Impacto del cierre de vertederos en la demográfia de especies oportunistas: El caso de la gaviota amarilla en el País Vasco.

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Tesis doctoral 2021

eman ta zabal zazu

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“Ya siente el impulso, el ***zugunruhe****.*

Ejercita sus alas, su habilidad para capturar insectos, su sentido de la orientación. Tras cada puesta de sol cubre un pequeño tramo hacia el sur.

Su viaje ha comenzado.”

¿Para qué sirven las aves?
Antonio Sandoval Rey.

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IMPACTO DEL CIERRE DE VERTEDEROS EN LA DEMOGRAFÍA DE ESPECIES OPORTUNISTAS:

EL CASO DE LA GAVIOTA PATIAMARILLA EN EL PAÍS VASCO

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Tesis Doctoral

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Contenido

RESUMEN

Las especies oportunistas se definen como aquellas que tienen un bajo nivel de especialización y que son capaces de adaptarse a una amplia variedad de condiciones, incluidos los cambios generados por la actividad humana. En este contexto, muchas especies de gaviotas (Aves, género *Larus*) se han visto beneficiadas por la superabundancia de alimento generado a partir de ciertas actividades, como la creación de vertederos a cielo abierto o la generación de descartes pesqueros. La gaviota patiamarilla *L. michahellis* es una de esas especies de gaviotas que han experimentado un crecimiento poblacional sin precedentes como consecuencia de su capacidad para explotar tales recursos tróficos. Esta superabundancia de alimento, que durante un tiempo ha beneficiado a las poblaciones de gaviotas patiamarrillas en prácticamente toda su área de distribución, se está revertiendo en la actualidad debido a las nuevas legislaciones que, en su conjunto, velan por una actividad humana más sostenible que no genere residuos orgánicos. Esto se traduce en el cierre de vertederos a cielo abierto, por un lado, así como la reducción de descartes pesqueros, por otro. Esta nueva realidad ha devuelto a la disponibilidad de alimento un peso demográfico importante como factor limitante del crecimiento poblacional.

Tras un crecimiento sostenido, la población de gaviota patiamarilla en País Vasco comienza a mostrar signos de estabilización o, incluso, regresión demográfica. Coincide este cambio con la aplicación de nuevos modelos de gestión de los residuos urbanos sólidos en el territorio, incluido el cierre de vertederos. En este contexto, esta población conforma un modelo biológico ideal para comprender la respuesta de una especie oportunista ante un cambio radical en la disponibilidad de un recurso trófico clave. El objetivo general de esta tesis, así, se centra en calcular los principales parámetros demográficos de la población de gaviota patiamarilla en País Vasco, determinar el cierre de vertederos en dichos parámetros y establecer modelos predictivos para

proyectar la tendencia poblacional de la especie ante diferentes escenarios tróficos.

En la primera parte de la tesis se calculan diferentes parámetros demográficos de la población estudiada. Tiene carácter descriptivo, ya que varios de estos parámetros no habían sido calculados previamente o no estaban actualizados. El **primer artículo** de la tesis se centra en describir los principales parámetros reproductivos de la población, con el objeto de obtener estimas realistas que puedan ser aplicadas, posteriormente, en modelos poblacionales. Este capítulo se fundamenta en un trabajo de campo llevado a cabo a lo largo de tres años en la colonia de Ulia, la más importante de Gipuzkoa. El **segundo artículo** analiza el patrón de movimientos de la población y su posible variación en función del sexo y edad. En un contexto demográfico, su valor radica en determinar el grado dispersión natal y/o dispersión reproductiva y su posible variación por sexos. El **tercer artículo** analiza la filopatría y flujo entre las tres colonias más importantes de Gipuzkoa, aspectos aplicados a evaluar si las colonias funcionan como una sola población o si presentan algún tipo de estructura de carácter meta-poblacional.

En la segunda parte de la tesis se estima específicamente el impacto del cierre de vertederos en la supervivencia y se construyen modelos poblacionales asumiendo diferentes escenarios que, en todo caso, parten de las estimas obtenidas en el resto de los artículos de la tesis. Así, en el **cuarto artículo** se examina el efecto del cierre de vertederos sobre la supervivencia de la población. Finalmente, en el **quinto artículo** se modeliza la tendencia de la población asumiendo diferentes escenarios, en los que existe afección sobre los parámetros reproductivos y/o la supervivencia de las diferentes clases de edad. Estos modelos asumen el impacto demográfico potencial asociado al cierre total de vertederos y, en general, descenso generalizado en la disponibilidad de recursos tróficos.

En su conjunto, la tesis no solo contribuye a estimar un amplio número de parámetros demográficos de la población de estudio, sino que aporta proyecciones realistas que permiten predecir cuál va a ser la evolución demográfica de esta población si la disponibilidad de recursos tróficos continúa su actual declive.

