Por el contrario, la alimentación basada en los vertederos proporcionada por los progenitores durante la fase de desarrollo en el nido podría acarrear una serie de patologías debido a una dieta pobre en nutrientes (Annett & Pierotti, 1999; Murray *et al.*, 2018), ingesta de plásticos (Peris, 2003; Henry *et al.*, 2011), intoxicación con metales pesados (Kulczykowska *et al.*, 2007; de la Casa-Resino *et al.*, 2014; De la Casa-Resino *et al.*, 2015b) y diferentes contaminantes (Muñoz-Arnanz *et al.*, 2011; Pérez-López *et al.*, 2016; Plaza & Lambertucci, 2018; Tauler-Ametlller *et al.*, 2019), y con una abundante proliferación de microorganismos (Camacho *et al.*, 2016; Plaza *et al.*, 2019; Höfle *et al.*, 2020; Martín-Maldonado *et al.*, 2020); lo que desemboca en un descenso de la supervivencia a medio plazo, es decir, en la etapa juvenil. En conjunto, un uso moderado del vertedero incrementa la supervivencia de la descendencia, pero un uso muy intenso del vertedero resulta en una reducción de la probabilidad de supervivencia de la descendencia, incluso menor que aquellos que presentan un uso nulo o poco frecuente del vertedero (Capitulo 3). En consecuencia, parece que la estrategia óptima sería el uso complementario de ambas fuentes de alimentación, incrementando el uso del vertedero como fuente alternativa en periodos o años de mayor escasez de otras fuentes naturales (Evans &

Gawlik, 2020; Zorrozua *et al.*, 2020a; O'Hanlon *et al.*, 2022). Esta estrategia incluso permitiría que las parejas de individuos jóvenes compensasen su inexperiencia en la cría y búsqueda de alimento (Barbraud *et al.*, 1999; Vergara *et al.*, 2007b) con los recursos que les brindan los vertederos. Este efecto puede extenderse a los individuos heridos, enfermos o de baja calidad, en los que un uso intenso del vertedero puede incrementar sus probabilidades de criar, aunque esto suponga un coste a largo plazo para sus descendientes (Genovart *et al.*, 2010; García-Heras *et al.*, 2013).

En conjunto, queda ampliamente demostrado que los vertederos como fuente de alimentación complementaria han supuesto un beneficio para esta especie a nivel poblacional, incrementando el número de parejas reproductoras e incluso alterando los flujos migratorios y el destino de una parte sustancial de las poblaciones de la porción occidental de esta especie (Flack *et al.*, 2016; Rotics *et al.*, 2017; Marcelino *et al.*, 2022). Bajo esta premisa y ante el necesario cambio de gestión de los mismos dictaminado por la Unión Europea (Directiva 1999/31/EC y 2018/850/EC), en el capítulo 4 se han explorado las consecuencias del cierre de vertederos en dos escenarios distintos de cambio climático. Como sucede con las gaviotas (Kilpi & Ost, 1998; Delgado *et al.*, 2021), la predicción de los modelos utilizados muestra una reducción drástica del área de ocupación de esta especie y un declive generalizado de la población. La falta de acceso a esta fuente de alimentación ocasionaría una disminución en la productividad de la especie y una disminución en la supervivencia (Steigerwald *et al.*, 2015; Langley *et al.*, 2021). Al disminuir la disponibilidad y abundancia de alimento se incrementará la competencia intra-específica (Carpenter, 1987; Burke & Nol, 1998), reduciendo la viabilidad de las actuales colonias y reduciendo, por tanto, la densidad de población (Denac, 2006; Djerdali *et al.*, 2008b, 2016b). Las parejas reproductoras restantes se verán desplazadas a zonas de hábitat propicios, con abundantes porcentajes de pastos y dehesas dónde poder obtener los recursos tróficos necesarios para mantener la viabilidad de sus eventos reproductivos y su propia supervivencia (Capitulo 4).

Pero no solamente el cierre de los vertederos presenta una amenaza. El incremento del rango diario y los valores máximos de temperaturas junto con los fenómenos climatológicos extremos (p. ej. sequías y lluvias torrenciales) asociados al cambio climático en un escenario de altas emisiones y contaminación supone un incremento aproximado del 20% en la pérdida del rango de distribución actual. Además, este cambio en las condiciones climatológicas futuras probablemente dará lugar a una mayor mortalidad de los pollos por inanición e hipotermia o hipertermia, al afectar a la extensión de las áreas de alimentación (p. ejem. humedales y áreas húmedas) (Schneider *et al.*, 2011; Lefebvre *et al.*, 2019) y debido a la falta de termorregulación en las primeras dos semanas de desarrollo de los pollos (Carrascal *et al.*, 1993; Jovani & Tella, 2004; Tobolka *et al.*, 2015; Fasolă-Mătăsaru *et al.*, 2018).

Como se ha señalado anteriormente, la dificultad para encontrar lugares de nidificación apropiados, la escasa disponibilidad de alimento, los conflictos con el ser humano y el estrés producido en los pollos por las actividades humanas (Kretser *et al.*, 2008; Soulsbury & White, 2015; Blas *et al.*, 2018); dificulta la nidificación y éxito reproductor de esta especie en entornos altamente urbanizados. Por tanto, el aumento de las áreas urbanas y de la densidad de población en las próximas décadas (Cohen, 2003; Jiang & O'Neill, 2017) parece que también desplazarán a esta especie de muchas de sus áreas de cría.

Con todo esto parece que la población de esta especie volverá a una situación similar, incluso peor, a la que se encontraba previamente al uso de los vertederos, cuando se registraron los mínimos históricos en Europa (Schulz, 1999). Aunque el cierre de los vertederos a cielo abierto es una medida necesaria debido al impacto ambiental de los mismos (Butt *et al.*, 2008; Matejczyk *et al.*, 2011; Vaverková, 2019), puede resultar catastrófica para las especies que dependen de estas instalaciones si no se toman las medidas adecuadas.

Medidas de gestión

Una parte importante de este estudio es proporcionar herramientas de gestión a los técnicos y autoridades competentes para el manejo de las especies que se alimentan en vertederos.

En esta tesis se ha señalado la importancia del hábitat circundante al nido sobre distintos aspectos relacionados con la ecología de la reproducción de la especie objeto estudio y puesto que se ha señalado la degradación del hábitat original como posible origen del declive poblacional acaecido hace más de 4 décadas (Lázaro *et al.*, 1986; Dallinga & Schoenmakers, 1987; Schulz, 1999), resulta esencial evaluar la disponibilidad, calidad y abundancia de las posibles fuentes de alimentación naturales o alternativas antes de que se produzca el cierre de los vertederos. Para ello, sería necesario identificar las zonas de pastos y praderas inundables mediante el uso de herramientas información geográfica ("Geographical Information System") combinado con puntos de muestreos de presas potenciales en estas zonas similares a los de Orlowski y colaboradores (2021). En caso de detectar un acuciante descenso de la población, se podrían articular medidas para el mantenimiento y la gestión adecuada de las mejores áreas de forrajeo para la especie (Guixé & Arroyo, 2011; Croxall *et al.*, 2012; Di Marco *et al.*, 2019). Estas áreas serían los pastos, dehesas, prados húmedos y los humedales.

Sin embargo, parece lógico que la especie descienda hasta alcanzar la capacidad de carga natural del sistema, ampliamente superada debido al uso de los vertederos como fuente de alimentación complementaria o alternativa (Duhem *et al.*, 2008; Real *et al.*, 2017; Delgado *et al.*, 2021). A priori, está situación no implica un impacto negativo sobre la especie y recuperar los niveles "naturales" podría ser positivo a nivel de ecosistema (Newsome *et al.*, 2015; Payo-Payo *et al.*, 2015). Por ello, sería necesario realizar previamente un programa de seguimiento con el fin de detectar de manera temprana cambios en la estructura y composición de las poblaciones (Vos *et al.*, 2008). El seguimiento de los nidos ocupados como índice del estado de la población reproductora puede ser tremendamente útil en esta especie. Un declive significativo en el número promedio de volantones, en las parejas reproductoras o en ambas; podría activar la puesta en marcha de medidas de conservación. Más aún, detectar los diferentes patrones individuales en la intensidad de uso del vertedero, puede ayudar a identificar los sectores de la población que son potencialmente más sensibles a estos cambios.

La corrección de tendidos eléctricos (Garrido & Fernandéz-Cruz, 2003; Maricato *et al.*, 2016; Moreira *et al.*, 2017) junto con la ubicación de los aerogeneradores en zonas alejadas de los pasos migratorios, puntos de invernada o zonas de cría (Martín *et al.*, 2018; Oloo *et al.*, 2018), ayudaría también a mitigar la mortandad en esta especie. Estas dos medidas, además, resultan beneficiosas para otras muchas especies, como aves esteparias y rapaces (Tellería, 2009; Jenkins *et al.*, 2010; Bernardino *et al.*, 2018; Serrano *et al.*, 2020). La clausura del vertedero "*per se*" también podría tener un efecto positivo en la supervivencia de los juveniles, tal como como indican nuestros resultados (Capítulo 3).

No obstante, estas medidas serán sólo efectivas a largo plazo. La instalación de comederos temporales en los vertederos clausurados podría paliar el efecto drástico del cierre de los vertederos (Hilgartner *et al.*, 2014), pero es una medida controvertida por el efecto que tiene en el comportamiento de los individuos y las poblaciones. Esta medida ha resultado muy efectiva en otros grupos de especies, como los buitres, que se han beneficiado de la implantación de muladares controlados (Cortés-Avizanda *et al.*, 2016). A través de trabajo técnico realizado en el contexto de esta tesis doctoral, esta medida ya fue propuesta por nuestro equipo en 2019 y el Grupo para la Recuperación de Fauna Autóctono la está llevando a cabo de manera experimental en el vertedero clausurado de Alcalá de Henares.

Dados los diversos estudios realizados, parece más que razonable que los resultados obtenidos en los vertederos de la Comunidad de Madrid sean extrapolables a otras poblaciones y a otros grupos de especies vinculados a los vertederos, como gaviotas y buitres (Steigerwald *et al.*, 2015; Tauler-Ametller *et al.*, 2017; Arévalo-Ayala *et al.*, 2022). Sería necesarios estudios de este tipo, utilizando metodologías equivalentes para establecer patrones a nivel de toda la Península Ibérica.

LÍNEAS DE INVESTIGACIÓN FUTURAS

La coyuntura de esta tesis no ha permitido abarcar todos los aspectos de estudio deseados debido a la insuficiente financiación y al descompás de los tiempos. Si bien es cierto que en el Capítulo 4 se aborda el efecto potencial de la clausura de los vertederos en la población reproductora en el área de estudio, durante el desarrollo de esta tesis no se ha podido confirmar el efecto real que tendría esta medida en la población estudiada. Sin embargo, dos interesantes modelos de estudio han surgido recientemente. Por un lado, la gestora del vertedero de Colmenar Viejo (La Mancomunidad del Noroeste) contrató a una empresa para empezar a realizar una serie de trabajos para disuasión y control de fauna en dicho vertedero a partir de septiembre de 2021. Estas medidas consisten en el empleo de pirotecnia, cetrería y perros en diferentes momentos del día. Actualmente estamos recopilando información para determinar el efecto de estas medidas en la productividad y número de parejas reproductoras a diferentes distancias del vertedero, que se materializará a finales de este año con la presentación de un Trabajo de Fin de Máster. Nuestros resultados preliminares parecen señalar que los nidos más cercanos han sufrido una disminución de la productividad, pero no ha disminuido sustancialmente el número de parejas reproductoras.

Por otro lado, en 2021 se ha clausurado el vertedero de Alcalá de Henares y se ha abierto uno nuevo en Loeches, pero sin acceso para la avifauna (Fig. 1). Para atenuar el impacto del cierre del vertedero, la Mancomunidad del Este ha iniciado un proyecto piloto de provisión de alimento a través de una fuente artificial. El fin del aprovisionamiento de este comedero abrirá un abanico de posibles estudios.

Otro aspecto que potencialmente puede abrir vías de investigación en el futuro se relaciona con la tipología de alimento recibido por los pollos en función de la intensidad de uso que sus progenitores hacen del vertedero. En esta tesis únicamente se ha determinado que la variación en el índice de uso de los mismos afecta a su supervivencia (Capítulo 3), pero no se ha determinado la composición de la dieta de los pollos. Hubiese sido necesario vincular el uso del vertedero de cada pareja con el origen del alimento proporcionado a su descendencia. Sin embargo, el uso de egagrópilas habría producido un sesgo en los resultados debido a que infrarrepresentaría las presas que carecen de partes duras que permitan identificarlas adecuadamente (Rosin & Kwieciñski, 2011) y no se disponía de recursos para realizar dicho estudio con isótopos. Esta cuestión pretende ser solventada en un futuro reciente mediante una colaboración con investigadores de la Universidad de Georgia (Estados Unidos) para determinar las diferencias en la dieta de pollos de la misma colonia en diferentes momentos del desarrollo y cuyos padres realizan distinto uso del vertedero usando diferentes isótopos.

Fig. 1. Fotografía de la planta de gestión de residuo de Loeches que muestra la inaccesibilidad de los residuos para las aves que supone el cambio de gestión de los vertederos. Fuente: madridiario y FCC.

Más aún, las diferencias en la intensidad en el uso del vertedero a nivel individual en el capítulo 3 genera múltiples cuestiones. La idea general es determinar las características individuales subyacentes que determinan la intensidad en el uso de los vertederos. Estás características pueden ser patrones comportamentales repetibles (“Personalidad”), perfiles fisiológicos o pueden ser factores biológicos, como la edad o el sexo de los individuos, o un conjunto de varias.

En este sentido, en 2018 y 2019 realizamos una serie de test comportamentales con el fin de determinar si la respuesta de corticosterona, una hormona relacionada con la respuesta al estrés en aves, estaba relacionada con el comportamiento exploratorio en esta especie. Estudios en Carbonero común (*Parus major*) y en otras especies de páridos parecen señalar que existe una relación entre ambas (Carere *et al.*, 2010; Stöwe *et al.*, 2010; Baugh *et al.*, 2012, 2013; Hau & Goymann, 2015). Una vez establecida la relación entre ambas, respuesta de corticosterona y comportamiento exploratorio, la idea era observar si los individuos con determinado patrón de corticosterona se observaban con mayor probabilidad en el vertedero. Debido al cierre por la COVID-19 en 2020, no se pudo continuar con los test comportamentales y los resultados preliminares obtenidos en 30 individuos sólo muestran una tendencia positiva entre el comportamiento exploratorio y la respuesta de corticosterona.

Por último, como señalamos en varios capítulos y en la discusión general, el desmesurado incremento de la población parece estar cambiando la percepción de las personas sobre esta icónica especie. Aunque el objetivo de esta tesis no era indagar en el cambio de la percepción de la especie, estamos realizando una serie de encuestas con el grupo de investigación liderado por Sonia Hernández de la Universidad de Georgia (Estados Unidos) para determinar diferencias en la percepción humana de la especie en función de la densidad de cigüeñas en cada municipio. Nuestra hipótesis es que la densidad de nidos en un municipio presenta una correlación negativa con la percepción de sus habitantes sobre la especie. De confirmarse esta hipótesis, sería un motivo más para justificar el cierre de los vertederos y que la población retornase a niveles poblacionales acordes con la capacidad de carga del sistema.

CONCLUSIONS

I. This thesis confirms the attraction of landfills to white storks. Breeding population in Madrid have thrived around landfills. The changes in the distribution of breeding pairs in the last four decades was not random. We observed an increase in the number and density of pairs near landfills over the years.

II. Anthropogenic food sources alter habitat selection and preference in the surrounded area of the nest. Breeders near landfills are less selective and show more tolerance to urban areas.

III. Habitat quality in the surrounding area of the nest determines success in breeding attempts, even more than distance to landfill. However, distance to landfill seems to be more relevant in relation to number of fledglings.

IV. The use of landfills implies a trade-off on breeding output. It increases offspring survival but an intense use of this resource has a negative impact on future offspring survival. Feeding on landfills improve hatching success probably by reducing the periods of food scarcity but reduces juvenile survival possibly caused by poor food quality and contaminants. Therefore, using landfill as an alternative food source seems to be the best strategy to white storks.

V. We identify landfill closure, climate change and urbanization as the main threats to the white stork breeding populations in the future.

VI. Landfill closure may produce breeding population returns to typical habitat carrying capacity levels with a significant reduction of human conflicts but an abrupt shift in food availability can also lead to very low population levels.

VII. Periodically breeding population monitoring may allow early detection on population changes and evaluation of implementation of conservation policies.

CONCLUSIONES

I. Esta tesis confirma la atracción de las cigüeñas blancas por los vertederos. La población reproductora de Madrid ha prosperado alrededor de los vertederos. Los cambios en la distribución de las parejas reproductoras en las últimas cuatro décadas no han sido al azar. Se ha observado un incremento del número y la densidad de parejas en las proximidades de los vertederos a lo largo de los años.

II. Las fuentes de alimentación antropogénica alteran la selección y preferencia del hábitat en el área circundante al nido. Los reproductores próximos a los vertederos son menos selectivos y muestran más tolerancia por las áreas urbanas.