INTRODUCCIÓN

La actividad humana contribuye de manera significativa a modificar la abundancia y disponibilidad de recursos tróficos que quedan a disposición de las especies animales que son capaces de explotarlos (Chamberlain *et al*. 2005; Agudo *et al*. 2010; Oro *et al*. 2013). El consumo de tales recursos puede tener importantes efectos en diversos aspectos de la biología de las especies, incluida la dinámica y tendencia demográfica de las poblaciones (Douglas *et al*. 2014), tanto en sentido negativo (desaparición de especies o alteraciones en la salud individual de los individuos (Matejczyk *et al*. 2011) como positivo (crecimiento de poblaciones). El medio marino no es ajeno a este fenómeno y, por ejemplo, es bien conocido que los descartes pesqueros han favorecido el crecimiento de poblaciones de aves marinas (Oro *et al*. 1995; Annett & Pierotti 1999; Weiser & Powell 2011; Cama *et al*. 2012). Algunas especies de aves marinas de carácter oportunista, como las gaviotas, han aprendido, además, a explotar otros recursos alternativos, como el caso de la basura orgánica presente en los vertederos a cielo abierto (Moreno *et al*. 2009; Ramos *et al*. 2009; Arizaga *et al*. 2013). Esta adaptación trófica también se da en otras especies de aves pertenecientes a diversos grupos taxonómicos, como cigüeñas, marabús y ardeidas (Nyangababo 2003; Ciach & Kruszyk 2010; Gilbert *et al*. 2016), rapaces diurnas como milanos, buitres o alimoches (De Giaccomo & Guerrieri 2008, Tauler-Ametller *et al*. 2018; Rabaca *et al. 2020*), o estorninos (Gewurtz *et al*. 2018). En este contexto es interesante destacar la revisión llevada a cabo por Plaza & Lambertucci (2017). Fuera de las aves, los vertederos han modificado también la conducta de diversas especies de mamíferos, como ocurre en el caso de ratas, (Schroder & Hulse 1979;), lobos (Meriggi & Lovari 1996), osos (Peirce & Van Daele 2006) o zorros (Hutchings 2003). Conocer en detalle la relación entre recursos tróficos y demografía es básico para cuantificar la influencia de la variabilidad ambiental causada por la existencia de alimento de origen humano en la dinámica de las poblaciones, así como para proporcionar a los gestores del medio natural información útil en la toma de decisiones (Duhem *et al.* 2002; Duhem *et al.* 2008; Payo-Payo *et al.* 2015; Steigerwald *et al.* 2015).

En el caso de las gaviotas, por ejemplo, ante su crecimiento poblacional en determinadas regiones las administraciones han desarrollado en ocasiones programas de descaste o erradicación (Smith & Carlile 1993, Sanz-Aguilar *et al.* 2009, Paracuellos & Nevado 2010), aunque a menudo de dudosa eficacia (Bosch *et al.* 2000, Castège *et al.* 2016) o con una efectividad a corto plazo (Magella & Brousseau 2001). A menudo, ocurre que los esfuerzos por controlar los efectos de estos recursos en las poblaciones animales se han concentrado en paliar las consecuencias, sin entrar a gestionar los factores que las causan o sin entender los efectos demográficos de las medidas tomadas (Baxter & Allan 2006; Baxter *et al.* 2007; Cook *et al.* 2008).

La relación entre los vertederos y las diferentes especies de gaviotas ha sido ampliamente estudiado a lo largo de las últimas décadas (Plaza & Lambertucci 2017). En general, esta relación se ha centrado en determinar el impacto de la apertura o existencia de los mismos sobre las diferentes poblaciones y especies (Fenlon 1983; Duhem *et al.* 2008; La Sala *et al.* 2013; Oro *et al.* 2013; Douglas *et al.* 2014; Coulson 2015; Plaza & Lambertucci 2017), mientras que el número de estudios donde se analiza el efecto contrario es mucho menor (Pons & Migot 1995; Kilpi & Öst 1998; Steigerwald *et al.* 2015). En parte, esto es debido a que el cierre de vertederos constituye un fenómeno relativamente reciente y es ahora, por tanto, cuando empiezan a acumularse un número suficiente de evidencias que permiten evaluar el impacto de esta nueva realidad. De forma generalizada, los vertederos tienen una influencia directa en el aumento de las poblaciones de gaviotas que se alimentan de los mismos (Bosch *et al.* 1994, Duhemn *et al.* 2008), favoreciendo además la creación de nuevas colonias (esto es, aumentando el área de distribución) (Lenda *et al.* 2010; Skorka *et al.* 2015) y promoviendo la sedentarización de las poblaciones (Jonsson 1998). En contraposición, parece ser que los vertederos aportan nutrientes de peor calidad (comparados por ejemplo con presas de origen marino como peces) (Ramos *et al.* 2009, Dosch 1997, Zorrozua *et al.* 2020a) y aumentan el riesgo de contraer enfermedades (Ortiz & Smith 1994; Fenlon 1983; La Sala *et al.* 2013; Jurinovic *et al.* 2014). La excesiva dependencia por los vertederos, además, vuelve a las poblaciones de gaviotas más vulnerables (a través de procesos de denso-dependencia) ante pequeños cambios en la disponibilidad de alimento, especialmente en aquellas zonas donde se han alcanzado niveles poblacionales muy elevados. En el lado opuesto, el cierre de vertederos está promoviendo reajustes en la ecología trófica (Zorrozua *et al.* 2020b) y se asocia al declive poblacional detectado ya en varias regiones dentro del área de distribución de varias especies de gaviotas (Steigerwald *et al.* 2015). En general, no obstante, el conocimiento sobre los efectos del cierre de vertederos sobre las poblaciones de gaviotas todavía es limitado y justifica, en consecuencia, una profundización en su análisis.

La gaviota patiamarilla en la costa vasca.

La gaviota patiamarilla *Larus michahellis* Naumann, 1840 es la especie de gaviota más común del Paleártico sudoccidental (Olsen & Larson 2004). Su área de distribución abarca la región circum-Mediterránea, Europa sudoccidental y Macaronesia (Olsen & Larson 2004). Durante las últimas décadas, además, la especie ha ido colonizando un buen número de zonas húmedas del interior de Europa, llegando incluso hasta Europa central (Jonsson 1998; Lenda *et al.* 2010; Skorka *et al.* 2015). El último censo estima una población de 409,000-534,000 parejas en el continente (BirdLife International 2021); 125.000-130.000 parejas (pp.) para toda España (Molina 2009), de las que unas 4.500 (3,5%) se reproducían en la costa vasca a mediados de la década de 2000 (Arizaga *et al.* 2009). En esta última región, la población aumento en torno a un 150% en tan solo 30 años (Arizaga *et al.* 2009), fenómeno en gran parte asociado a la existencia de grandes vertederos a cielo abierto. No obstante, a partir de mediados de la década de 2000 y, muy particularmente durante la década de 2010, empiezan a acumularse las evidencias que sugieren un declive poblacional importante, de nuevo asociado a vertederos, aunque esta vez a su cierre. Así, hasta la fecha, las colonias de cría ubicadas en Bizkaia han registrado un descenso de hasta un 50% en apenas 7 años (Galarza 2015). En el caso de las colonias que se encuentran en la provincia de Gipuzkoa, sin embargo, tal descenso no se llega a constatar o, al menos, las estimas que se han obtenido hasta la fecha no permiten concluir un declive tan acusado como el reportado para Bizkaia (Arizaga *et al.* 2014a). En todo caso, globalmente sí parece que la gestión de vertederos, encaminada a su cierre, sí está teniendo y/o va a tener efectos claros en la demografía de la especie.

Los estudios llevados a cabo en la última década constatan que la distancia al vertedero es un factor clave a la hora de determinar la importancia relativa de la basura orgánica en la dieta de la especie (Egunez *et al.* 2017). En ese contexto, el impacto demográfico de un vertedero sobre una colonia será tanto mayor cuanto más cerca esté esa colonia del vertedero. En términos globales, se estimó que en torno a un 40% de la dieta de las colonias de Gipuzkoa provenía de vertederos, otro 40% de recursos marinos (una parte de los cuales serían descartes pesqueros) y un 20% restante de presas terrestres, principalmente lombrices y babosas (Arizaga *et al.* 2011; Arizaga *et al.* 2013). En virtud de la actual política de la UE en materia de residuos (1999/31/EU and 2008/98/CE), los países europeos deben cerrar o gestionar de una forma más sostenible los vertederos a cielo abierto. Como consecuencia, nos encontramos actualmente en un escenario de cierre progresivo de vertederos, lo cual ha generado un marco experimental idóneo para comprobar el efecto de estos recursos sobre los parámetros demográficos de especies oportunistas, como la gaviota patiamarilla. Para estas poblaciones se ha comprobado que el cierre de vertederos afecta directamente a la dieta, en especial en época de cría, donde los individuos han ido supliendo la materia orgánica procedente de vertederos por presas terrestres, que han pasado a suponer hasta un 40-60% de la dieta (Zorrozua *et al.* 2020a).

Previsiblemente, la desaparición de este recurso podría acarrear una reducción de la capacidad de carga del sistema, lo cual podría tener efectos negativos sobre parámetros demográficos como la productividad y la supervivencia, así como favorecer la dispersión de los individuos en periodo no reproductor (Arizaga *et al.* 2014b). El hecho de que la población de gaviota patiamarilla en el País Vasco sea residente, y de que la mayoría de los individuos no suelen dispersarse más allá de 50 km desde sus colonias de origen (nacimiento) (Arizaga *et al.* 2010), permite estudiar adecuadamente el impacto de los cambios ambientales a escala local sobre los parámetros demográficos arriba mencionados. Hasta la fecha, se han visto ya efectos de dicho cierre en la ecología trófica, dispersión y tamaño de ciertas colonias. Falta determinar, sin embargo, el posible efecto del cierre de vertederos en los diferentes parámetros que, en última instancia, participan en los procesos que desembocan en los posibles cambios poblacionales. Tales parámetros serían, como ya se ha apuntado, la reproducción y la supervivencia. Saber en qué medida cada uno de estos parámetros es afectado es clave para elaborar modelos poblacionales que, en última instancia, nos permitan predecir la evolución demográfica de las poblaciones de gaviota patiamarilla en función del valor de cambio generado por el cierre de vertederos.

OBJETIVOS

La tesis integra dos objetivos principales:

1. Describir los principales parámetros demográficos de la población de gaviota patiamarilla en País Vasco en el escenario actual.