III. La calidad del hábitat en el área circundante al nido determina el éxito de cría, incluso más que la distancia al vertedero. Sin embargo, la distancia al vertedero es más relevante en relación al número de volantones.

IV. El uso de los vertederos implica un compromiso en el éxito reproductor. Se incrementa la supervivencia de la descendencia, pero un uso intenso de este recurso tiene un impacto negativo en la supervivencia futura de la descendencia. Probablemente, forrajear en los vertederos mejora el éxito de eclosión reduciendo los periodos de escasez de comida, pero disminuye la supervivencia juvenil debido, posiblemente, a la pobre calidad del alimento y los contaminantes presentes en ellos. Por tanto, el uso del vertedero como recurso trófico alternativo parece ser la mejor estrategia para las cigüeñas blancas.

V. Las principales amenazas identificadas para las poblaciones reproductoras de cigüeña blanca en el futuro son el cierre de los vertederos, el cambio climático y la urbanización.

VI. Tras el cierre de los vertederos, la población reproductora podría retornar a los niveles de capacidad de carga típicos de sus hábitats con una significante reducción de los conflictos con los seres humanos, pero un cambio abrupto en la disponibilidad de alimento puede llevar a tamaños poblacionales muy pequeños.

VII. El seguimiento periódico de la población reproductora puede permitir la detección temprana de cambios poblacionales y la evaluación de la implementación de políticas de conservación.

REFERENCIAS

Aebischer, N.J., Robertson, P.A. & Kenward, R.E. (1993) Compositional analysis of habitat use from animal radio-tracking data. *Ecology*, **74**, 1313–1325.

Aguirre, J.I. & Atienza, J.C. (2002) *Censo de la población reproductora de Cigüeña Blanca (Ciconia ciconia) en la Comunidad de Madrid. Año 2001*. *Anuario Ornitológico de Madrid 2001* (ed. by Seo/Birdlife), pp. 114–125. Seo/Birdlife, Madrid.

Aguirre, J.I. & Vergara, P. (2009) Census methods for White stork (*Ciconia ciconia*): Bias in sampling effort related to the frequency and date of nest visits. *Journal of Ornithology*, **150**, 147–153.

Alonso, J.C., Alonso, J.A. & Carrascal, L.M. (1991) Habitat Selection By Foraging White Storks, Ciconia-Ciconia, During the Breeding-Season. *Canadian Journal of Zoology-Revue Canadienne de Zoologie*, **69**, 1957–1962.

Ancillotto, L., Santini, L., Ranc, N., Maiorano, L. & Russo, D. (2016) Extraordinary range expansion in a common bat: The potential roles of climate change and urbanisation. *Science of Nature*, **103**, 1-8.

Annett, C.A. & Pierotti, R. (1999) Long-term reproductive output in western gulls: Consequences of alternate tactics in diet choice. *Ecology*, **80**, 288–297.

Antczak, M., Konwerski, S., Grobelny, S. & Tryjanowski, P. (2002) The Food Composition of Immature and Non-breeding White Storks in Poland. *Waterbirds*, **25**, 424–428.

Araújo, M.S., Bolnick, D.I. & Layman, C.A. (2011) The ecological causes of individual specialisation. *Ecology Letters*, **14**, 948–958.

Archaux, F., Balanca, G., Henry, P.Y. & Zapata, G. (2004) Wintering of white storks in Mediterranean France. *Waterbirds*, **27**, 441–445.

Archaux, F., Henry, P.Y. & Balança, G. (2008) High turnover and moderate fidelity of White Storks Ciconia ciconia at a European wintering site. *Ibis*, **150**, 421–424.

Arévalo-Ayala, D.J., Real, J., Durà, C., Aymerich, J. & Hernández-Matías, A. (2022) Reduction of organic waste in a landfill lowers the visitation probability but not the local abundance of a long-lived scavenger species. *Bird Conservation International*, 1–13.

Arizaga, J., Aguirre, J.I., Arroyo, B., Aymí, R., Banda, E., Barba, E., Borràs, A., Bota, G., Carrascal, L.M. & Gutiérrez-Expósito, C. (2021) *Contribución del anillamiento al conocimiento y conservación de las aves en España: pasado, presente y futuro*.

Arizaga, J., Resano-Mayor, J., Villanúa, D., Alonso, D., Barbarin, J.M., Herrero, A., Lekuona, J.M. & Rodríguez, R. (2018) Importance of artificial stopover sites through avian migration flyways a landfill-based assessment with the White Stork Ciconia ciconia. *Ibis*, 1–12.

Arnold, Z.J., Wenger, S.J. & Hall, R.J. (2021) Not just trash birds: Quantifying avian diversity at landfills using community science data. *PLoS ONE*, **16**.

Ashrafzadeh, M.R., Naghipour, A.A., Haidarian, M., Kusza, S. & Pilliod, D.S. (2019) Effects of climate change on habitat and connectivity for populations of a vulnerable, endemic salamander in Iran. *Global Ecology and Conservation*, **19**, doi.org/10.1016/j.gecco.2019.e00637.

Auman, H.J., Meathrel, C.E. & Richardson, A. (2008) Supersize Me: Does Anthropogenic Food Change the Body Condition of Silver Gulls? A Comparison Between Urbanized and Remote, Non-urbanized Areas. *Waterbirds*, **31**, 122–126.

Axelsson, E., Ratnakumar, A., Arendt, M.L., Maqbool, K., Webster, M.T., Perloski, M., Liberg, O., Arnemo, J.M., Hedhammar, Å. & Lindblad-Toh, K. (2013) The genomic signature of dog domestication reveals adaptation to a starch-rich diet. *Nature*, **495**, 360–364.

Bairlein, F. (2003) The study of bird migrations – some future perspectives. *Bird Study*, **50**, 243–253.

Balmford, A., Green, R.E. & Jenkins, M. (2003) Measuring the changing state of nature. *Trends in Ecology and Evolution*, **18**, 326–330.

Bar-On, Y.M., Phillips, R. & Milo, R. (2018) The biomass distribution on Earth. *Proceedings of the National Academy of Sciences of the United States of America*,

115, 6506–6511.

Barbet-Massin, M., Thuiller, W. & Jiguet, F. (2012) The fate of European breeding birds under climate, land-use and dispersal scenarios. *Global Change Biology*, **18**, 881–890.

Barbraud, C., Barbraud, J. & Barbraud, M. (1999) Population dynamics of the White Stork *Ciconia ciconia* in western France. *Ibis*, **141**, 469–479.

Barlein, F. (1991) Population studies of white storks (*Ciconia ciconia*) in Europe. Bird population studies: Relevance to conservation and Management (ed. by C. Perrins, J.D. Lebreton, and R. Hirons), pp. 207–229. Oxford University Press, Oxford.

Barton, K. (2022) Multi-Model Inference Version. *R Package 1.46.0*.

Bates, D., Maechler, M., Bolker, B., Walker, S., Haubo, R., Christensen, B., Singmann, H., Dai, B., Scheipl, F., Grothendieck, G. & Green, P. (2022) Linear Mixed-Effects Models using “Eigen” and S4. *R Package Version 1.1-29*.

Baugh, A.T., van Oers, K., Naguib, M. & Hau, M. (2013) Initial reactivity and magnitude of the acute stress response associated with personality in wild great tits *(Parus major)*. *General and Comparative Endocrinology*, **189**, 96–104.

Baugh, A.T., Schaper, S. V., Hau, M., Cockrem, J.F., de Goede, P. & Oers, K. van (2012) Corticosterone responses differ between lines of great tits *(Parus major)* selected for divergent personalities. *General and Comparative Endocrinology*, **175**, 488–494.

Bécares, J., Blas, J., López-López, P., Schulz, H., Torres-Medina, F., Höfle, U., Flack, A., Enggist, P., Onrubia, A., Torralvo, C., Martín, B., Bermejo, A. & De la Puente, J. (2019) *Migración y ecología espacial de la cigüeña blanca en España*, Monografía. SEO/BirdLife, Madrid.

Becker, D.J., Streicker, D.G. & Altizer, S. (2015) Linking anthropogenic resources to wildlife-pathogen dynamics: A review and meta-analysis. *Ecology Letters*, **18**, 483–495.

Beckman, A.K., Richey, B.M.S. & Rosenthal, G.G. (2022) Behavioral responses of wild animals to anthropogenic change: insights from domestication. *Behavioral Ecology* *and Sociobiology*, **76**, 105.

Belabed, B.E., Athamnia, M., Touati, L., Samraoui, F., Boucheker, A. & Samraoui, B. (2019) The early bird catches the worm: age-specific arrival time influences reproductive performance in the White Stork Ciconia ciconia. *Bird Study*, **66**, 121–129.

Belant, J.L. (1997) Gulls in urban environments: Landscape-level management to reduce conflict. *Landscape and Urban Planning*, **38**, 245–258.

Belant, J.L., Ickes, S.K. & Seamans, T.W. (1998) Importance of landfills to urban-nesting herring and ring-billed gulls. *Landscape and Urban Planning*, **43**, 11–19.

Bernardino, J., Bevanger, K., Barrientos, R., Dwyer, J.F., Marques, A.T., Martins, R.C., Shaw, J.M., Silva, J.P. & Moreira, F. (2018) Bird collisions with power lines: State of the art and priority areas for research. *Biological Conservation*, **222**, 1–13.

Bernis, F. (1981) *La población de cigüeñas españolas. Estudios y tablas de censos, período 1948 1974.*, Cátedera de Zoología de vertebrados, Universidad Complutense, Madrid.

Bialas, J.T., Dylewski, Ł., Dylik, A., Janiszewski, T., Kaługa, I., Królak, T., Kruszyk, R., Pawlukojć, K., Pestka, Z., Polakowski, M., Zbyryt, A. & Tobolka, M. (2021) Impact of land cover and landfills on the breeding effect and nest occupancy of the white stork in Poland. *Scientific Reports*, **11**, 1–14.

Bialas, J.T., Dylewski, Ł. & Tobolka, M. (2020) Determination of nest occupation and breeding effect of the white stork by human-mediated landscape in Western Poland. *Environmental Science and Pollution Research*, **27**, 4148–4158.

Bino, G., Dolev, A., Yosha, D., Guter, A., King, R., Saltz, D. & Kark, S. (2010) Abrupt spatial and numerical responses of overabundant foxes to a reduction in anthropogenic resources. *Journal of Applied Ecology*, **47**, 1262–1271.

Blair, R.B. (1996) Land use and avian species diversity along an urban gradient. *Ecological Applications*, **6**, 506–519.

Blanco, G. (1996) Population Dynamics and Communal Roosting of White Storks

Foraging at a Spanish Refuse Dump. *Colonial Waterbirds*, **19**, 273–276.

Blas, J., Torres-Medina, F., Cabezas, S., Sergio, F. & Marchant, T. a. (2018) Effects of Ecotourism on the Physiological Response To Stress in White Storks From Doñana National Park. Proyectos de investigación en parques nacionales: 2013-2017, pp. 143–160.

Bocheński, M. & Jerzak, J. (2006) Behaviour of the White Stork *Ciconia ciconia* : a review. *The White Stork in Poland: studies in biology, ecology and conservation* (ed. by P. Tryjanowski, T.H. Sparks, and L. Jerzak), pp. 295–324. Bogucki Wydawnictwo Naukowe, Poznan.

Boivin, N.L., Zeder, M.A., Fuller, D.Q., Crowther, A., Larson, G., Erlandson, J.M., Denhami, T. & Petraglia, M.D. (2016) Ecological consequences of human niche construction: Examining long-term anthropogenic shaping of global species distributions. *Proceedings of the National Academy of Sciences of the United States of America*, **113**, 6388–6396.

Bolnick, D.I., Svanbäck, R., Fordyce, J. a, Yang, L.H., Davis, J.M., Hulsey, C.D. & Forister, M.L. (2003) The ecology of individuals: incidence and implications of individual specialization. *American Naturalist*, **161**, 1–28.

Bonier, F., Martin, P.R. & Wingfield, J.C. (2007) Urban birds have broader environmental tolerance. *Biology Letters*, **3**, 670–673.

Bouten, W., Baaij, E.W., Shamoun-Baranes, J. & Camphuysen, K.C.J. (2013) A flexible GPS tracking system for studying bird behaviour at multiple scales. *Journal of Ornithology*, **154**, 571–580.

Bradley, C.A. & Altizer, S. (2007) Urbanization and the ecology of wildlife diseases. *Trends in Ecology and Evolution*, **22**, 95–102.

Bradsworth, N., White, J.G., Isaac, B. & Cooke, R. (2017) Species distribution models derived from citizen science data predict the fine scale movements of owls in an urbanizing landscape. *Biological Conservation*, **213**, 27–35.

Briga, M., Koetsier, E., Boonekamp, J.J., Jimeno, B. & Verhulst, S. (2017) Food availability affects adult survival trajectories depending on early developmental conditions. *Proceedings of the Royal Society B: Biological Sciences*, **284**, 20162287.

Bryant, D.M. (1988) Energy Expenditure and Body Mass Changes as Measures of Reproductive Costs in Birds. *Functional Ecology*, **2**, 23.

Burke, D.M. & Nol, E. (1998) Influence of Food Abundance, Nest-Site Habitat, and Forest Fragmentation on Breeding Ovenbirds. *Habitat*, **115**, 96–104.

Burnham, K.P. & Anderson, D.R. (2002) *Model Selection and Multimodel Inference: a Practical Information-Theoretic Approach*, New York.

Burns, D.D.R., Franks, D.W., Parr, C. & Robinson, E.J.H. (2021) Ant colony nest networks adapt to resource disruption. *Journal of Animal Ecology*, **90**, 143–152.

Butt, T.E., Lockley, E. & Oduyemi, K.O.K. (2008) Risk assessment of landfill disposal sites - State of the art. *Waste Management*, **28**, 952–964.

Calenge, C. (2020) Analysis of Habitat Selection by Animals. *R Package 0.3.15*.

Camacho, M.C., Hernández, J.M., Lima-Barbero, J.F. & Höfle, U. (2016) Use of wildlife rehabilitation centres in pathogen surveillance: A case study in white storks (*Ciconia ciconia*). *Preventive Veterinary Medicine*, **130**, 106–111.

Cambefort, J.P. (1981) A comparative study of culturally transmitted patterns of feeding habits in the chacma baboon *Papio ursinus* and the vervet monkey *cercopithecus aethiops*. *Folia Primatologica*, **36**, 243–263.

Candeias, D.R. & Araujo, A. (1989) *The White Stork in Portugal*. *Proceedings of the I International Stork Conservation Symposium* (ed. by G. Rheinwald, J. Ogden, and H. Schulz), pp. 236–267.

Capilla-Lasheras, P., Dominoni, D.M., Babayan, S.A., O'Shaughnessy, P.J., Mladenova, M., Woodford, L., Pollock, C.J., Barr, T., Baldini, F. & Helm, B. (2017) Elevated immune gene expression is associated with poor reproductive success of urban blue tits. *Frontiers in Ecology and Evolution*, **5**, 1–13.

Carere, C., Caramaschi, D. & Fawcett, T.W. (2010) Covariation between personalities and individual differences in coping with stress: Converging evidence and hypotheses. *Current Zoology*, **56**, 728–740.

Carlson, B.S., Rotics, S., Nathan, R., Wikelski, M. & Jetz, W. (2021) Individual environmental niches in mobile organisms. *Nature Communications*, **12**, 4572.

Carpenter, F.L. (1987) Food abundance and territoriality: To defend or not to defend? *Integrative and Comparative Biology*, **27**, 387–399.

Carrascal, L.M., Alonso, J.C. & Alonso, J. A (1990) Aggregation size and foraging behaviour of white storks *Ciconia ciconia* during the breeding season. *Ardea*, **78**, 399–404.

Carrascal, L.M., Bautista, L.M. & Encarnacin, L. (1993) Geographical variation in the density of the white stork *Ciconia ciconia* in Spain: influence of habitat structure and climate. *Biological Conservation*, **65**, 83–87.

Castillo-Huitrón, N.M., Naranjo, E.J., Santos-Fita, D. & Estrada-Lugo, E. (2020) The Importance of Human Emotions for Wildlife Conservation. *Frontiers in Psychology*, **11**, 1–11.

Catoni, C., Peters, A. & Martin Schaefer, H. (2008) Life history trade-offs are influenced by the diversity, availability and interactions of dietary antioxidants. *Animal Behaviour*, **76**, 1107–1119.

Ceballos, G., Ehrlich, P.R., Barnosky, A.D., García, A., Pringle, R.M. & Palmer, T.M. (2015) Accelerated modern human–induced species losses: Entering the sixth mass extinction. *Science*, 1–5.

Ceballos, G., Ehrlich, P.R. & Dirzo, R. (2017) Biological annihilation via the ongoing sixth mass extinction signaled by vertebrate population losses and declines. *Proceedings of the National Academy of Sciences of the United States of America*, **114**, E6089–E6096.

Chen, D.M.C., Bodirsky, B.L., Krueger, T., Mishra, A. & Popp, A. (2020) The world's growing municipal solid waste: trends and impacts. *Environmental Research Letters*, **15**, 074021.

Chenchouni, H. (2016) Variation in White Stork (*Ciconia ciconia*) diet along a climatic gradient and across rural-to-urban landscapes in North Africa. *International Journal of Biometeorology*, 1–16.

Chenchouni, H., Si Bachir, A. & AlRashidi, M. (2015) Trophic niche and feeding strategy of the White Stork (*Ciconia ciconia*) during different phases of the breeding season. *Avian Biology Research*, **8**, 1–13.

Cheng, Y., Fiedler, W., Wikelski, M. & Flack, A. (2019) “Closer-to-home” strategy benefits juvenile survival in a long-distance migratory bird. *Ecology and Evolution*, **9**, 8945–8952.

Choquet, R., Lebreton, J.D., Gimenez, O., Reboulet, A.M. & Pradel, R. (2009a) U-CARE: Utilities for performing goodness of fit tests and manipulating CApture-REcapture data. *Ecography*, **32**, 1071–1074.

Choquet, R., Rouan, L. & Pradel, R. (2009b) *Program E-SURGE: A Software Application for Fitting Multievent Models*. *Modeling Demographic Processes In Marked Populations* (ed. by D.L. Thomson, E.G. Cooch, and M.J. Conroy), pp. 845–865. Springer, Berlin, Germany.

Choquet, R., Sanz-Aguilar, A., Doligez, B., Nogué, E., Pradel, R., Gustafsson, L. & Gimenez, O. (2013) Estimating demographic parameters from capture-recapture data with dependence among individuals within clusters. *Methods in Ecology and Evolution*, **4**, 474–482.

Chozas, P. (1983) Estudio general sobre la dinámica de la población de la Cigüeña Blanca, *Ciconia ciconia* (L.), en España.

Ciach, M. & Kruszyk, R. (2010) Foraging of White Storks *Ciconia ciconia* on Rubbish Dumps on Non-Breeding Grounds. *Waterbirds*, **33**, 101–104.

Cohen, J.E. (2003) Human Population: The Next Half Century. *Science*, **302**, 1172–1175.

Corman, A.M., Mendel, B., Voigt, C.C. & Garthe, S. (2016) Varying foraging patterns in response to competition? A multicolony approach in a generalist seabird. *Ecology and Evolution*, **6**, 974–986.

Cortés-Avizanda, A., Blanco, G., Devault, T.L., Markandya, A., Virani, M.Z., Brandt, J. & Donázar, J.A. (2016) Supplementary feeding and endangered avian scavengers: Benefits, caveats, and controversies. *Frontiers in Ecology and the Environment*, **14**, 191–199.

Cowie, R.J. (1977) Optimal foraging in great tits. *Nature*, **268**, 137–139.

Croxall, J.P., Butchart, S.H.M., Lascelles, B., Stattersfield, A.J., Sullivan, B., Symes, A. & Taylor, P. (2012) Seabird conservation status, threats and priority actions: A global assessment. *Bird Conservation International*, **22**, 1–34.

Crutzen, P.J. (2002) Geology of mankind. *Nature*, **415**, 2002.

Cuadrado, M., Sánchez, Í., Barcell, M. & Armario, M. (2016) Reproductive data and analysis of recoveries in a population of white stork, *Ciconia ciconia* , in southern Spain : a 24 – year study. *Animal Biodiversity and Conservation*, **1**, 37–44.

Cuevas, J.Á. (2003) *Inventario y descripción de los hábitats incluidos en la Directiva 92/43/CEE presentes en la Comunidad de Madrid*, (ed. by Consejería de Medio Ambiente) Madrid. España.

Dallinga, J.. H.. & Schoenmakers, S. (1987) Regional decrease in the number of White storks (*Ciconia ciconia*) in relation to food resources. *Colonial Waterbirds*, **10**, 167–177.

Delgado, S., Herrero, A., Galarza, A., Aldalur, A., Zorrozua, N. & Arizaga, J. (2021) Demographic impact of landfill closure on a resident opportunistic gull. *Population Ecology*, **63**, 238–246.

Denac, D. (2006) Intraspecific exploitation competition as cause for density dependent breeding success in the white stork. *Waterbirds*, **29**, 391–394.

Dirzo, R., Young, H.S., Galetti, M., Ceballos, G., Isaac, N.J.B. & Collen, B. (2014) Defaunation in the Anthropocene. *Science*, **345**, 401–406.

Djerdali, S., Guerrero-Casado, J. & Tortosa, F.S. (2016a) Food from dumps increases the reproductive value of last laid eggs in the White Stork *Ciconia ciconia*. *Bird Study*, **3657**, 1–8.

Djerdali, S., Guerrero-Casado, J. & Tortosa, F.S. (2016b) The effects of colony size interacting with extra food supply on the breeding success of the White Stork (*Ciconia ciconia*). *Journal of Ornithology*, **157**, 941–947.

Djerdali, S., Tortosa, F.S. & Doumandji, S. (2008a) Do white stork (*Ciconia ciconia*) parents exert control over food distribution when feeding is indirect? *Ethology Ecology and Evolution*, **20**, 361–374.

Djerdali, S., Tortosa, F.S., Hillstrom, L. & Doumandji, S. (2008b) Food supply and external cues limit the clutch size and hatchability in the White Stork *Ciconia ciconia*. *Acta Ornithologica*, **43**, 145–150.

Doligez, B., Thomson, D.L. & van Noordwijk, A.J. (2004) Using large–scale data analysis to assess life history and behavioural traits: the case of the reintroduced White stork. *Animal Biodiversity and Conservation*, **27**, 387–402.

Donázar, J.A., Blanco, G., Hiraldo, F., Soto-Largo, E. & Oria, J. (2002) Effects of forestry and other land-use practices on the conservation of cinereous vultures. *Ecological Applications*, **12**, 1445–1456.

Dormann, C.F., Elith, J., Bacher, S., Buchmann, C., Carl, G., Carré, G., Marquéz, J.R.G., Gruber, B., Lafourcade, B., Leitão, P.J., Münkemüller, T., Mcclean, C., Osborne, P.E., Reineking, B., Schröder, B., Skidmore, A.K., Zurell, D. & Lautenbach, S. (2013) Collinearity: A review of methods to deal with it and a simulation study evaluating their performance. *Ecography*, **36**, 27–46.

Duhem, C., Roche, P., Vidal, E. & Tatoni, T. (2008) Effects of anthropogenic food resources on yellow-legged gull colony size on Mediterranean islands. *Population Ecology*, **50**, 91–100.

Eley, R.M., Strum, S.C., Muchemi, G. & Reid, G.D.F. (1989) Nutrition, body condition, activity patterns, and parasitism of free-ranging troops of olive baboons (*Papio anubis*) in Kenya. *American Journal of Primatology*, **18**, 209–219.

Ellenberg, U., Setiawan, A.N., Cree, A., Houston, D.M. & Seddon, P.J. (2007) Elevated hormonal stress response and reduced reproductive output in Yellow-eyed penguins exposed to unregulated tourism. *General and Comparative Endocrinology*, **152**, 54–63.

Evans, B.A. & Gawlik, D.E. (2020) Urban food subsidies reduce natural food limitations and reproductive costs for a wetland bird. *Scientific Reports*, **10**, 1–12.

Evans, K.L., Hatchwell, B.J., Parnell, M. & Gaston, K.J. (2010) A conceptual framework for the colonisation of urban areas: The blackbird *Turdus merula* as a case study. *Biological Reviews*, **85**, 643–667.

FAO (2011) *Global food losses and food waste – Extent, causes and prevention.* Rome.

Fasolă-Mătăsaru, L., Baltag, E.Ş., Ichim, P. & Cojocaru, D. (2018) Factors influencing the breeding success of white storks ciconia ciconia in Eastern Romania. *Ardeola*, **65**, 271–282.

Fiedler, W. (2009) New technologies for monitoring bird migration and behaviour. *Ringing and Migration*, **24**, 175–179.

Fitton, N., Alexander, P., Arnell, N., Bajzelj, B., Calvin, K., Doelman, J., Gerber, J.S., Havlik, P., Hasegawa, T., Herrero, M., Krisztin, T., van Meijl, H., Powell, T., Sands, R., Stehfest, E., West, P.C. & Smith, P. (2019) The vulnerabilities of agricultural land and food production to future water scarcity. *Global Environmental Change*, **58**, 101944.

Flack, A., Fiedler, W., Blas, J., Pokrovsky, I., Kaatz, M., Mitropolsky, M., Aghababyan, K., Fakriadis, I., Makrigianni, E., Jerzak, L., Azafzaf, H., Feltrup-Azafzaf, C., Rotics, S., Mokotjomela, T.M., Nathan, R. & Wikelski, M. (2016) Costs of migratory decisions: A comparison across eight white stork populations. *Science advances*, **2**, e1500931.

Florczyk, A.J., Corbane, C., Ehrlich, D., Freire, S., Kemper, T., Maffenini, L., Melchiorri, M., Politis, P., Schiavina, M., Sabo, F. & Zanchetta, L. (2019) *GHSL Data Package 2019*, Publications Office of the European Union, Luxembourg.

Fourcade, Y. & Vercauteren, M. (2022) Predicted changes in the functional structure of earthworm assemblages in France driven by climate change. *Diversity and Distributions*, **28**, 1050–1066.

Frigerio, D., Pipek, P., Kimmig, S., Winter, S., Melzheimer, J., Diblíková, L., Wachter, B. & Richter, A. (2018) Citizen science and wildlife biology: Synergies and challenges. *Ethology*, **124**, 365–377.

Galbraith, J.A., Jones, D.N., Beggs, J.R., Parry, K. & Stanley, M.C. (2017) Urban Bird Feeders Dominated by a Few Species and Individuals. *Frontiers in Ecology and Evolution*, **5**, doi: 10.3389/fevo.2017.00081.

García-Heras, M.S., Cortés-Avizanda, A. & Donázar, J.A. (2013) Who are we feeding? Asymmetric individual use of surplus food resources in an insular population of the endangered Egyptian vulture Neophron percnopterus. *PLoS ONE*, **8**, 1–7.

Garrido, J.R. & Fernandéz-Cruz, M. (2003) Effects of Power Lines on a White Stork Ciconia ciconia population in central Spain. *Ardeola*, **50**, 191–200.

Garroway, C.J. & Sheldon, B.C. (2013) Urban behavioural adaptation. *Molecular Ecology*, **22**, 3430–3432.

Genovart, M., Negre, N., Tavecchia, G., Bistuer, A., Parpal, L. & Oro, D. (2010) The young, the weak and the sick: Evidence of natural selection by predation. *PLoS ONE*, **5**, 1–5.

Gilbert, N.I., Correia, R.A., Silva, J.P., Pacheco, C., Catry, I., Atkinson, P.W., Gill, J.A. & Franco, A.M.A. (2016) Are white storks addicted to junk food? Impacts of landfill use on the movement and behaviour of resident white storks (*Ciconia ciconia*) from a partially migratory population. *Movement Ecology*, **4**, 7.

Gilchrist, J.S. & Otali, E. (2002) The effects of refuse-feeding on home-range use, group size, and intergroup encounters in the banded mongoose. *Canadian Journal of Zoology*, **80**, 1795–1802.

Golawski, A. & Kasprzykowski, Z. (2021) Alternative foraging strategies in the white stork Ciconia ciconia: The effect of mowing meadows. *Agriculture, Ecosystems and Environment*, **319**, 107563.

Greenwood, J.J.D. (2007) Citizens, science and bird conservation. *Journal of Ornithology*, **148**, 77-124.

Gregory, R.D., Van Strien, A., Vorisek, P., Meyling, A.W.G., Noble, D.G., Foppen, R.P.B. & Gibbons, D.W. (2005) Developing indicators for European birds. *Philosophical Transactions of the Royal Society B: Biological Sciences*, **360**, 269–288.

Grémillet, D., Pichegru, L., Kuntz, G., Woakes, A.G., Wilkinson, S., Crawford, R.J.M. & Ryan, P.G. (2008) A junk-food hypothesis for gannets feeding on fishery waste. *Proceedings of the Royal Society B: Biological Sciences*, **275**, 1149–1156.

Guixé, D. & Arroyo, B. (2011) Appropriateness of Special Protection Areas for wide-ranging species: The importance of scale and protecting foraging, not just nesting habitats. *Animal Conservation*, **14**, 391–399.

Harrell, F.E. (2001) With applications to linear models, logistic regression, and survival analysis. *Regression modeling strategies.*, 106–107.

Hatch, J.J. (1996) Threats to public health from gulls (Laridae). *International Journal of Environmental Health Research*, **6**, 5–16.

Hau, M. & Goymann, W. (2015) Endocrine mechanisms, behavioral phenotypes and plasticity: known relationships and open questions. *Frontiers in Zoology*, **12**, 1–15.

Henry, P.Y., Wey, G. & Balanca, G. (2011) Rubber Band Ingestion by a Rubbish Dump Dweller, the White Stork (*Ciconia ciconia*). *Waterbirds*, **34**, 504–508.

Hijmans, R.J., Phillips, S., Leathwick, J. & Elith, J. (2017) Species Distribution Modeling. *R Package 1.3-5*, 55.

Hilgartner, R., Stahl, D. & Zinner, D. (2014) Impact of supplementary feeding on reproductive success of white storks. *PLoS ONE*, **9**, e104276.

Hixon, M.A. (1980) Food Production and Competitor Density as the Determinants of Feeding Territory Size. *The American Naturalist*, **115**, 510–530.

Hmamouchi, M.J., Agharroud, K., Dahmani, J. & Hanane, S. (2020a) Landscape and coloniality are robust predictors of White Stork nest habitat selection in a coastal urban environment. *Estuarine, Coastal and Shelf Science*, **242**, 106835.

Hmamouchi, M.J., Agharroud, K., Dahmani, J. & Hanane, S. (2020b) Seeking the least urbanized landscape: white stork nest abundance variation in a Mediterranean capital city. *European Journal of Wildlife Research*, **66**, doi.org/10.1007/s10344-020-01411-z.

Höfle, U., Jose Gonzalez-Lopez, J., Camacho, M.C., Solà-Ginés, M., Moreno-Mingorance, A., Manuel Hernández, J., De La Puente, J., Pineda-Pampliega, J., Aguirre, J.I., Torres-Medina, F., Ramis, A., Majó, N., Blas, J. & Migura-Garcia, L. (2020) Foraging at Solid Urban Waste Disposal Sites as Risk Factor for Cephalosporin and Colistin Resistant *Escherichia coli* Carriage in White Storks (*Ciconia ciconia*). *Frontiers in Microbiology*, **11**, 1–13.

Höfle, U., Krone, O., Blanco, J.M.M., Pizarro, M., Hofle, U., Krone, O., Blanco, J.M.M. & Pizarro, M. (2003) *Chaunocephalus ferox* in free-living white storks in central Spain. *Avian diseases*, **47**, 506–12.

Holt, C.A., Fuller, R.J. & Dolman, P.M. (2010) Experimental evidence that deer browsing reduces habitat suitability for breeding Common Nightingales Luscinia megarhynchos. *Ibis*, **152**, 335–346.

Del Hoyo, J., Collar, N.J., Christie, D.A., Elliott, A. & Fishpool, L.D.C. (2014) *Ciconia ciconia*. *HBW and BirdLife International Illustrated Checklist of the Birds of the World*, pp. 397–397. Lynx Edicions BirdLife International, Barcelona, Spain and Cambridge, UK.

Hurlbert, S.H. (1984) Pseudoreplication and the Design of Ecological Field Experiments. *Ecological Monographs*, **54**, 187–211.

Jagiello, Z., López-García, A., Aguirre, J.I. & Dylewski, Ł. (2020) Distance to landfill and human activities affects the debris incorporation into the white stork nests in urbanized landscape in central Spain, **27**, 30893–30898. *Environmental Science and Pollution Research*.

Jagiello, Z.A., Dylewski, Ł., Winiarska, D., Zolnierowicz, K.M. & Tobolka, M. (2018) Factors determining the occurrence of anthropogenic materials in nests of the white stork *Ciconia ciconia*, **25**, 14726-14733.

James Reynolds, S., Ibáñez-Álamo, J.D., Sumasgutner, P. & Mainwaring, M.C. (2019)

Urbanisation and nest building in birds: a review of threats and opportunities. *Journal of Ornithology*, **160**, 841–860.

Jenkins, A.R., Smallie, J.J. & Diamond, M. (2010) Avian collisions with power lines: A global review of causes and mitigation with a South African perspective. *Bird Conservation International*, **20**, 263–278.

Jiang, L. & O'Neill, B.C. (2017) Global urbanization projections for the Shared Socioeconomic Pathways. *Global Environmental Change*, **42**, 193–199.

Johnson, M.T.J. & Munshi-South, J. (2017) Evolution of life in urban environments. *Science*, **358**, eaam8327.

Johst, K., Brandl, R. & Pfeifer, R. (2001) Foraging in a patchy and dynamic landscape: Human land use and the white stork. *Ecological Applications*, **11**, 60–69.