2. Cuantificar el impacto del cierre de vertederos en la demografía de la población y desarrollar proyecciones a futuro con el fin de valorar la capacidad de la población para adaptarse a nuevos escenarios en los que existe una disminución sustancial de recursos tróficos clave.

Para alcanzar estos objetivos generales se establecen los siguientes objetivos parciales:

1. Para elaborar modelos poblacionales es necesario disponer de buenas estimas sobre los parámetros reproductores. Ante la ausencia de esta información actualizada para las colonias de estudio, el primer objetivo parcial tiene como fin describir los principales parámetros reproductivos de la población estudiada, incluyendo un análisis sobre los posibles efectos del hábitat sobre la reproducción. Hipótesis: el éxito de eclosión es mayor en nidos que se encuentran protegidos por vegetación frente aquellos ubicados en zonas despejadas de la colonia.

2. Cuantificar el alcance de los movimientos de las poblaciones reproductoras de la costa vasca considerando posibles diferencias según el sexo y las clases de edad. Hipótesis: no existen diferencias relevantes entre las distancias recorridas entre ambos sexos a lo largo de los diferentes periodos del año. Los individuos juveniles e inmaduros realizaran desplazamientos superiores que los adultos en busca de nuevas áreas de alimentación y de cría.

3. Analizar el grado de filopatría de la población. Hipótesis por verificar: las diferentes colonias de cría localizadas en la costa de guipuzkoana presentan una alta filopatría, concordante con un comportamiento altamente sedentario que se asocia a un escenario donde la disponibilidad de alimento es alta y predecible en el espacio y el tiempo.

4. Determinar la tasa de supervivencia de la población estudiada desde 2005 hasta la actualidad y analizar su posible variación (previsible declive) a lo largo del tiempo, hasta la actualidad. El periodo de estudio abarca un escenario de cierre progresivo de vertederos, por lo que en este objetivo es posible comparar la supervivencia antes, durante y después del cierre de los mismos en el territorio. Hipótesis por verificar: la supervivencia de la población estudiada ha disminuido tras el cierre de vertederos y este efecto será más pronunciado en aquellas colonias más dependientes de este recurso trófico.

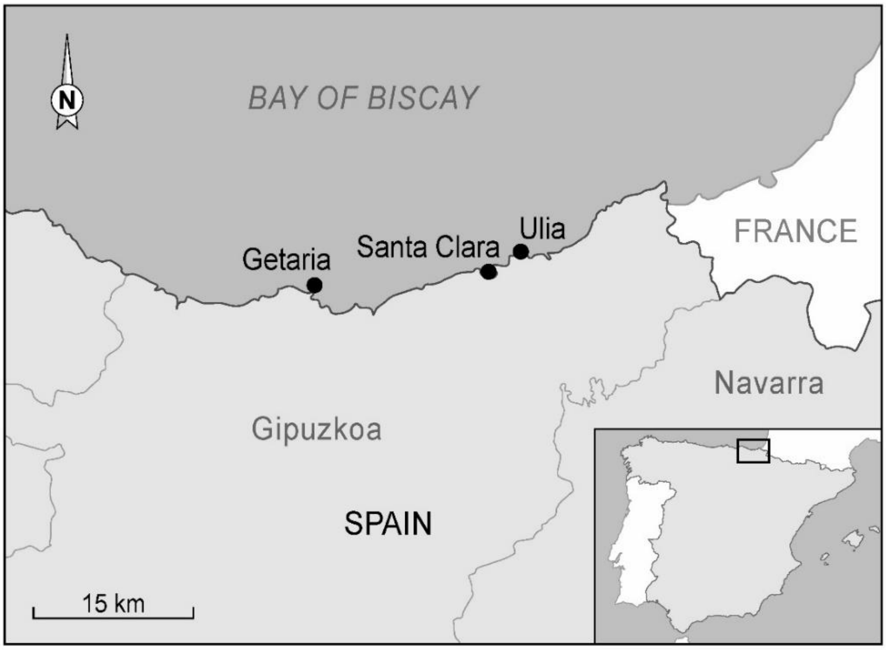
5. Construir un modelo demográfico que permita predecir la tendencia de la población estudiada a partir de diferentes escenarios donde varían los valores de los parámetros demográficos (reproducción y supervivencia). Hipótesis por verificar: La población de gaviotas patiamarrillas se encuentra en un punto en el que cualquier descenso de algún parámetro provoca el declive poblacional. Como especie que presenta una esperanza de vida elevada, la población será más sensible a cambios en la supervivencia de los adultos que a los posibles efectos en la supervivencia de otros grupos de edad o en la reproducción.

METODOLOGÍA

Área de estudio.

La mayor parte de esta tesis se ha desarrollado en la costa vasca, que es el tramo de litoral del mar cantábrico situado en el extremo oriental del norte de España y el extremo suroeste del lateral oeste de Francia. Los estudios desarrollados se han realizado mayoritariamente en las colonias ubicadas en la provincia de Gipuzkoa (Figura 1). Adicionalmente, se han empleado datos procedentes de una cuarta colonia, Izaro, situada en la vecina provincia de Bizkaia (esto es, al oeste de las citadas colonias de Gipuzkoa).