Jovani, R. & Tella, J.L. (2004) Age-related environmental sensitivity and weather mediated nestling mortality in white storks *Ciconia ciconia*. *Ecography*, **27**, 611–618.

Kanyamibwa, S., Bairlein, F. & Schierer, A. (1993) Comparison of survival rates between populations of the white stork *Ciconia ciconia* in central Europe. *Ornis Scandinavica*, **24**, 297–302.

Kark, S., Iwaniuk, A., Schalimtzek, A. & Banker, E. (2007) Living in the city: Can anyone become an "urban exploiter"? *Journal of Biogeography*, **34**, 638–651.

Kilpi, M. & Ost, M. (1998) Reduced availability of refuse and breeding output in a herring gull (*Larus argentatus*) colony. *Annales Zoologici Fennici*, **35**, 37–42.

Kosicki, J.Z., Profus, P., Dolata, P.T. & Tobółka, M. (2006) Food composition and energy demand of the White Stork Ciconia ciconia breeding population. Literature survey and preliminary results from Poland. *White Stork study in Poland: biology, ecology and conservation* (ed. by P. Tryjanowski, T.H. Sparks, and L. Jerzak), Bogucki Wydawnictwo Naukowe, Poznań, 169–183.

Koskimies, P. (1989) Birds as a tool in environmental monitoring. *Anales Zoologici Fennici*, **26**, 153–166.

Krause, E.T., Honarmand, M., Wetzel, J. & Naguib, M. (2009) Early fasting is long lasting: Differences in early nutritional conditions reappear under stressful conditions in adult female zebra finches. *PLoS ONE*, **4**, 1–6.

Kretser, H.E., Sullivan, P.J. & Knuth, B.A. (2008) Housing density as an indicator of spatial patterns of reported human-wildlife interactions in Northern New York. *Landscape and Urban Planning*, **84**, 282–292.

Kulczykowska, E., Kasprzak, M., Kalamarz, H., Kuriata, M., Nietrzeba, M., Jerzak, L. & Kamiński, P. (2007) Melatonin and thyroxine response to pollution in white stork nestlings (*Ciconia ciconia*): Aspects of rhythmicity and age. *Comparative Biochemistry and Physiology - C Toxicology and Pharmacology*, **146**, 392–397.

Kwieciński, Z., Kwiecińska, H., Botko, P., Wysocki, A., Jerzak, L. & Tryjanowski, P. (2006) Plastic strings as the cause of leg bone degeneration in the White Stork *(Ciconia ciconia)*. *White Stork study in Poland: biology, ecology and conservation* (ed. by P. Tryjanowski, T.H. Sparks, and L. Jerzak), Bogucki Wydawnictwo Naukowe, Poznań.

De la Casa-Resino, I., Hernández-Moreno, D., Castellano, A., Pérez-López, M. & Soler, F. (2014) Breeding near a landfill may influence blood metals (Cd, Pb, Hg, Fe, Zn) and metalloids (Se, As) in white stork (*Ciconia ciconia*) nestlings. *Ecotoxicology*, **23**, 1377–1386.

De la Casa-Resino, I., Hernández-Moreno, D., Castellano, A., Pérez-López, M. & Soler, F. (2015a) Chlorinated pollutants in blood of White stork nestlings (*Ciconia ciconia*) in different colonies in Spain. *Chemosphere*, **118**, 367–372.

De la Casa-Resino, I., Hernández-Moreno, D., Castellano, A., Soler Rodríguez, F. & Pérez-López, M. (2015b) Biomarkers of oxidative status associated with metal pollution in the blood of the white stork (*Ciconia ciconia*) in Spain. *Toxicological and Environmental Chemistry*, **97**, 588–598.

Lambrechts, M.M., Caro, S., Charmantier, A., Gross, N., Galan, M.J., Perret, P., Cartan-Son, M., Dias, P.C., Blondel, J. & Thomas, D.W. (2004) Habitat quality as a predictor of spatial variation in blue tit reproductive performance: A multi-plot analysis in a heterogeneous landscape. *Oecologia*, **141**, 555–561.

Langley, L.P., Bearhop, S., Burton, N.H.K., Banks, A.N., Frayling, T., Thaxter, C.B., Clewley, G.D., Scragg, E. & Votier, S.C. (2021) GPS tracking reveals landfill closures induce higher foraging effort and habitat switching in gulls. *Movement Ecology*, **9**, 1–13.

Lázaro, E. (1982) Contribución al estudio de la alimentación de la cigüeña blanca, *Ciconia C. Ciconia* (L.) en España. Universidad Complutense de Madrid. Madrid.

Lázaro, E., Chozas, P. & Fernandéz-Cruz, M. (1986) Demografía de la cigüeña blanca (*Ciconia ciconia*) en España. Censo Nacional de 1984. *Ardeola*, **33**, 131–169.

Lebreton, J.D., Burnham, K.P., Clobert, J. & Anderson, D.R. (1992) Modeling survival and testing biological hypotheses using marked animals: a unified approach with case studies. *Ecological Monographs*, **62**, 67–118.

Lefebvre, G., Redmond, L., Germain, C., Palazzi, E., Terzago, S., Willm, L. & Poulin, B. (2019) Predicting the vulnerability of seasonally-flooded wetlands to climate change across the Mediterranean Basin. *Science of the Total Environment*, **692**, 546–555.

Lewis, S.L. & Maslin, M.A. (2015) Defining the Anthropocene. *Nature*, **519**, 171–180.

Lin, D., Foster, D.P. & Ungar, L.H. (2011) VIF regression: A fast regression algorithm for large data. *Journal of the American Statistical Association*, **106**, 232–247.

Lindström, J. (1999) Early development and fitness in birds and mammals. *Trends in Ecology and Evolution*, **14**, 343–348.

López-García, A. & Aguirre, J.I. (2023) Censo de la población reproductora de cigüeña blanca (*Ciconia ciconia*) en la Comunidad de Madrid 2021. *Anuario Ornitológico de Madrid*.

López-García, A., Sanz-Aguilar, A. & Aguirre, J.I. (2021) The trade-offs of foraging at landfills: Landfill use enhances hatching success but decrease the juvenile survival of their offspring on white storks (*Ciconia ciconia*). *Science of the Total Environment*, **778**, 146217.

Loss, S.R., Loss, S.S., Will, T. & Marra, P.P. (2015) Linking place-based citizen science with large-scale conservation research: A case study of bird-building collisions and the role of professional scientists. *Biological Conservation*, **184**, 439–445.

Lowry, H., Lill, A. & Wong, B.B.M. (2013) Behavioural responses of wildlife to urban environments. *Biological Reviews*, **88**, 537–549.

Luthin, C.S. (1987) Status of and Conservation Priorities for the World's Stork Species Author. *Colonial Waterbirds*, **10**, 181–202.

Madroño, A., González, C. & Atienza, J.C. (2004) *Libro Rojo de las Aves de España*. Dirección General para la Biodiversidad-SEO/Birdlife, Madrid.

Mainwaring, M.C. (2015) The use of man-made structures as nesting sites by birds: A review of the costs and benefits. *Journal for Nature Conservation*, **25**, 17–22.

Malekian, M., Hadi, M. & Tarkesh, M. (2021) Landscape features affecting bird diversity and abundance at an urban landfill site: a case study in Central Iran. *Bird Study*, **68**, 21–29.

Manly, B.F.J., McDonald, L.L., Thomas, D.., McDonald, T.L. & Erickson, W.P. (2002) *Resource Selection by Animals Statistical Design and Analysis for Field Studies*, Second Edi. Kluwer Academic Publishers, Dordrecht.

Marcelino, J., Franco, A.M.A., Acácio, M., Soriano-Redondo, A., Moreira, F. & Catry, I. (2022) Anthropogenic food subsidies reshape the migratory behaviour of a long-distance migrant. *Science of The Total Environment*, **858**, 159992.

Marcelino, J., Moreira, F., Franco, A.M.A., Soriano-Redondo, A., Acácio, M., Gauld, J., Rego, F.C., Silva, J.P. & Catry, I. (2021) Flight altitudes of a soaring bird suggest landfill sites as power line collision hotspots. *Journal of Environmental Management*, **294**, 113149.

Marchamalo, J. (1994) La Invernada de la Cigüeña Blanca en España. *Proceedings of the International Symposium on the White Stork (Western Population)* (ed. by O. Biber, P. Enggist, C. Martí, and T. Salathe), pp. 77–80. Basel.

Marchamalo, J., Blanco, G. & Prieto, J. (1997) Presencia durante el otoño de la Cigüeña Blanca (*Ciconia ciconia*) en la Comunidad de Madrid. *Anuario Ornitologico de Madrid*, **1997**, 14–21.

Marchetti, K. & Price, T. (1989) Differences in the foraging of juvenile and adult birds: the importance of developmental constraints. *Biological Reviews*, **64**, 51–70.

Di Marco, M., Ferrier, S., Harwood, T.D., Hoskins, A.J. & Watson, J.E.M. (2019) Wilderness areas halve the extinction risk of terrestrial biodiversity. *Nature*, **573**, 582–585.

Di Marco, M., Venter, O., Possingham, H.P. & Watson, J.E.M. (2018) Changes in human footprint drive changes in species extinction risk. *Nature Communications*, **9**, 4621.

Maricato, L., Faria, R., Madeira, V., Carreira, P. & de Almeida, A.T. (2016) White stork risk mitigation in high voltage electric distribution networks. *Ecological Engineering*, **91**, 212–220.

Markovic, D., Carrizo, S., Freyhof, J., Cid, N., Lengyel, S., Scholz, M., Kasperdius, H. & Darwall, W. (2014) Europe's freshwater biodiversity under climate change: Distribution shifts and conservation needs. *Diversity and Distributions*, **20**, 1097–1107.

Martín-Maldonado, B., Vega, S., Mencía-Gutiérrez, A., Lorenzo-Rebenaque, L., de Frutos, C., González, F., Revuelta, L. & Marin, C. (2020) Urban birds: An important source of antimicrobial resistant Salmonella strains in Central Spain. *Comparative Immunology, Microbiology and Infectious Diseases*, **72**, 101519.

Martín, B., Onrubia, A. & Ferrer, M. (2016) Migration timing responses to climate change differ between adult and juvenile white storks across Western Europe. *Climate Research*, **69**, 9–23.

Martín, B., Perez-Bacalu, C., Onrubia, A., De Lucas, M. & Ferrer, M. (2018) Impact of wind farms on soaring bird populations at a migratory bottleneck. *European Journal of Wildlife Research*, **64**, 1-10.

Martin, T.E. (1987) Food as a limit on breeding birds: a life-history perspective. *Annual review of ecology and systematics*, **18**, 453–487.

Martínez-Miranzo, B., Banda, E.I. & Aguirre, J.I. (2016) Multiscale analysis of habitat selection by Bonelli's eagle (*Aquila fasciata*) in NE Spain. *European Journal of*

Wildlife Research, **62**, 673–679.

Massemin-Challet, S., Gendner, J.P., Samtmann, S., Pichegru, L., Wulgué, A. & Le Maho, Y. (2006) The effect of migration strategy and food availability on White Stork *Ciconia ciconia* breeding success. *Ibis*, **148**, 503–508.

Matejczyk, M., Płaza, G.A., Nałcz-Jawecki, G., Ulfig, K. & Markowska-Szczupak, A. (2011) Estimation of the environmental risk posed by landfills using chemical, microbiological and ecotoxicological testing of leachates. *Chemosphere*, **82**, 1017–1023.

Mendes, R.F., Ramos, J.A., Paiva, V.H., Calado, J.G., Matos, D.M. & Ceia, F.R. (2018) Foraging strategies of a generalist seabird species, the yellow-legged gull, from GPS tracking and stable isotope analyses. *Marine Biology*, **165**, 1-14.

Molina, B. & Del Moral, J.C. (2005) *La cigüeña blanca en España. VI Censo de Internacional (2004)*, (ed. by SEO/Birdlife), Madrid.

Møller, A.P. (2009) Successful city dwellers: A comparative study of the ecological characteristics of urban birds in the Western Palearctic. *Oecologia*, **159**, 849–858.

Monsarrat, S., Benhamou, S., Sarrazin, F., Bessa-Gomes, C., Bouten, W. & Duriez, O. (2013) How Predictability of Feeding Patches Affects Home Range and Foraging Habitat Selection in Avian Social Scavengers? *PLoS ONE*, **8**, 1–11.

Moreira, F., Encarnacão, V., Rosa, G., Gilbert, N., Infante, S., Costa, J., D'Amico, M., Martins, R.C. & Catry, I. (2017) Wired: Impacts of increasing power line use by a growing bird population. *Environmental Research Letters*, **12**, 024019.

Moreira, F., Martins, R.C., Catry, I. & D'Amico, M. (2018) Drivers of power line use by white storks: A case study of birds nesting on anthropogenic structures. *Journal of Applied Ecology*, **55**, 2263–2273.

Moritzi, M., Maumary, L. & Schmid, D. (2001) Time budget, habitat use and breeding success of White Storks *Ciconia ciconia* under variable foraging conditions during the breeding season in. *Ardea*, **1950**, 457–470.

Moyers, S.C., Adelman, J.S., Farine, D.R., Thomason, C.A. & Hawley, D.M. (2018) Feeder density enhances house finch disease transmission in experimental epidemics. *Philosophical Transactions of the Royal Society B: Biological Sciences*, **373**, 20170090.

Muñoz-Arnanz, J., Sáez, M., Aguirre, J.I., Hiraldo, F., Baos, R., Pacepavicius, G., Alaee, M. & Jiménez, B. (2011) Predominance of BDE-209 and other higher brominated diphenyl ethers in eggs of white stork (*Ciconia ciconia*) colonies from Spain. *Environment International*, **37**, 572–576.

Murray, M.H., Kidd, A.D., Curry, S.E., Hepinstall-Cymerman, J., Yabsley, M.J., Adams, H.C., Ellison, T., Welch, C.N. & Hernandez, S.M. (2018) From wetland specialist to hand-fed generalist: Shifts in diet and condition with provisioning for a recently urbanized wading bird. *Philosophical Transactions of the Royal Society B: Biological Sciences*, **373**, 20170100.

Nanda, S. & Berruti, F. (2021) Municipal solid waste management and landfilling technologies: a review. *Environmental Chemistry Letters*, **19**, 1433–1456.

Navarro, J., Grémillet, D., Afán, I., Miranda, F., Bouten, W., Forero, M.G. & Figuerola, J. (2019) Pathogen transmission risk by opportunistic gulls moving across human landscapes. *Scientific Reports*, **9**, 1–5.

Nevoux, M., Barbraud, J.C. & Barbraud, C. (2008) Nonlinear impact of climate on survival in a migratory white stork population. *Journal of Animal Ecology*, **77**, 1143–1152.

Newsome, T.M., Dellinger, J.A., Pavey, C.R., Ripple, W.J., Shores, C.R., Wirsing, A.J. & Dickman, C.R. (2015) The ecological effects of providing resource subsidies to predators. *Global Ecology and Biogeography*, **24**, 1–11.

Newton, I. (1992) Experiments on the limitation of bird numbers by territorial behaviour. *Biological Reviews of the Cambridge Philosophical Society*, **67**, 129–173.

Newton, I. (1989) *Lifetime reproduction in birds*, Academic Press.

Newton, I. (2007) Population Limitation in Birds. *British Birds*, **100**, 518–539.

Noguera, J.C., Metcalfe, N.B., Surai, P.F. & Monaghan, P. (2015) Are you what you eat? Micronutritional deficiencies during development influence adult personality-related traits. *Animal Behaviour*, **101**, 129–140.

Nowakowski, J.J. (2003) Habitat structure and breeding parameters of the White Stork *Ciconia ciconia* in the Kolno Upland (NE Poland). *Acta Ornithologica*, **38**, 39–46.

O'Hanlon, N.J., Thaxter, C.B., Burton, N.H.K., Grant, D., Clark, N.A., Clewley, G.D., Conway, G.J., Barber, L.J., McGill, R.A.R. & Nager, R.G. (2022) Habitat Selection and Specialisation of Herring Gulls During the Non-breeding Season. *Frontiers in Marine Science*, **9**, 816881.

Oloo, F., Safi, K. & Aryal, J. (2018) Predicting migratory corridors of white storks, ciconia ciconia, to enhance sustainablewind energy planning: A data-driven agent-based model. *Sustainability (Switzerland)*, **10**, 1470.

Olsson, O. & Bolin, A. (2014) A model for habitat selection and species distribution derived from central place foraging theory. *Oecologia*, **175**, 537–548.

Orłowski, G., Karg, J., Jerzak, L., Bocheński, M., Profus, P., Książkiewicz-Parulska, Z., Zub, K., Ekner-Grzyb, A. & Czarnecka, J. (2018) Data exploration on diet, and composition, energy value and functional division of prey items ingested by White Storks *Ciconia ciconia* in south-western Poland: Dietary variation due to land cover, reproductive output and colonial breeding. *Data in Brief*, **21**, 1186–1203.

Orłowski, G., Karg, J., Jerzak, L., Bocheński, M., Profus, P., Książkiewicz-Parulska, Z., Zub, K., Ekner-Grzyb, A. & Czarnecka, J. (2019) Linking land cover satellite data with dietary variation and reproductive output in an opportunistic forager: Arable land use can boost an ontogenetic trophic bottleneck in the White Stork *Ciconia ciconia*. *Science of the Total Environment*, **646**, 491–502.

Oro, D., Genovart, M., Tavecchia, G., Fowler, M.S. & Martínez-Abraín, A. (2013) Ecological and evolutionary implications of food subsidies from humans. *Ecology Letters*, **16**, 1501–1514.

Payo-Payo, A., Genovart, M., Bertolero, A., Pradel, R. & Oro, D. (2016) Consecutive cohort effects driven by density-dependence and climate influence early-life survival in a long-lived bird. *Proceedings of the Royal Society B: Biological Sciences*,

283, 20153042.

Payo-Payo, A., Oro, D., Igual, J.M., Jover, L.S., Sanpera, C. & Tavecchia, G. (2015) Population control of an overabundant species achieved through consecutive anthropogenic perturbations. *Ecological Applications*, **25**, 2228–2239.

Pereira, H.M., Leadley, P.W., Proença, V., Alkemade, R., Scharlemann, J.P.W., Fernandez-Manjarrés, J.F., Araújo, M.B., Balvanera, P., Biggs, R., Cheung, W.W.L., Chini, L., Cooper, H.D., Gilman, E.L., Guénette, S., Hurtt, G.C., Huntington, H.P., Mace, G.M., Oberdorff, T., Revenga, C., Rodrigues, P., Scholes, R.J., Sumaila, U.R. & Walpole, M. (2010) Scenarios for global biodiversity in the 21st century. *Science*, **330**, 1496–1501.

Pérez-López, M., De la Casa-Resino, I., Hernández-Moreno, D., Galeano, J., Míguez-Santiyán, M.P., de Castro-Lorenzo, A., Otero-Filgueiras, M., Rivas-López, O. & Soler, F. (2016) Concentrations of Metals, Metalloids, and Chlorinated Pollutants in Blood and Plasma of White Stork (*Ciconia ciconia*) Nestlings From Spain. *Archives of Environmental Contamination and Toxicology*, **71**, 313–321.

Peris, S.J. (2003) Feeding in urban refuse dumps:ingestion of plastic objects by the White Stork (*Ciconia ciconia*). *Ardeola*, **50**, 81–84.

Pfeiffer, M.B., Blackwell, B.F. & DeVault, T.L. (2020) Collective effect of landfills and landscape composition on bird-aircraft collisions. *Human-Wildlife Interactions*, **14**, 43–54.

Pineda-Pampliega, J., Herrera-Dueñas, A., Mulder, E., Aguirre, J.I., Höfle, U. & Verhulst, S. (2020) Antioxidant supplementation slows telomere shortening in free-living white stork chicks. *Proceedings of the Royal Society B: Biological Sciences*, **287**, 4–11.

Pineda-Pampliega, J., Ramiro, Y., Herrera-Dueñas, A., Martinez-Haro, M., Hernández, J.M., Aguirre, J.I. & Höfle, U. (2021) A multidisciplinary approach to the evaluation of the effects of foraging on landfills on white stork nestlings. *Science of the Total Environment*, **775**, 145197.

Pinto, E., Bastos, R., Luís, A. & Cabral, J.A. (2021) Localized control of opportunistic, overabundant species in protected areas: a retrospective modelling approach encompassing future scenarios. *Animal Conservation*, **24**, 798–809.

Plaza, P.I., Blanco, G., Madariaga, M.J., Boeri, E., Teijeiro, M.L., Bianco, G. & Lambertucci, S.A. (2019) Scavenger birds exploiting rubbish dumps: Pathogens at the gates. *Transboundary and Emerging Diseases*, **66**, 873–881.

Plaza, P.I. & Lambertucci, S.A. (2017) How are garbage dumps impacting vertebrate demography, heath, and conservation? *Global Ecology and Conservation*, **12**, 9–20.

Plaza, P.I. & Lambertucci, S.A. (2018) More massive but potentially less healthy: Black vultures feeding in rubbish dumps differed in clinical and biochemical parameters with wild feeding birds. *PeerJ*, **2018**, e4645.

Polidori, C., García-Gila, J., Blasco-Aróstegui, J. & Gil-Tapetado, D. (2021) Urban areas are favouring the spread of an alien mud-dauber wasp into climatically non-optimal latitudes. *Acta Oecologica*, **110**, 103678.

Powers, R.P. & Jetz, W. (2019) Global habitat loss and extinction risk of terrestrial vertebrates under future land-use-change scenarios. *Nature Climate Change*, **9**, 323–329.

Pradel, R., Gimenez, O. & Lebreton, J.-D. (2005) Principles and interest of GOF tests for multistate capture–recapture models. *Animal Biodiversity and Conservation*, **2**, 189–204.

QGIS Development Team (2022) QGIS Geographic Information System. QGIS Association. *QGIS 3.16*.

R Core Team (2020) R: A Language and Environment for Statistical Computing. Vienna: R Foundation for Statistical Computing.

Radović, A., Kati, V., Perčec Tadić, M., Denac, D. & Kotrošan, D. (2015) Modelling the spatial distribution of white stork Ciconia ciconia breeding populations in Southeast Europe. *Bird Study*, **62**, 106–114.

Real, E., Oro, D., Martínez-Abraín, A., Igual, J.M., Bertolero, A., Bosch, M. & Tavecchia,

G. (2017) Predictable anthropogenic food subsidies, density-dependence and socio-economic factors influence breeding investment in a generalist seabird. *Journal of Avian Biology*, **48**, 1462–1470.

Redondo, T., Tortosa, F.S. & Arias de Reyna, L. (1995) Nest switching and alloparental care in colonial white storks. *Animal Behaviour*, **49**, 1097–1110.

Restani, M., Marzluff, J.M. & Yates, R.E. (2001) Effects of anthropogenic food sources on movements, survivorship, and sociality of Common Ravens in the Arctic. *Condor*, **103**, 399–404.

Riahi, K., van Vuuren, D.P., Kriegler, E., Edmonds, J., O'Neill, B.C., Fujimori, S., Bauer, N., Calvin, K., Dellink, R., Fricko, O., Lutz, W., Popp, A., Cuaresma, J.C., KC, S., Leimbach, M., Jiang, L., Kram, T., Rao, S., Emmerling, J., Ebi, K., Hasegawa, T., Havlik, P., Humpenöder, F., Da Silva, L.A., Smith, S., Stehfest, E., Bosetti, V., Eom, J., Gernaat, D., Masui, T., Rogelj, J., Strefler, J., Drouet, L., Krey, V., Luderer, G., Harmsen, M., Takahashi, K., Baumstark, L., Doelman, J.C., Kainuma, M., Klimont, Z., Marangoni, G., Lotze-Campen, H., Obersteiner, M., Tabeau, A. & Tavoni, M. (2017) The Shared Socioeconomic Pathways and their energy, land use, and greenhouse gas emissions implications: An overview. *Global Environmental Change*, **42**, 153–168.

Richardson, K.M., Parlato, E.H., Walker, L.K., Parker, K.A., Ewen, J.G. & Armstrong, D.P. (2019) Links between personality, early natal nutrition and survival of a threatened bird. *Philosophical Transactions of the Royal Society B: Biological Sciences*, **374**, 20190373.

Ricklefs, R. (1973) *Breeding Biology of Birds*, National Academy of Sciences, Washington, DC.

Ricklefs, R.E. (1979) Adaptation, constraint, and compromise in avian postnatal development. *Biological reviews of the Cambridge Philosophical Society*, **54**, 269–290.

Rodríguez, A., Negro, J.J., Mulero, M., Rodríguez, C., Hernández-Pliego, J. & Bustamante, J. (2012) The Eye in the Sky: Combined Use of Unmanned Aerial Systems and GPS Data Loggers for Ecological Research and Conservation of Small Birds. *PLoS ONE*, **7**, e50336.

Rolland, C., Danchin, E. & De Fraipont, M. (1998) The evolution of coloniality in birds in relation to food, habitat, predation, and life-history traits: A comparative analysis. *American Naturalist*, **151**, 514–529.

Rosin, Z.M. & Kwiecinski, Z. (2011) Digestibility of prey by the White Stork (*Ciconia ciconia*) under experimental conditions. *Ornis Fennica*, **80**, 40–50.

Rotics, S., Kaatz, M., Resheff, Y.S., Turjeman, S.F., Zurell, D., Sapir, N., Eggers, U., Flack, A., Fiedler, W., Jeltsch, F., Wikelski, M., Nathan, R. & Both, C. (2016) The challenges of the first migration: movement and behaviour of juvenile vs. adult white storks with insights regarding juvenile mortality. *Journal of Animal Ecology*, **85**, 938–947.

Rotics, S., Turjeman, S., Kaatz, M., Resheff, Y.S., Zurell, D., Sapir, N., Eggers, U., Fiedler, W., Flack, A., Jeltsch, F., Wikelski, M. & Nathan, R. (2017) Wintering in Europe instead of Africa enhances juvenile survival in a long-distance migrant. *Animal Behaviour*, **126**, 79–88.

Rumbold, D.G., Morrison, M. & Bruner, M.C. (2009) Assessing the ecological risk of a municipal solid waste landfill to surrounding wildlife: A case study in florida. *Environmental Bioindicators*, **4**, 246–279.

Sáez, M., Aguirre, J.I., Blázquez, E. & Jiménez, B. (2008) Organochlorines in White Stork (*Ciconia ciconia*): A comparison of levels in eggs and nestlings. *Organohalogen Compounds*, **70**, 1133–1136.

Saez, M., Aguirre, J.I. & Jiménez, B. (2009) Organochlorines in plasma of nestlings and adults white stork (*Ciconia ciconia*) from Madrid, Spain. *Organohalogen Compounds*, **71**, 2915–2918.

Sanz-Aguilar, A., Jovani, R., Melián, C.J., Pradel, R. & Tella, J.L. (2015) Multievent capture-recapture analysis reveals individual foraging specialisation in a generalist species. *Ecology*, **96**, 1650–1660.

Schaub, M., Kania, W. & Köppen, U. (2005) Variation of primary production during winter induces synchrony in survival rates in migratory white storks *Ciconia ciconia*.

Journal of Animal Ecology, **74**, 656–666.

Schneider, C., Flörke, M., Geerling, G., Duel, H., Grygoruk, M. & Okruszko, T. (2011) The future of European floodplain wetlands under a changing climate. *Journal of Water and Climate Change*, **2**, 106–122.

Schoener, T.W. (1971) Theory of Feeding Strategies. *Annual Review of Ecology and Systematics*, **2**, 369–404.

Schulz, H. (1999) *The world population of the White Stork (Ciconia ciconia)*. *White Storks on the up? Proc. International Symposium on the White Stork*, pp. 351–365. Hamburg.

Seress, G. & Liker, A. (2015) Habitat urbanization and its effects on birds. *Acta Zoologica Academiae Scientiarum Hungaricae*, **61**, 373–408.

Seress, G., Sándor, K., Evans, K.L. & Liker, A. (2020) Food availability limits avian reproduction in the city: An experimental study on great tits *Parus major*. *Journal of Animal Ecology*, **89**, 1570–1580.

Sergio, F. & Newton, I. (2003) Occupancy as a measure of territory quality. *Journal of Animal Ecology*, **72**, 857–865.

Serrano, D., Margalida, A., Juan, M., Juste, J., Traba, J., Carrete, M. & Aihartza, J. (2020) Renewables in Spain threaten biodiversity Small Aral Sea brings hope for Lake Balkhash Brazil’s areas of not-so- permanent preservation. *Science*, **370**, 1282–1283.

Sharps, K., Henderson, I., Conway, G., Armour-Chelu, N. & Dolman, P.M. (2015) Home-range size and habitat use of European Nightjars *Caprimulgus europaeus* nesting in a complex plantation-forest landscape. *Ibis*, **157**, 260–272.

Shephard, J.M., Rycken, S., Almalik, O., Struyf, K. & Van Erp-van der Kooij, L. (2015) Migration strategies revealed by satellite tracking among descendants of a population of European white stork (*Ciconia ciconia*) reintroduced to Belgium. *Journal of Ornithology*, **156**, 943–953.

Siriwardena, G.M., Baillie, S.R., Buckland, S.T., Fewster, R.M., Marchant, J.H. & Wilson, J.D. (1998) Trends in the abundance of farmland birds: A quantitative comparison of smoothed Common Birds Census indices. *Journal of Applied Ecology*, **35**, 24–43.

Smits, J.E., Bortolotti, G.R., Baos, R., Jovani, R., Tella, J.L. & Hoffmann, W.E. (2007) Disrupted bone metabolism in contaminant-exposed white storks (*Ciconia ciconia*) in southwestern Spain. *Environmental Pollution*, **145**, 538–544.

Smits, J.E.G., Bortolotti, G.R., Baos, R., Blas, J., Hiraldo, F. & Xie, Q. (2005) Skeletal Pathology in White Storks (*Ciconia ciconia*) Associated With Heavy Metal Contamination in Southwestern Spain. *Toxicologic Pathology*, **33**, 441–448.

Soriano-Redondo, A., Franco, A.M.A., Acácio, M., Martins, B.H., Moreira, F. & Catry, I. (2021) Flying the extra mile pays-off: Foraging on anthropogenic waste as a time and energy-saving strategy in a generalist bird. *Science of the Total Environment*, **782**, 146843.

Soulsbury, C.D. & White, P.C.L. (2015) Human-wildlife interactions in urban areas: a review of conflicts. *Csiro*, **42**, 541–533.

Spelt, A., Soutar, O., Williamson, C., Memmott, J., Shamoun-Baranes, J., Rock, P. & Windsor, S. (2021) Urban gulls adapt foraging schedule to human-activity patterns. *Ibis*, **163**, 274–282.

Spelt, A., Williamson, C., Rock, P., Windsor, S., Shamoun-baranes, J. & Shepard, E. (2019) Habitat use of urban-nesting lesser black-backed gulls during the breeding season. *Scientific Reports*, **9**, 10527.

Steigerwald, E.C., Igual, J.M., Payo-Payo, A. & Tavecchia, G. (2015) Effects of decreased anthropogenic food availability on an opportunistic gull: Evidence for a size-mediated response in breeding females. *Ibis*, **157**, 439–448.

Stenmarck, Å., Jensen, C., Quested, T., Moates, G., Cseh, B., Juul, S., Parry, A., Politano, A., Redlingshofer, B., Scherhaufer, S., Silvennoinen, K., Soethoudt, H., Zübert, C. & Östergren, K. (2016) *Estimates of European food waste levels*, Stockholm.

Stephens, D.W. & Krebs, J.R. (1986) *Foraging Theory*, Princeton University Press, Princeton, New Jersey.

Stine, R.A. (1995) Graphical interpretation of variance inflation factors. *American Statistician*, **49**, 53–56.

Stöwe, M., Rosivall, B., Drent, P.J. & Möstl, E. (2010) Selection for fast and slow exploration affects baseline and stress-induced corticosterone excretion in Great tit nestlings, *Parus major*. *Hormones and Behavior*, **58**, 864–871.

Strasser, E.H. & Heath, J.A. (2013) Reproductive failure of a human-tolerant species, the American kestrel, is associated with stress and human disturbance. *Journal of Applied Ecology*, **50**, 912–919.

Sullivan, B.L., Wood, C.L., Iliff, M.J., Bonney, R.E., Fink, D. & Kelling, S. (2009) eBird: A citizen-based bird observation network in the biological sciences. *Biological Conservation*, **142**, 2282–2292.

Sumner, M., Cook, I. & Baston, D. (2022) Simple Features for R. *R Package 1.0-7*.

Szabóa, S., Bódisa, K., Kougiasa, I., Moner-Gironaa, M., Jäger-Waldaua, A., Bartonb, G. & Szabóc, L. (2017) A methodology for maximizing the benefits of solar landfills on closed sites. *Renewable and Sustainable Energy Reviews journal homepage: www.elsevier.com/locate/rser A*, **76**, 1291–1300.

Tauler-Ametller, H., Hernández-Matías, A., Pretus, J.L.L. & Real, J. (2017) Landfills determine the distribution of an expanding breeding population of the endangered Egyptian Vulture *Neophron percnopterus*. *Ibis*, **159**, 757–768.

Tauler-Ametlller, H., Pretus, J.L., Hernández-Matías, A., Ortiz-Santaliestra, M.E., Mateo, R. & Real, J. (2019) Domestic waste disposal sites secure food availability but diminish plasma antioxidants in Egyptian vulture. *Science of the Total Environment*, **650**, 1382–1391.

Tella, J.L., Hiraldo, F., Donázar-Sancho, J.A. & Negro, J.J. (1996) Costs and Benefits of Urban Nesting in the Lesser Kestrel. *Raptors in Human Landscapes*, 53–60.

Tellería, J.L. (2009) Wind power plants and the conservation of birds and bats in Spain: A geographical assessment. *Biodiversity and Conservation*, **18**, 1781–1791.