Figura 1. Distribución espacial de las colonias más importantes de gaviota patiamarilla en la provincia de Gipuzkoa.



La colonia de Getaria se sitúa en el término municipal de Getaria, en las laderas más abruptas del cabo de San Antón o Ratón de Getaria, situado al norte del casco urbano (43°31'N, 2°20'W). Esta colonia surgió a comienzos del presente siglo, aparentemente tras la apertura del vertedero de Urteta, hoy clausurado. Sin embargo, es una colonia que, a diferencia de las otras dos, presenta un alto consumo de presas de origen marino, lo cual se vincula a la actividad del propio puerto de Getaria (Zorrozua *et al*. 2020a). El último censo de la colonia (2017) estima 165 parejas reproductoras. La colonia de la isla de Santa Clara (43°32'N, 1°99'W) se sitúa en el término municipal de Donostia, en el centro de la Bahía de La Concha. La población reproductora en el año 2017 se calculó en 100 parejas. La colonia de Ulia (43°33'N, 01°95'W; Imagen 1) se sitúa, también, en el municipio de Donostia. Se extiende a lo largo de prácticamente toda la cara norte de la montaña de Ulia, asociada siempre a los acantilados más abruptos. El censo de 2017 arrojo 660 parejas reproductoras para el núcleo occidental de la colonia (el oriental, que cuenta con otras 100 parejas aproximadamente, se sitúa en el Faro de la Plata y no es objeto de muestreo). Esta colonia se sitúa a escasa distancia del puerto de Pasaia e, históricamente, como Santa Clara, ha dependido más de vertederos. Las tres colonias se asientan sobre acantilados de arenisca cubiertos por manchas más o menos variables de vegetación herbácea.

Imagen 1. Vista parcial de la colonia de Ulia, en la costa de Gipuzkoa.

Adicionalmente, para alguno de los artículos se han utilizado también datos provenientes del marcaje de pollos en la vecina colonia de Izaro, situada en Bizkaia. Izaro (43°25'N, 2°40'W; Imagen 2) es una isla situada en la desembocadura del río Oka (Urdaibai) y pertenece al municipio de Bermeo. En la actualidad cuenta con 400 parejas reproductoras, si bien llegó a alcanzar las más de 1600 a comienzos de la década de 2000 (Galarza 2015).

Imagen 2.. Vista de la isla de Izaro, situada frente a la costa de Bermeo, en Bizkaia.

Metodología.

A nivel metodológico, esta tesis se nutre de diferentes fuentes de información:

En lo que respecta al trabajo de campo y recolección de datos, buena parte de la tesis se basa en el anillamiento de pollos en colonias de cría de Gipuzkoa y su posterior avistamiento, a través de anillas de lectura a distancia (Imagen 3), una vez los juveniles ya volanderos abandonan las colonias. Este trabajo comenzó en 2005 y se ha llevado a cabo de manera ininterrumpida en las tres principales colonias de la región: Ulia, Santa Clara y Getaria. En total, durante el periodo 2005-2019 se han anillado más de 3.600 pollos que han dado lugar a más de 21.000 recuperaciones, casi todas ellas debido a avistamientos de ejemplares vivos. 7

Imagen 3. *Ejemplos de individuos marcados con anillas de PVC/PMMA de lectura a distancia utilizadas en el marcaje de gaviotas patiamarillas en la costa vasca*.

El anillamiento de gaviotas conlleva, invariablemente, la colocación de anilla metálica (remite Aranzadi) y anilla de PVC (al comienzo del proyecto) o PMMA (en la actualidad) para poder ser leída a distancia. Todos los marcajes son llevados a cabo en colonias, siendo la anilla de PVC/PMMA roja con código alfanumérico de cuatro caracteres en blanco, en los que el primer o último de ellos es la letra "G". La captura de pollos se desarrolla a mano, cuando éstos tienen, aproximadamente, 20 días de edad, entre los meses de junio y julio (normalmente, durante la segunda quincena de junio). Se anilla sólo una fracción del total existente en cada una de las colonias, estimada, globalmente, en un valor menor al 10% sobre el total de pollos presentes cada temporada (J. Arizaga, obs. per.).

Por otro lado, las estimas sobre parámetros reproductivos de esta tesis se han llevado a cabo mediante los correspondientes muestreos en la colonia de Ulia, durante las temporadas de cría de 2018, 2019 y de 2020. Estos muestreos han consistido en el seguimiento de la reproducción a través de la identificación y seguimiento de nidos en los que se anotó el tamaño de puesta, tamaño de los huevos y tasa de eclosión (porcentaje de huevos eclosionados). El motivo de elegir la colonia de Ulia fue por causas logísticas (el acceso a las otras colonias es muy complicado y no es posible plantear un seguimiento exhaustivo de la reproducción) y porque, además, actualmente es la colonia más grande del País Vasco. Los detalles de este seguimiento se describen en el Capítulo 1 de esta tesis.