Thomsen, K.-M., Kaatz, C., Kaatz, M. & Ludwig, B. (2017) *Verbreitung und Bestand. Der*

Weißstorch – Ciconia ciconia (ed. by C. Kaatz, D. Wallschläger, K. Dziewiaty, and U. Eggers), pp. 148–205. Magdeburg: VerlagsKG Wolf.

Thorup, K., Korner-Nievergelt, F., Cohen, E.B. & Baillie, S.R. (2014) Large-scale spatial analysis of ringing and re-encounter data to infer movement patterns: A review including methodological perspectives. *Methods in Ecology and Evolution*, **5**, 1337–1350.

Thuiller, W. (2004) Patterns and uncertainties of species' range shifts under climate change. *Global Change Biology*, **10**, 2020–2027.

Tinbergen, J.M. & Dietz, M.W. (1994) Parental Energy Expenditure During Brood Rearing in the Great Tit (*Parus major*) in Relation to Body Mass, Temperature, Food Availability and Clutch Size. *Functional Ecology*, **8**, 563.

Tobolka, M. (2014) Importance of juvenile mortality in birds' population: Early post-fledging mortality and causes of death in white stork *Ciconia ciconia*. *Polish Journal of Ecology*, **62**, 807–813.

Tobolka, M., Kuźniak, S., Zolnierowicz, K.M., Sparks, T.H. & Tryjanowski, P. (2013) New is not always better: Low breeding success and different occupancy patterns in newly built nests of a long-lived species, the white stork *Ciconia ciconia*. *Bird Study*, **60**, 399–403.

Tobolka, M., Zolnierowicz, K.M. & Reeve, N.F. (2015) The effect of extreme weather events on breeding parameters of the White Stork *Ciconia ciconia*. *Bird Study*, **62**, 377–385.

Tortosa, F.S., Caballero, J.M. & Reyes-López, J. (2002) Effect of rubish dumps on breeding success in the White stork in Southern Spain. *Waterbirds*, **25**, 39–43.

Tortosa, F.S., Pérez, L. & Hillström, L. (2003) Effect of food abundance on laying date and clutch size in the White Stork *Ciconia ciconia*. *Bird Study*, **50**, 112–115.

Tortosa, F.S. & Redondo, T. (1992) Motives for Parental Infanticide in White Storks *Ciconia ciconia*. *Ornis Scandinavica*, **23**, 185–189.

Tortosa, F.S. & Villafuerte, R. (1999) Effect of nest microclimate on effective endothermy in white stork *Ciconia ciconia* nestlings. *Bird Study*, **46**, 336–341.

Tourani, M. (2022) A review of spatial capture–recapture: Ecological insights, limitations, and prospects. *Ecology and Evolution*, **12**, 1–13.

Tryjanowski, P., Grzywaczewski, G. & Zbyryt, A. (2018) Foraging of white stork ciconia ciconia in forests-The heritage of an ancient behaviour? *Polish Journal of Ecology*, **66**, 250–256.

Tryjanowski, P., Jerzak, L. & Radkiewicz, J. (2005) Effect of water level and livestock on the productivity and numbers of breeding White Storks. *Waterbirds*, **28**, 378–382.

Tulloch, A.I.T., Possingham, H.P., Joseph, L.N., Szabo, J. & Martin, T.G. (2013) Realising the full potential of citizen science monitoring programs. *Biological Conservation*, **165**, 128–138.

Tuomainen, U. & Candolin, U. (2011) Behavioural responses to human-induced environmental change. *Biological Reviews*, **86**, 640–657.

Vaverková, M.D. (2019) Landfill impacts on the environment— review. *Geosciences (Switzerland)*, **9**, 1–16.

Venter, O., Sanderson, E.W., Magrach, A., Allan, J.R., Beher, J., Jones, K.R., Possingham, H.P., Laurance, W.F., Wood, P., Fekete, B.M., Levy, M.A. & Watson, J.E.M. (2016) Sixteen years of change in the global terrestrial human footprint and implications for biodiversity conservation. *Nature Communications*, **7**, 1–11.

Vergara, P. & Aguirre, J.I. (2006) Age and breeding success related to nest position in a White stork *Ciconia ciconia* colony. *Acta Oecologica*, **30**, 414–418.

Vergara, P., Aguirre, J.I., Fargallo, J. a. & Davila, J. a. (2006) Nest-site fidelity and breeding success in White Stork *Ciconia ciconia*. *Ibis*, **148**, 672–677.

Vergara, P., Aguirre, J.I. & Fargallo, J.A. (2007a) Economical versus ecological development: A case study of white storks in a cattle farm. *Ardeola*, **54**, 217–225.

Vergara, P., Aguirre, J.I. & Fernández-Cruz, M. (2004) Fidelidad a los sitios y fenologia en la invernada de la Cigüeña Blanca (*Ciconia ciconia*) en la Comunidad de Madrid

(1998-2002). *Anuario Ornitologico de Madrid 2003* (ed. by J. de la Puente, J. Pérez-Tris, A. Bermejo, and M. Juan), pp. 74–85. SEO-Monticola, Madrid.

Vergara, P., Gordo, O. & Aguirre, J.I. (2010) Nest size, nest building behaviour and breeding success in a species with nest reuse: the white stork *Ciconia ciconia*. *Annales Zoologici Fennici*, **47**, 184–194.

Vergara, P., I. Aguirre, J. & Fernández-Cruz, M. (2007b) Arrival date, age and breeding success in white stork *Ciconia ciconia*. *Journal of Avian Biology*, **38**, 573–579.

Vitousek, P.M., Mooney, H.A., Lubchenco, J. & Melillo, J.M. (1997) Human domination of Earth's ecosystems. *Science*, **277**, 494–499.

Vos, C.C., Berry, P., Opdam, P., Baveco, H., Nijhof, B., O'Hanley, J., Bell, C. & Kuipers, H. (2008) Adapting landscapes to climate change: Examples of climate-proof ecosystem networks and priority adaptation zones. *Journal of Applied Ecology*, **45**, 1722–1731.

Vrezec, A. (2009) Insects in the White Stork *Ciconia ciconia* diet as indicators of its feeding conditions: the first diet study in Slovenia. *Acrocephalus*, **30**, 25–29.

Watson, J.E.M., Shanahan, D.F., Di Marco, M., Allan, J., Laurance, W.F., Sanderson, E.W., Mackey, B. & Venter, O. (2016) Catastrophic Declines in Wilderness Areas Undermine Global Environment Targets. *Current Biology*, **26**, 2929–2934.

Wessely, J., Hülber, K., Gattringer, A., Kuttner, M., Moser, D., Rabitsch, W., Schindler, S., Dullinger, S. & Essl, F. (2017) Habitat-based conservation strategies cannot compensate for climate-change-induced range loss. *Nature Climate Change*, **7**, 823–827.

Wikelski, M., Quetting, M., Cheng, Y., Fiedler, W., Flack, A., Gagliardo, A., Salas, R., Zannoni, N. & Williams, J. (2021) Smell of green leaf volatiles attracts white storks to freshly cut meadows. *Scientific Reports*, **11**, 1–9.

Zbyryt, A., Dylewski, Ł. & Neubauer, G. (2021) Mass of white stork nests predicted from their size: Online calculator and implications for conservation. *Journal for Nature Conservation*, **60**, 125967.

Zbyryt, A., Sparks, T.H. & Tryjanowski, P. (2020) Foraging efficiency of white stork *Ciconia ciconia* significantly increases in pastures containing cows. *Acta Oecologica*, **104**, 103544.

Zorrozua, N., Aldalur, A., Herrero, A., Diaz, B., Delgado, S., Sanpera, C., Jover, L. & Arizaga, J. (2020a) Breeding Yellow-legged Gulls increase consumption of terrestrial prey after landfill closure. *Ibis*, **162**, 50–62.

Zorrozua, N., Egunez, A., Aldalur, A., Galarza, A., Diaz, B., Hidalgo, J., Jover, L., Sanpera, C., Castège, I. & Arizaga, J. (2020b) Evaluating the effect of distance to different food subsidies on the trophic ecology of an opportunistic seabird species. *Journal of Zoology*, **311**, 45–55.

Zurell, D., Eggers, U., Kaatz, M., Rotics, S., Sapir, N., Wikelski, M., Nathan, R. & Jeltsch, F. (2015) Individual-based modelling of resource competition to predict density-dependent population dynamics: A case study with white storks. *Oikos*, **124**, 319–330.

Zurell, D., Wehrden, H. Von, Rotics, S., Kaatz, M., Groß, H., Schlag, L., Schäfer, M., Sapir, N., Turjeman, S. & Wikelski, M. (2018) Home Range Size and Resource Use of Breeding and Non-breeding White Storks Along a Land Use Gradient. *Frontiers in Ecology and Evolution*, **6**, 1–11.

ANEXO I: CENSO REPRODUCTOR DE CIGÜEÑA BLANCA 2021 EN MADRID

Basado en el manuscrito: López-García, A. y Aguirre, J. I. 2023. Censo de la población reproductora de cigüeña blanca (*Ciconia ciconia*) en la Comunidad de Madrid 2021. En: Juan, M.; Martín, M. y De la Torre, V. (ed.) Anuario Ornitológico de Madrid 2021: 133-143. SEO-Monticola. Madrid. *Aceptado*.

Introducción

La cigüeña blanca (*Ciconia ciconia*) es una de las especies más ampliamente estudiadas por su asociación con ambientes antrópicos y su fácil identificación y popularidad, que la convierten en un candidato ideal para estudios que cuenten con participación de ciencia ciudadana. Así lo atestiguan los seis censos nacionales, los múltiples censos regionales y las diversas publicaciones llevadas a cabo en España (Bernis, 1981; Lázaro *et al.* 1986; SEO/Birdlife, 1995; Aguirre y Atienza 2002; Prieto 2002; Molina y Del Moral 2005; Barbarín *et al.* 2021).

Las poblaciones europeas de cigüeña blanca sufren un marcado declive desde el comienzo de la segunda mitad del siglo XX (Bernis, 1981; Barlein, 1991) y que, en España, se materializa en mínimos históricos en el IV censo internacional de 1984 (Lázaro *et al.* 1986). Como resultado de estos estudios se toman medidas de protección para esta especie que, unidas al creciente uso de vertederos como fuentes de alimentación, llevan al crecimiento de la especie en la década siguiente y a su recuperación a principios del siglo XXI (Schulz, 1999; Tortosa *et al.* 2002; Massemin-Challet *et al.* 2006). La Comunidad de Madrid no es una excepción a esta tendencia generalizada, pasa de un mínimo de 215 parejas en 1984, a 582 en 1994 y 1.221 en 2004 (Lázaro *et al.* 1986; SEO/Birdlife, 1995; Molina y Del Moral 2005).

La cigüeña blanca está incluida en el Listado de Especies Silvestres en Régimen de Protección Especial (Real Decreto 139/2011) y catalogada como “Vulnerable” en el Catálogo Regional de Especies Amenazadas de la Comunidad de Madrid (Decreto 18/1992). Sin embargo, se desconoce la situación actual de esta especie en España puesto que no se realiza el VII Censo Internacional en 2014, a diferencia de otros países europeos, como Alemania, Francia, Países Bajos, Polonia y Suiza entre otros (Thomsen *et al.* 2017; Wuczyński *et al.* 2021).

Por tanto, se hace necesaria una actualización de la información de la población reproductora de esta especie en la Comunidad de Madrid, especialmente tras el comienzo de la aplicación de la normativa en el cambio de entrada y gestión del tratamiento de los residuos en los vertederos en la región, de acuerdo a la legislación europea (European Directive 1999/31/EC, European Directive 2018/850/EC), como es el reciente caso de Alcalá de Henares, y el número de permisos para la retirada de nidos e instalación de medidas disuasorias, concedidos desde el último censo regional de 2004.

En este estudio se presentan los resultados obtenidos en el censo de la población reproductora de cigüeña blanca en la Comunidad de Madrid en 2021, con el objetivo principal de evaluar su evolución desde 2004.

Metodología

Para este censo se siguió la metodología de observación directa de los nidos definida en los tres últimos censos de 1994, 2001 y 2004 (Schulz 1999; Aguirre y Atienza 2002; Molina y Del Moral, 2005; Aguirre y Vergara, 2009), y se prospectó la totalidad del territorio de la Comunidad de Madrid.

Siempre que fue posible se realizaron al menos tres visitas para determinar el número de volantones: una primera visita durante la segunda quincena de marzo o primera quincena de abril, una segunda visita durante el mes de mayo, y una tercera visita en junio, idealmente en la primera quincena. De no ser posible realizar las tres visitas, se optó por realizar las dos primeras.

En aquellas zonas o fincas donde sólo se pudo realizar una única visita por restricciones de acceso o cuestiones logísticas, esta se realizó durante la segunda quincena de abril o el mes de mayo con el objetivo de poder confirmar que se trataba de individuos reproductores. Esta metodología aporta más de un 90% de precisión a la hora de determinar la ocupación de los nidos (Aguirre y Vergara 2009).

En cada visita se anotaron las coordenadas del nido (punto GPS), el término municipal, el substrato de nidificación (construcción humana, árbol, tendido eléctrico, poste *ad hoc*, silo u otro), el estado del nido, la presencia o ausencia de adultos, el comportamiento de los adultos (de pie, echados, con pollos) y el número de pollos.

Se determinaron como coloniales a todos aquellos nidos separados por una distancia menor de 500 metros entre ellos.

Debido al gran número de participantes en este tipo de estudios y el uso extendido de los teléfonos móviles, se optó por diseñar un cuestionario adaptado a este censo en la aplicación Zamiadroid (http://biodiver.bio.ub.es/zamiaDroid) para recoger los datos.

Además, con el fin de comparar los datos obtenidos en censos anteriores, se utilizaron esos mismos parámetros reproductivos (Schulz 1999) definidos como:

- Número de parejas que ocupan nido (HPa): nidos en los que se observa al menos un adulto en cualquiera de las dos visitas.
- Número de parejas con pollos volantones (HPm): incluimos todos los nidos que tuviesen pollos a partir del 10 de mayo, con al menos 30 días de edad.
- Número de parejas que ocupan un nido, pero sin volantones (HPo).
- Número de parejas con éxito reproductivo desconocido (HPx): nidos ocupados de los que no se pudo realizar un seguimiento o no se pudo visitar entre mediados de mayo y principios de junio.
- Número total de volantones en la Comunidad de Madrid (JZG).
- Productividad total (JZa): número de pollos totales (JZG) dividido entre el número de nidos con reproducción constatada.

$$\mathrm{JZa} = \mathrm{JZG} / (\mathrm{HPa} - \mathrm{HPx})$$

- Productividad parcial (JZm): número de pollos totales en la región dividido entre el número de parejas con pollos volantones.

$$\mathrm{JZm} = \mathrm{JZG} / \mathrm{HPm}$$

- Densidad de nidos en la región (StD): número de nidos ocupados por cada 100 km^2.

Para las estimas de los parámetros reproductivos se consideró el número máximo de nidos con algún signo de ocupación en al menos una visita.

Todas las observaciones se realizaron siempre a una distancia prudencial para evitar interferir en la conducta normal de la especie, y se contó con los permisos pertinentes y de acuerdo a la normativa vigente.

Resultados y discusión

Tamaño poblacional

Este censo arrojó una cifra de 2.327 nidos ocupados. Este valor podría ser algo menor, de 2.176 nidos, si descartamos aquellos de ocupación dudosa puesto que sólo se pudo realizar una única visita sin signos de reproducción (p. ej. adulto echado o pollos), y los que se retiraron entre visitas una vez constatada la ocupación.

Sin embargo, el número potencial de nidos ocupados podría ser ligeramente mayor debido a que la Consejería de Medio Ambiente, Vivienda y Agricultura confirmó que serían más de 150 los nidos retirados por cuestiones de seguridad, principalmente en tendidos eléctricos durante esta temporada (A. López-García com. pers.), de los cuáles al menos un tercio se habrían retirado en periodo reproductor.

Por ello, la población reproductora de cigüeña blanca en la Comunidad de Madrid se establece entre 2.176 y 2.327 parejas, siendo la mayor cifra hasta ahora registrada para esta región (Molina y Del Moral 2005; Aguirre y Vergara 2009).

Distribución de la población

En este estudio se confirmó la reproducción de cigüeña blanca en 107 municipios madrileños, más de la mitad de los nidos se concentraron en tan sólo siete municipios (tabla 1). En el otro extremo, 45 de los 107 municipios (42%) tan sólo registró una pareja reproductora.

La mayor colonia se registró entre los municipios de Soto del Real y Colmenar Viejo y estuvo formada por 212 parejas reproductoras. En la ciudad de Madrid destacó la colonia del parque Casa de Campo que englobó más del 80% de los nidos de este municipio.