En lo que respecta al análisis de datos y elaboración de modelos estadísticos, se han utilizado diferentes aproximaciones basadas en (1) modelos lineales mixtos generalizados (GLMM), corridos en R (RStudio, 2019), mediante los paquetes 'lme4' (Bates *et al.* 2015), 'lmerTest' (Kuznetsova *et al.* 2017) y 'MuMIn' (Barton 2014), aplicados principalmente en el Capítulo 1 y 2, sobre reproducción y movimientos, respectivamente; (2) modelos Cormack-Jolly-Seber y modelos multi-sate, basados en historiales individuales de captura-recaptura (esto es, anillamiento-avistamiento), ejecutados en MARK (White & Burnham 1999), aplicados a los Capítulos 3 a 5, sobre estimas de supervivencia y flujo entre colonias; (3) modelos para estimar la tasa de crecimiento poblacional a partir de matrices de población post-época de cría, ejecutadas a través del paquete 'popbio' para R (Caswell 2001; Stubben & Milligan, 2007), aplicados en los artículos 4 y 5.

RESULTADOS GENERALES

Artículo 1. No evidence of habitat effect on clutch size, egg quality and hatching success of the Yellow-legged Gull Larus michahellis at a micro-spatial scale.

El análisis de 267 nidos (752 huevos) durante tres periodos de cría consecutivos (2018-2020) ha permitido describir varios parámetros reproductivos, así como determinar el impacto del hábitat circundante sobre los mismos. Los resultados obtenidos en la colonia de Ulia ponen de manifiesto que variaciones en el porcentaje de cobertura vegetal (frente a suelo desnudo) alrededor del nido no afecta al tamaño de puesta, volumen de los huevos y éxito de eclosión.

Paralelamente, se ha observado que la fecha de puesta incide negativamente sobre el volumen de los huevos y el éxito de eclosión. Además, el volumen de los huevos y el éxito de eclosión tienden a reducirse a lo largo de las tres temporadas de cría analizadas. Las variables climatológicas (temperatura y precipitación durante el periodo reproductor) no afectan a ninguno de los parámetros reproductivos estudiados.

Artículo 2. No evidence supporting sex-dependent differential movements and survival in Yellow-Legged gulls.

El análisis de los datos aportados por 155 individuos marcados como pollos y sexados (mediante técnicas moleculares) muestra una dispersión media máxima de 60 km ± 6 km. Esta dispersión varía con la edad y la época del año, pero en ningún caso en función del sexo. Las distancias de dispersión son menores en adultos e inmaduros en la época de cría (de abril a junio) respecto a los juveniles.

Respecto a los valores su supervivencia se han obtenido unas tasas anuales de supervivencia aparente de 0,55 ± 0,05 para juveniles; 0.87 ± 0.03 para inmaduros; y 0.90 ± 0.04 para adultos. La supervivencia tampoco varía entre sexos.

Artículo 3. High philopatry rates of Yellow-legged gulls in the southeastern part of the Bay of Biscay.

Se analizaron los avistamientos en colonias en época de cría, generados por el marcaje 3285 pollos anillados en las colonias de cría de Ulia, Getaria y Santa Clara durante un periodo de 13 años. La inmensa mayoría de los ejemplares avistados tienen ya edad adulta, lo que indica que los ejemplares inmaduros permanecen alejados de los núcleos reproductores en el área de estudio al menos en periodo de cría. Solo 18 de un total de 853 avistamientos ocurrieron en otra colonia diferente a la colonia natal del ejemplar. Esto indica una alta filopatría (cercana al 99%). En individuos inmaduros estos valores caen hasta el 88% en el caso de Santa Clara, y suben para las colonias de Getaria y Ulia (92% y 95% respectivamente). Estos valores muestran una baja movilidad de los ejemplares entre las diferentes colonias de cría que se localizan cercanas.

En este trabajo también se analizó la supervivencia anual aparente de los diferentes grupos de edad, que varió de un 0.27 ± 0.02 para el primer año de vida hasta el 0.86 ± 0.01 registrado en aves de más de un año de vida.

Artículo 4. Demographic impact of landfill closure on a resident opportunistic gull.

El cierre de los diferentes vertederos presentes en el Pais Vasco y el efecto que ha podido tener en las diferentes colonias de gaviotas de la costa vasca se ha analizado gracias al anillamiento de 4437 individuos (marcados como pollos) y el posterior avistamiento de 2245 ejemplares del total de anillados. Los tres mejores modelos desarrollados ponen de manifiesto un efecto negativo del cierre de los vertederos sobre la supervivencia, especialmente cuando el vertedero se sitúa a menos de 10 km de la colonia. En particular, se observa un descenso acusado (efecto significativo) de la supervivencia en los ejemplares de primer año, no así en el caso de individuos de más edad, lo que sugiere que estas aves experimentadas, en términos globales, son capaces de compensar el cierre del vertedero.

El articulo también estima la tendencia poblacional de las cuatro colonias de estudio, antes y después del cierre de los vertederos localizados a menos de 10km. En términos generales, los modelos predicen una tendencia negativa una vez se cierra el vertedero.

Artículo 5. Modelling population trends of an opportunistic seabird species: the case of Yellow legged-gull in the Bay of Biscay.

Se elaboran modelos predictivos sobre la evolución de la tendencia poblacional ante diferentes escenarios que, en general, contemplan un impacto negativo sobre la reproducción y la supervivencia por reducción de recursos tróficos clave.