Tabla 1: Número de nidos por municipio. Se han encontrado nidos de cigüeña blanca en un total de 107 municipios de la Comunidad de Madrid

Municipio	Nidos
Soto del Real	290
Colmenar Viejo	253
Getafe	222
Alcalá de Henares	129
Guadalix de la Sierra	122
Manzanares El Real	115
Madrid	105
Arganda del Rey	87
Boadilla del Monte	78
El Boalo	71
Rivas – Vaciamadrid	69
San Martín de la Vega	66
El Escorial	50
San Lorenzo de El Escorial	48
Ciempozuelos	45
San Fernando de Henares	39
Fuente el Saz de Jarama	36
Pinilla del Valle	34
Guadarrama	32
Buitrago del Lozoya	28
Velilla de San Antonio	28
Meco	19
Valdetorres de Jarama	19
Alameda del Valle	18
Rascafría	18
Lozoya	16
Torrelaguna	16
Gargantilla del Lozoya y Pinilla de Buitrago	15
Valdemorillo	15
San Agustín del Guadalix	14
Titulcia	14
Pedrezuela	13
Piñuécar – Gandullas	12
Paracuellos de Jarama	11
Tres Cantos	11
Aranjuez	10
Los Molinos	10
San Sebastián de los Reyes	10
Pinto	9
Villa del Prado	9
Garganta de los Montes	8
Gascones	7
Brunete	6
Moralzarzal	5
Villavieja del Lozoya	5
Cercedilla	4
Valdemoro	4

Municipio	**Nidos**
Chapinería	3
Galapagar	3
Majadahonda	3
Mejorada del Campo	3
Parla	3
Torrejón de Ardoz	3
Torrejón de Velasco	3
Algete	2
Becerril de la Sierra	2
Collado Villalba	2
La Serna del Monte	2
Las Rozas de Madrid	2
Navarredonda y San Mamés	2
Navas del Rey	2
Valdeolmos - Alalpardo	2
Ajalvir	1
Aldea del Fresno	1
Alpedrete	1
Arroyomolinos	1
Braojos	1
Bustarviejo	1
Cabanillas de la Sierra	1
Cadalso de los Vidrios	1
Camarma de Esteruelas	1
Canencia	1
Casarrubuelos	1
Cenicientos	1
Cobeña	1
Colmenar del Arroyo	1
Colmenarejo	1
Cubas de la Sagra	1
Daganzo de Arriba	1
El Álamo	1
El Vellón	1
Fresnedillas de la Oliva	1
Fresno de Torote	1
Griñón	1
Horcajo de la Sierra - Aoslos	1
Hoyo de Manzanares	1
Humanes de Madrid	1
La Cabrera	1
Lozoyuela - Navas – Sieteiglesias	1
Miraflores de la Sierra	1
Montejo de la Sierra	1
Moraleja de En medio	1
Navacerrada	1
Navalafuente	1

Municipio	Nidos
Navalcarnero	1
Quijorna	1
Ribatejada	1
Rozas de Puerto Real	1
San Martín de Valdeiglesias	1
Serranillos del Valle	1
Sevilla la Nueva	1
Torremocha de Jarama	1
Valdepiélagos	1
Villamanta	1
Villamantilla	1
Villaviciosa de Odón	1
Zarzalejo	1

La densidad de nidos en la Comunidad de Madrid prácticamente se duplicó desde el último censo, 29,0 pp/100 km^2 en 2021 frente a 15,7 pp/100 km^2 en 2004 (Molina y Del Moral 2005).

Por tanto, se confirma una vez más la fuerte agregabilidad de esta especie en esta región, con un 95% de nidos coloniales respecto del total censado.

El incremento en el número de nidos se tradujo en siete nuevas cuadrículas UTM de 10x10 km en las que se registraron nidos ocupados con respecto al censo de 2004 (UK96, VK25, VK28, VK32, VK34, VK49 y VL52), aunque dejó de observarse nidos ocupados en tres cuadrículas en las que se detectaron parejas en 2004 (VK43, VK65 y VL21). Siguió sin reproducirse en el sector sureste de la Comunidad de Madrid (figura 1).

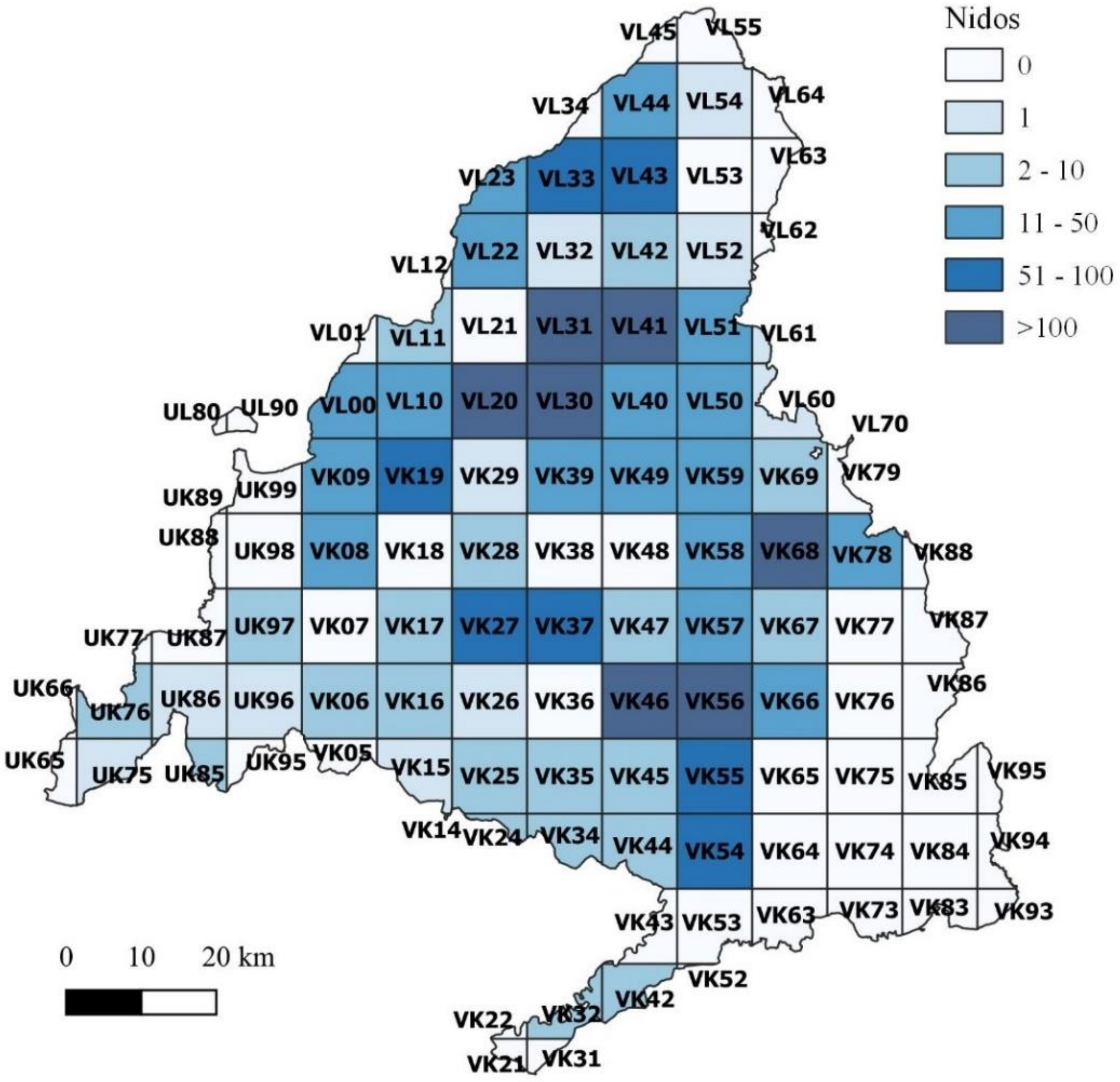


Fig. 1. Distribución de la población reproductora de cigüeña blanca en la Comunidad de Madrid en 2021. Se representa el número de nidos por cuadrículas UTM 10x10km en un gradiente de tonos azules y se representa el código de cada cuadrícula UTM en negrita.

Parámetros reproductivos

Se contabilizaron 2.729 pollos de 1.182 nidos. El número de pollos por nido varió de cero a cinco en la región, con una productividad total (JZa) de 1,91 volantones por nido. El éxito reproductor fue de 2,31 pollos por nido, similar al encontrado en censos anteriores y algo superior al del último (tabla 2).

Un 82% de las parejas de cigüeña que se observaron criando (HPa-HPx) sacaron adelante al menos un pollo este año.

Tabla 2. Comparación de los parámetros poblacionales y reproductivos de los últimos cuatro censos llevados a cabo en la Comunidad de Madrid (Lázaro *et al.,* 1986; SEO/BirdLife, 1995; Aguirre y Atienza, 2002; Molina y Del Moral, 2005) y el censo actual. **HPa**, Número de nidos ocupados; **HPm,** Número de parejas con pollos volantones; **HPo,** Número de parejas que fallan en la cría; **HPx**, Número de parejas en las que se desconoce el éxito reproductivo; **JZG,** Número total de volantones en la Comunidad de Madrid; **JZa,** Productividad total; **JZm,** Productividad parcial.

	1984	**1994**	**2001**	**2004**	**2021**
HPa	215	582	916	1.221	2.327
HPm	88	446	688	680	1.182
HPo	13	85	169	82	250
HPx	114	51	59	459	895
JZG	177	1.014	1.720	1.419	2.729
JZa	2,01	1,70	1,90	1,86	1,91
JZm	2,36	2,30	2,50	2,06	2,31

Nidotópica

Más de la mitad de los nidos (54%) se ubicaron en árboles mientras que un 43% se ubicó en sustratos no naturales (figura 2). Cambia por tanto la tendencia que había hasta ahora de disminución de nidos en sustrato natural frente a sustrato no natural: 47% en árboles en 2001 (Aguirre y Atienza 2002) y 43% en árboles en 2004 (Molina y Del Moral 2005). La preferencia por los árboles como lugares de nidificación posiblemente se deba al incremento de medidas disuasorias y nidos retirados en sustratos no naturales como edificios, apoyos de tendidos eléctricos y antenas de telefonía (Maricato *et al.*, 2016).

Se incrementó el número de nidos en antenas de telefonía móvil y ya supuso un 4% del total frente al 2,5% en 2001 (Aguirre y Atienza 2002), casi duplicó a los nidos en poste *ad hoc* (2,7%) y quintuplicó a los nidos en silo (0,7%; figura 2).

Cabe resaltar que el número de nidos en tendidos eléctricos sería sustancialmente más alto de no ser por la continua retirada de los mismos en estas infraestructuras debido al riesgo de interrupciones del suministro eléctrico o de incendio.

La categoría “Otros” engloba nidos en vallas publicitarias, farolas, grúas o vehículos abandonados; y muestra la versatilidad de esta especie para criar.

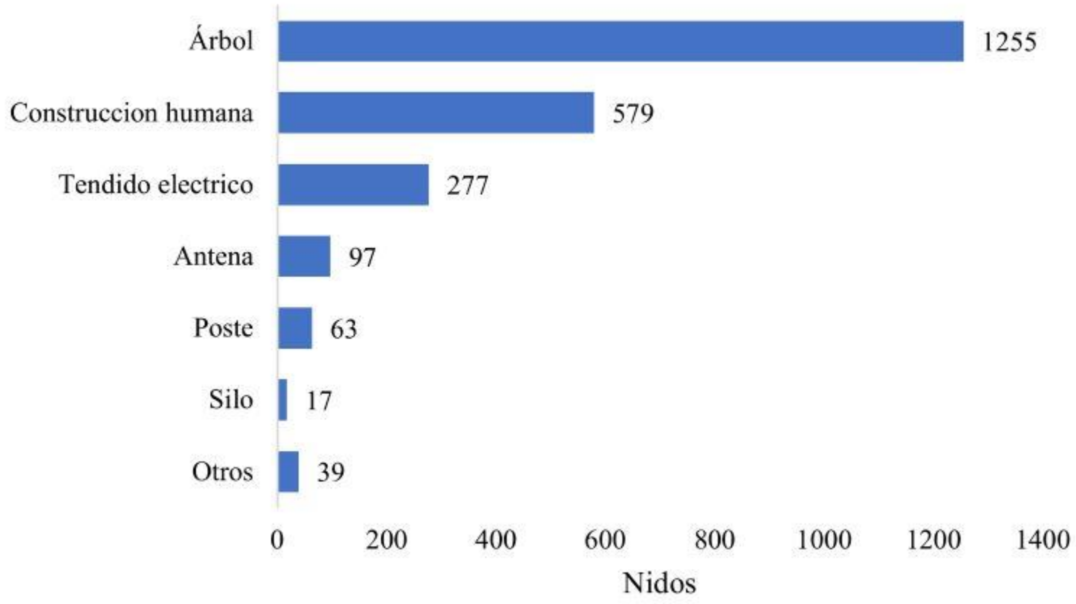


Fig. 2. Número de nidos de cigüeña blanca en función del tipo del sustrato de nidificación en la Comunidad de Madrid en 2021

Conclusión

Los resultados del censo muestran que la evolución de la especie ha sido positiva, casi duplicó la población reproductora de hace 17 años (tabla 2). Este fuerte crecimiento se asocia habitualmente al uso de vertederos, que lleva consigo un incremento en el número y tamaño de los huevos, así como un incremento en el número de pollos y mejor condición corporal de los mismos, un incremento en la supervivencia de los juveniles al reducir o eliminar la migración transahariana y los costes asociados a la misma (Tortosa *et al.* 2002; Djerdali *et al.* 2008, 2016; Pineda-Pampliega *et al.* 2021). Sin ir más lejos, es curioso observar que los ochos municipios con mayor número de nidos (tabla 1), se encuentran en las proximidades de los principales vertederos de la Comunidad de Madrid (vertederos de Colmenar Viejo, Alcalá de Henares y Madrid ciudad). Además, la alimentación en los vertederos induce el acortamiento, o incluso, la supresión de la migración de esta especie transahariana, reduce los costes asociados a este comportamiento e incrementa la supervivencia juvenil (Rotics *et al.* 2017; Bécares *et al.* 2019; Cheng *et al.* 2019).

Si bien el seguimiento de determinadas colonias en Madrid señala la disminución en el crecimiento de sus efectivos (Prieto 2002, datos propios), es difícil saber si la población se encuentra cerca de su estabilización o si ya se estabilizó con los datos que disponemos.

La productividad total y parcial fue similar a la de los censos anteriores (tabla 2), lo que apoya el buen estado de la población actual.

No obstante, la población reproductora no creció al mismo ritmo que en el periodo comprendido entre 1984-2004 (tabla 2). Posiblemente esto se deba a que en 1984 la población partía de una situación de declive, por lo que el crecimiento potencial de la especie era mayor debido a una menor competencia intraespecífica y mayor disponibilidad de nicho y al mayor apoyo de la sociedad (Ave del año en 1992, campañas para postes de nidificación artificiales, etc.). Actualmente, aunque el estado poblacional de esta especie parece asegurado, se enfrenta a una serie de amenazas potenciales que deben tenerse en cuenta. La dependencia de los vertederos como fuente de alimentación hace que los cambios en su gestión indicados por la normativa europea puedan provocar la falta de accesibilidad a este recurso por parte de las aves. La deficiencia en la calidad de los alimentos recibidos por parte de los pollos que proceden de los vertederos puede hacer también que la aparente buena condición a corto plazo se traduzca en disminución de la supervivencia a medio y largo plazo (Peris 2003; López-García *et al.* 2021; Pineda-Pampliega *et al.* 2021). A esta situación se une el riesgo de colisión contra las líneas de alta tensión y la degradación o desaparición de áreas de alimentación naturales, debido fundamentalmente a la intensificación agrícola (Garrido y Fernandéz-Cruz 2003; Tobolka 2014; Moreira *et al.*, 2017; Orłowski *et al.*, 2019; Marcelino *et al.*, 2021). Unas densidades más altas, pueden también provocar un aumento en los conflictos de esta especie con el ser humano.

Finalmente, pese a la buena situación de la especie en esta región, existe la necesidad de realizar un seguimiento periódico de dicha población con el fin de analizar los efectos del cierre de vertederos y los cambios de gestión de los residuos, como el de Alcalá de

Henares. Esperamos que, a diferencia de este censo, que se realizó de forma altruista, las administraciones públicas competentes y otras entidades relevantes tengan a bien financiar y apoyar el censo nacional decenal, que se realizó hasta 2004, y no pudo realizarse en 2014 en España y que consideramos imprescindible que se realice en 2024.

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Bibliografía

Aguirre J.I. y Atienza J.C. 2002. Censo de la población reproductora de Cigüeña Blanca (*Ciconia ciconia*) en la Comunidad de Madrid. Año 2001. En: Seo/Birdlife (ed.) *Anuario Ornitológico de Madrid 2001*: 114–125. Seo/Birdlife, Madrid.

Aguirre J.I. & Vergara P. 2009. Census methods for White stork (*Ciconia ciconia*): Bias in sampling effort related to the frequency and date of nest visits. *Journal of Ornithology*: 150:147–153.

Barbarin J.M.; Alonso D.; Arizaga J.; *et al.* 2021. Breeding population trends and recent changes in the nesting behaviour of the White Stork *Ciconia ciconia* L., 1758 in Navarre, north of Spain. *Munibe Ciencias Naturales*. doi: 10.21630/mcn.2021.69.01

Barlein F. 1991. Population studies of white storks (*Ciconia ciconia*) in Europe. En: Perrins C.; Lebreton J.D.; Hirons R. (ed.) *Bird population studies: Relevance to conservation and Management*: 207–229. Oxford University Press, Oxford.

Bécares J.; Blas J.; López-López P.; *et al.* 2019. Migración y ecología espacial de la cigüeña blanca en España, Monografía. SEO/BirdLife, Madrid.

Bernis F. 1981. La población de cigüeñas españolas. Estudios y tablas de censos, período 1948-1974. Cátedera de Zoología de vertebrados, Universidad Complutense, Madrid.

Cheng Y.; Fiedler W.; Wikelski M.; Flack A. 2019. “Closer-to-home” strategy benefits juvenile survival in a long-distance migratory bird. *Ecology Evolution*, 9:8945–8952.

Djerdali S.; Guerrero-Casado J.; Tortosa F.S. 2016. Food from dumps increases the reproductive value of last laid eggs in the White Stork *Ciconia ciconia*. *Bird Study* 3657:1–8.

Djerdali S.; Tortosa F.S.; Hillstrom L.; Doumandji S. 2008. Food supply and external cues limit the clutch size and hatchability in the White Stork *Ciconia ciconia*. *Acta Ornithologica*, 43:145–150.

Garrido J.R. y Fernandéz-Cruz M. 2003. Effects of Power Lines on a White Stork *Ciconia ciconia* population in central Spain. *Ardeola*, 50:191–200

Lázaro E.; Chozas P.; Fernandéz-Cruz M. 1986. Demografía de la cigüeña blanca (*Ciconia ciconia*) en España. Censo Nacional de 1984. *Ardeola*, 33:131–169

López-García A.; Sanz-Aguilar A.; Aguirre J.I. 2021. The trade-offs of foraging at landfills: Landfill use enhances hatching success but decrease the juvenile survival of their offspring on white storks (*Ciconia ciconia*). *Science of the Total Environment*, 778:146217. doi: 10.1016/j.scitotenv.2021.146217

Marcelino J.; Moreira F.; Franco A.M.A.; *et al.* 2021. Flight altitudes of a soaring bird suggest landfill sites as power line collision hotspots. *Journal of Environmental Management*, 294. doi: 10.1016/j.jenvman.2021.113149

Maricato L.; Faria R.; Madeira V.; *et al.* 2016. White stork risk mitigation in high voltage electric distribution networks. *Ecological Engineering*, 91: 212–220.

Massemin-Challet S; Gendner J.P; Samtmann S.; *et al.* 2006. The effect of migration strategy and food availability on White Stork *Ciconia ciconia* breeding success. *Ibis*, 148:503–508.

Molina B. y Del Moral J.C. 2005. La cigüeña blanca en España. VI Censo de Internacional. 2004. SEO/Birdlife, Madrid.

Moreira F.; Encarnacão V.; Rosa G.; *et al* (2017) Wired: Impacts of increasing power line use by a growing bird population. *Environmental Research Letters*, 12: doi: 10.1088/1748-9326/aa5c74

Orłowski G.; Karg J.; Jerzak L.; *et al.* 2019. Linking land cover satellite data with dietary variation and reproductive output in an opportunistic forager: Arable land use can boost an ontogenetic trophic bottleneck in the White Stork *Ciconia ciconia*. *Science of the Total Environment*, 646:491–502.

Peris S.J. 2003. Feeding in urban refuse dumps: ingestion of plastic objects by the White Stork (*Ciconia ciconia*). *Ardeola*, 50:81–84.

Pineda-Pampliega J.; Ramiro Y.; Herrera-Dueñas A.; *et al.* 2021. A multidisciplinary approach to the evaluation of the effects of foraging on landfills on white stork nestlings. *Science of the Total Environment*, 775: doi: 10.1016/j.scitotenv.2021.145197

Prieto J. 2002. Las Cigüeñas de Alcalá. Escuela Taller de Medio Ambiente Albardín. Madrid, Alcalá de Henares.

Rotics S.; Turjeman S.; Kaatz M.; *et al.* 2017. Wintering in Europe instead of Africa enhances juvenile survival in a long-distance migrant. *Animal Behaviour*, 126:79–88.

Schulz H. 1999. The world population of the White Stork (*Ciconia ciconia*). En: Schulz H. (ed.) *White Storks on the up? Proc. International Symposium on the White Stork, NABU*: 351–365. Bonn, Hamburg.

SEO/Birdlife. 1995. *V Censo Nacional de Cigüeña Blanca 1994*. Informe inédito para la Comunidad de Madrid.

Thomsen K-M.; Kaatz C.; Kaatz M.; Ludwig B. 2017. Verbreitung und Bestand. En: Kaatz C.; Wallschläger D.; Dziewiaty K.; Eggers U. (eds.) *Der Weißstorch Ciconia ciconia.* NBB, 682: 148–205- Magdeburg: Verlags K.G. Wolf.

Tobolka M. 2014. Importance of juvenile mortality in birds' population: Early post-fledging mortality and causes of death in white stork *Ciconia ciconia*. *Polish Journal of Ecology*, 62: 807–813.

Tortosa F.S.; Caballero J.M.; Reyes-López J. 2002. Effect of rubbish dumps on breeding success in the White stork in Southern Spain. *Waterbirds*, 25: 39–43.

Wuczyński A.; Krogulec G.; Jakubiec Z.; *et al.* 2021. Population size and spatial distribution of the white stork *Ciconia ciconia* in Poland in 1958 with insights into long-term trends in regional and global population. *European Zoological Journal*, 88: 525–539.

GLOSARIO DE TABLAS Y FIGURAS

INTRODUCCIÓN GENERAL

Fig. 1. Mapa de la cantidad de residuos orgánicos generados (Kg) por habitante al año en cada país del mundo **22**

Fig. 2. Efectos del uso de los vertederos en las poblaciones reproductoras de cigüeña blanca analizados en esta tesis **27**

MATERIAL Y MÉTODOS

Fig. 1. Número de parejas reproductoras de cigüeña blanca en la Comunidad de Madrid desde 1948.......... **32**

Tabla 1. Principales Vertederos de Residuos Sólidos Urbanos de la Comunidad de Madrid hasta la actualidad, ordenados según fecha de apertura **33**

Fig. 2. Mapa de los vertederos y los tipos de hábitat principales de la Comunidad de Madrid. **34**

Fig. 3. Cigüeña marcada con anilla de lectura a distancia alimentándose en vertedero.......... **38**

CHAPTER 1 - CAPÍTULO 1

Table 1. Output from the GLM for probability of nest occupation before white storks began to use landfills (1984).......... **49**

Fig. 1 The probability of nest occupation in relation to the percentage of other agricultural fields (A), arable lands (B), forest (C), pastures & dehesas (D), degree of urbanization (E) and, distance to water bodies (F) **50**

Table 2. Nest density relationship to the landfill distance, controlling by percentage of other agricultural fields (Agri), arable lands (Arable), forest (Forest), pasture and dehesas (PD), degree of urbanization (Rural, Periurban or Urban), and distance to water bodies (Water) **51**

Fig. 2. Effect of landfill distance on nest density. Nest density is higher in 2021 and 2004 than in 1984, before white storks began utilizing landfills.......... **52**

Fig. 3. Differences in nest density (nests/km^2) as a function of degree of urbanization (with 95% confidence intervals) **52**

Table 3. Differences amongst permanent breeding areas to new nesting sites and abandoned nesting sites.......... **53**

Fig. 4. The relationship between the probability of new nesting sites in 2004 and 2021 and the percentage of other agricultural fields (Agri), percentage of arable lands (Arable), percentage of forest (Forest), percentage of pastures & dehesas (Pastures & dehesas), distance to the nearest landfill (Landfill distance), degree of urbanization, and distance to water bodies **55**

Fig. 5. The relationship between sites in 2004 and the percentage of other arable lands (A), percentage of forest (B), distance to the nearest landfill (C), and degree of urbanization (D) **56**

Fig. S1. Changes on nest density (nests/ km2) and distribution of the breeding population of white storks in the province of Madrid **60**

Table S1. Model selection based on the Akaike Information Criterion adjusted for small sample sizes (AICc) in probability of nest occupation before the use of landfills (1984), nest density, new nesting sites and abandoned nesting sites. **60**

CHAPTER 2 - CAPÍTULO 2

Fig. 1. The map representing the limits of the Madrid region (black solid line), the buffer of 30 km around each landfill (dashed line), and the locations of the landfills (triangles) **71**

Table 1. Variation in the nest-site selection index in relation to the landfill distance and census year. We include additive (+) and interaction (∗) effects **77**

Fig. 2. Effect of the landfill distance category on the nest-site selection index: a) Arable lands, b) Urban area, c) Forest habitat, and d) Pasture & dehesas area **78**

Fig. 3. Probability of producing at least one nestling (breeding outcome) related to the proportion of Pastures & Dehesas surrounding the nests **79**

Table 2. Effects of landfill distance and the proportion of habitat selected on breeding success **80**

Fig. 4. Number of fledglings in relation to landfill distance **81**

Table S1. Corine Land Cover classes into seven Habitats categories, as functional Land cover groups, and percentage of cover of each habitat category in our study area in 2001, 2004, and 2021 .... **88**

Table S2. Nest-site Ranking Index by breeding white storks in relation to distance to landfill along several years **89**

Fig. S1. Nest-site Ranking Index by white storks in Madrid region for 2001 (A), 2004 (B) and 2021 (C) **91**

CHAPTER 3 - CAPÍTULO 3

Fig. 1. Hatching success in relation to Landfill Use Index **103**

Table 1. Reproductive parameters in relation to Landfill Use Index by breeders, age of breeder, and year of breeding **103**

Table 2. Modelling of White Stork survival probabilities depending on age structure, age of parents, Landfill Use Index by parents, and family resemblance (cluster effect)............ **106**

Fig. 2. Effect of Landfill Use by parents on future juvenile survival probability (and 95% CI given by dashed lines) **107**

Fig. 3. Representation of the annual probability that a laid egg becomes a one-year-old stork (HS*FS*ϕj) next year as a function of parental age and landfill use............................... **107**

Table A.1: Number of pairs classified by age and year of breeding. We represent number of pairs of each age during our four breeding seasons **114**

Table A.2: Modelling of White Stork resighting probabilities... **114**

Fig. A.1: Representation of the annual probability that a laid egg becomes a one-year-old stork (HS*FS*ϕj) next year as a function of parental age and landfill use............................... **115**

CHAPTER 4 - CAPÍTULO 4

Fig. 1. Map of our study area, the Community of Madrid. **123**

Table 1. Predicted suitability variation considering different environmental variables under each respective climate-change scenario **128**

Fig. 2. Habitat suitability in 2021 (A) and in future scenarios (B) **130**

Fig. 3. Importance of the variables considered in our models: in 2021 (A) and in 1984 (B) with different algorithms... **131**

Fig. 4. Differences in the environmental variables between the presence points and absence points **132**

Fig. 5. Changes on potential breeding population distribution of White stork in Community of Madrid in absence of food availability from landfills and different climate change future scenarios (ssp1 and ssp5) **133**

Table S1. Description of variables included in our models, including the year and source of the dataset used. **138**

Fig. S1. Dendrogram obtained from the cluster analysis that was employed to select the relevant variables (highlighted in blue); the red horizontal line indicates the distance-threshold to form the clusters (0.3, i.e., less than 70% correlation)............................... **139**

LÍNEAS DE INVESTIGACIÓN FUTURAS

Fig. 1. Fotografía de la planta de gestión de residuo de Loeches que muestra la inaccesibilidad de los residuos para las aves que supone el cambio de gestión de los vertederos **150**

ANEXO I

Tabla 1: Número de nidos por municipio. Se han encontrado nidos de cigüeña blanca en un total de 107 municipios de la Comunidad de Madrid **194**

Fig. 1. Distribución de la población reproductora de cigüeña blanca en la Comunidad de Madrid en 2021 .... **197**

Tabla 2. Comparación de los parámetros poblacionales y reproductivos de los últimos cuatro censos llevados a cabo en la Comunidad de Madrid (Lázaro et al., 1986; SEO/BirdLife, 1995; Aguirre y Atienza, 2002; Molina y Del Moral, 2005) y el censo actual.................. **198**

Fig. 2. Número de nidos de cigüeña blanca en función del tipo del sustrato de nidificación en la Comunidad de Madrid en 2021.. **199**

FE DE ERRATAS

En esta tesis Titulada:

"Impact of landfills on breeding populations of White stork (*Ciconia ciconia*) and future perspectives"

"Impacto de los vertederos en las poblaciones reproductoras de Cigüeña blanca (*Ciconia ciconia*) y perspectivas de futuro"

En la Fig.4 de la página 55, las gráficas 2021c y 2021d son erróneas y deberían ser como se encuentran en la hoja adjunta y no como está incluida dentro del documento final.

Aguirre

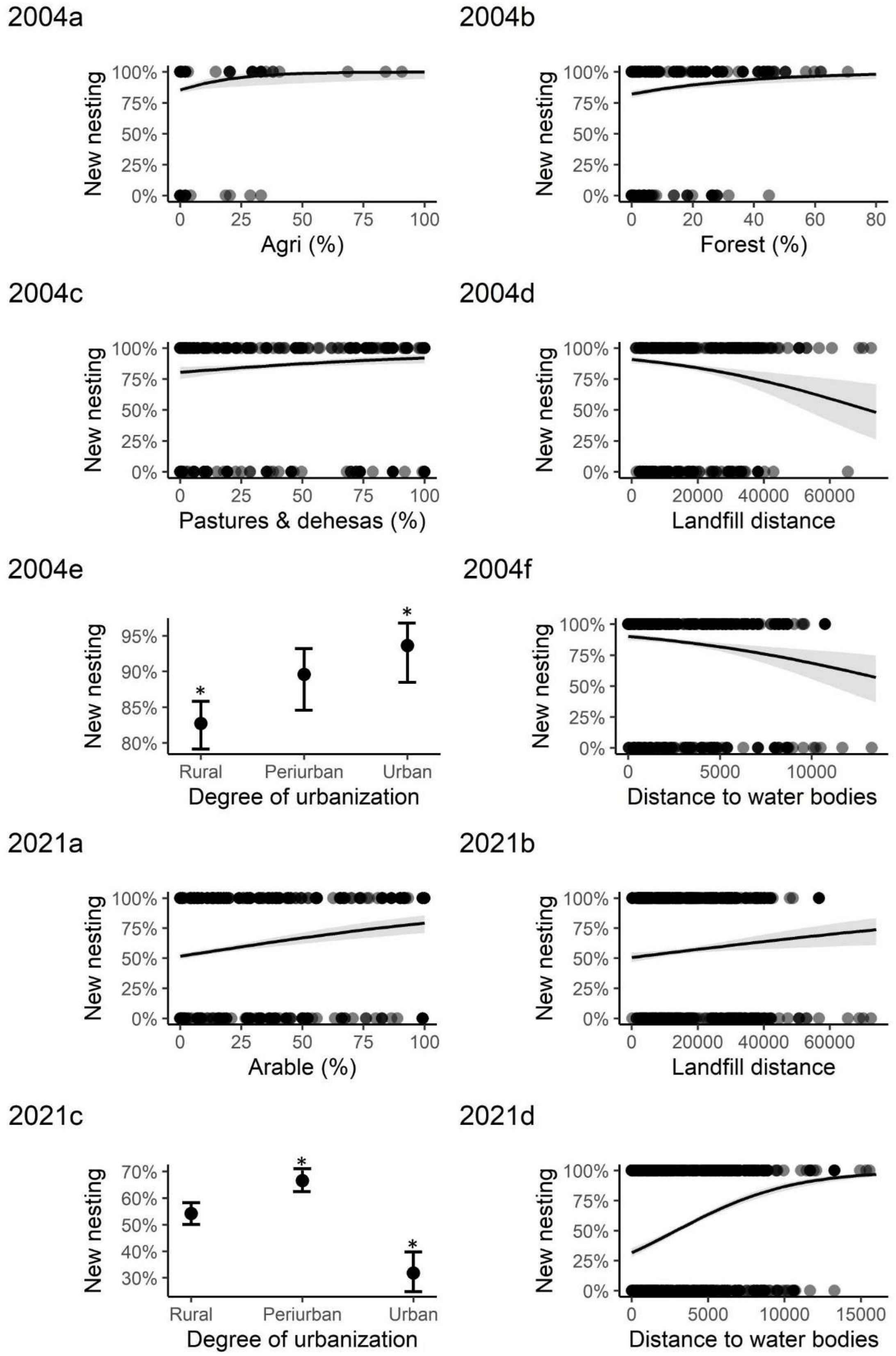

Fig. 4. The relationship between the probability of new nesting sites in 2004 and 2021 and the percentage of other agricultural fields (Agri), percentage of arable lands (Arable), percentage of forest (Forest), percentage of pastures & dehesas (Pastures & dehesas), distance to the nearest landfill (Landfill distance), degree of urbanization, and distance to water bodies. In general, the probability of new nesting sites was higher in 2004 than in 2021. Shaded areas show the 95% confidence intervals. In the degree of urbanization plot, whiskers show the 95% CI and asterisks represent significant differences. We only show significant differences.