Primeramente, se desarrollan modelos de supervivencia para comprobar si este parámetro ha variado durante los últimos 15 años. Se observa una paulatina disminución de los valores de supervivencia anual tanto en aves de primer año como en aves de más edad. Por un lado, los individuos juveniles pasan de tener una supervivencia anual de 0.54 al principio del estudio a 0.24 en el último año analizado. Este descenso se da en menor medida en aquellos individuos que superan esta edad, siendo de 0.86 al comenzar el estudio a un valor ligeramente menor (0.78) en el año 2020.

Los modelos que estiman la tasa de crecimiento poblacional a partir de estos valores de supervivencia muestran para el año 2020 un descenso considerable de las poblaciones (0.913, lo que equivale a un descenso cercano al 9% anual). Un escenario más conservador tiendo en cuenta valores de supervivencia intermedios (año 2013) deriva en una tasa anual de crecimiento igual a 1.006. Los modelos estocásticos, que tienen en cuenta la variabilidad asociada a las estimas de los parámetros reproductivos o la supervivencia, predicen una población estabilizada al estimar un intervalo de confianza para la tasa de crecimiento poblacional anual que va de 0,97 a 1,06. A partir de este escenario de partida se llevaron a cabo modelos alternativos modificando (a la baja y en diferentes combinaciones) cinco variables (3 relacionadas con la supervivencia de los ejemplares y 2 relacionadas con variables reproductivas). De todos los modelos realizados, la disminución de hasta un 50% en parámetros como la supervivencia de juveniles/inmaduros o en las dos variables reproductivas tienen un efecto pequeño en el valor poblacional. Este efecto es mayor, sin embargo, cuando se trabaja en alguno de los diferentes escenarios con la supervivencia de los ejemplares adultos. Es decir, la población estudiada es especialmente sensible a cambios acontecidos en la supervivencia de las aves adultos, no así sobre la supervivencia de ejemplares de menor edad o sobre la reproducción.

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RELACIÓN DE ARTÍCULOS

1. Delgado, S., Nere Zorrozua, N. & Arizaga, J. (2021). No evidence of habitat effect on clutch size, egg quality and hatching success of the Yellow-legged Gull *Larus michahellis* at a micro-spatial scale. *Marine Ornithology,* 49 (2), 241-246.

2. Delgado, S., Aldalur, A., Herrero, A. & Arizaga, J. (2020). No evidence supporting sex-dependent differential movements and survival in Yellow-Legged gulls. *Ardea,* 108(2), 183-190. DOI: 10.5253/arde.v108i2.a4

3. Delgado, S., Herrero, A., Aldalur, A. & Arizaga, J. (2021). High philopatry rates of Yellow-legged gulls in the southeastern part of the Bay of Biscay. *Avian Research,* 12(36). DOI: 10.1186/s40657-021-00271-8

4. 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(3), 238-246. DOI: 10.1002/1438-390X.12083

5. Delgado, S., Arizaga, J., Aldalur, A., Herrero, A., & Tavecchia, G. (2021). Modelling population trends of an opportunistic seabird species: the case of Yellow legged-gull in the Bay of Biscay. *Pendiente de publicación..*

ARTÍCULO 1

NO EVIDENCE OF HABITAT EFFECT ON CLUTCH SIZE, EGG QUALITY AND HATCHING SUCCESS OF THE YELLOW-LEGGED GULL *LARUS MICHAHELLIS* AT A MICRO-SPATIAL SCALE.

Abstract

In colonial seabirds, the nest substrate that is available and, in particular, the vegetation cover around the nest, are important environmental factors that drive an individual's nesting selection process and, ultimately, reproductive performance. Using data collected during three consecutive years in a Yellow-legged Gull *Larus michahellis* colony from the Bay of Biscay, Spain, we tested whether clutch size, egg volume, and hatching success covaried with the proportion of vegetation cover around nests. We found no effect of vegetation cover on breeding performance. Laying date showed a positive effect on egg volume and a negative effect on hatching success and the number of hatched eggs. Egg volume tended to be smaller in 2020 compared to 2018 and 2019, and hatching success decreased through the sampling period, with the lowest hatching success occurring in 2020. Our findings agree with a previous study in which vegetation had no or unclear effects on breeding performance in Yellow-legged Gulls; however, they contradict other seabird studies that found a positive correlation between the two variables. The role of vegetation on breeding performance could vary not only between gull species, but also geographically, with ecological drivers such as intra- and interspecific interactions and climate playing key roles in observed differences. Finally, the use of egg volume and hatching success as proxies for breeding output could be used for the long-term monitoring of the relationship between breeding performance and factors such as landfill management, fishing, or climate change in the Yellow-legged Gull and other gull species.

Delgado, S., Nere Zorrozua, N. & Arizaga, J. (2021). No evidence of habitat effect on clutch size, egg quality and hatching success of Yellow-Legged Gull *Larus michahellis* at a micro-spatial scale. *Marine Ornithology,* 49(X), 241-246. DOI

Reproductive performance is a capital process that strongly affects the demographics of bird populations. Therefore, understanding the factors that influence breeding performance is crucial to assess their impact at the population level (Coulson 2001). Gulls are colonial seabirds and breeding in dense colonies can potentially result in very high competition for the best nesting sites (Gaston 2004). In general, birds that breed in the center and/or the preferred places within the colony breed with greater success than birds occupying a colony´s margins; also, at a disadvantage are birds that are exposed to more predators or birds that are less protected against adverse weather conditions (Gaston 2004).

In this context, the availability of nest substrate is an important environmental factor driving the nesting selection process and, ultimately, reproductive performance (Bongiorno 1970, Skorka *et al.* 2005). In a Mediterranean colony of Yellow-legged Gulls *Larus michahellis*, individuals were observed to first occupy zones with a higher percentage of vegetation (relative to zones that were bare; Bosch & Sol 1998), a behaviour also observed in other gulls (Yorio *et al.* 1995, Ellis 2005). Moreover, gull density was higher in these vegetated zones (Bosch *et al.* 1994), although breeding success was similar between habitats (Bosch & Sol 1998). Clutch size differed between these habitat types (increasing in zones with more vegetation), although this difference was attributed to a possible late-seasonal laying (Bosch & Sol 1998). Overall, it appears that dense vegetation in gull colonies improves breeding output by increasing rates of hatching and fledging success (Montevecchi 1978, Parsons 1982, Craik 1999, Garcia-Borboroglu & Yorio 2004, Kim & Monaghan 2005a). This improvement occurs because vegetated areas offer higher protection against bad weather (Kim & Monagham 2005b) and/or predation (Burger & Shisler 1978). Consequently, nesting density can increase in areas in or close to dense vegetation (Becker & Erdelen 1986). Over time, however, high nest density could produce an impoverishment of vegetation and reduce long-term breeding success (Vidal *et al.* 2000, Ellis *et al.* 2005). It is less clear, however, whether breeding in areas with dense vegetation has any significant relationship to other parameters, such as egg size (Bosch & Sol 1998, but see Becker & Erdelen 1986).

Our aim in this study was to evaluate the effect of nesting habitat, measured at a micro-scale level, on the breeding performance of a Yellow-legged Gull colony. To accomplish this goal, we analysed data collected during three consecutive years from a colony of Yellow-legged Gulls located in the Bay of Biscay, Spain. We hypothesised that a higher proportion of vegetation, compared to bare soil (rock substrate), would be associated with greater breeding output (clutch size, egg volume, hatching success) in the colony under study.

STUDY AREA AND METHODS

The avian model

The Yellow-legged Gull is the most abundant gull of the southwestern Palaearctic region, with a population of *ca*. 150 000–200 000 breeding pairs (Olsen & Larson 2004). In Spain, there are more than 80 000 pairs (Molina 2009), and a significant fraction breeds in the Bay of Biscay along the coast from Galicia (west) to the Basque region (east). The population that breeds along this coast is resident (Munilla, 1997) and has a very restricted flow among colonies; hence, it likely constitutes a meta-population (SD, unpubl. Data) and is considered to belong to a subspecies, *L. m. lusitanius*, breeding from the Basque coast to southwest Iberia (Olsen & Larson 2004). In the Basque region, where several studies have been carried out, the population has increased 146% since 2000 (Arizaga *et al*. 2009) due to foraging opportunities provided by landfills (Arizaga *et al*. 2011) and discards from the fisheries industry (Foster *et al*. 2017). More recently, the population has been stable or may have declined (Juez *et al*. 2015), principally due to the closure of landfills and increased controls on fisheries discards (Egunez *et al*. 2018, Zorrozua *et al*. 2020a).

Study area and data collection

The data used in this study were obtained from the colony of Ulia (43°20’N, 01°57’W), province of Gipuzkoa, Basque Country, Spain. This colony totals *ca*. 660 pairs, which makes it the main colony in Gipuzkoa and one of the most important Yellow-legged Gull colonies in the Bay of Biscay. The colony is located on coastal cliffs, with nests spread across a mixture of patches of bare soil (sandstone rock) and vegetation. The vegetation within the colony is mostly composed of herbaceous species (Eagle Fern *Pteridium aquilinum,* Cornish Heath *Erica vagans,* Coastal Spleenwort *Asplenium maritimum*) mixed with small stands of American Pokeweed *Phytolacca americana* and Saltbush *Baccharis halimiolia*, the latter two of which are invasive American plants. Plants within the colony usually achieve a height < 1 m.

This study extended over three breeding years, 2018–2020. The breeding season lasted from mid-April (when the eggs are laid) to the end of June (when most chicks are about to fledge; Mínguez 1988, Galarza 2008, Arizaga *et al*. 2012). The colony was surveyed every two days from 20 April to 25 June. Visits were cancelled/postponed in cases of adverse weather (e.g., rain or very strong winds). Once in the colony, we looked for nests during the entire laying period and marked each nest with a stake displaying a small, numbered flag (40 cm high). Nests were selected randomly within zones that were relatively easy to survey (nests in very steep, inaccessible cliffs could not be monitored). Eggs were marked using a felt-tipped pen according to their laying order—both with letters (A, B, and C) and their laying day (assuming a maximum laying interval of 48 h). If, in each visit, a nest had two or more new eggs, eggs were marked as AB, BC, or ABC because we were unable to determine laying order. At the end of the incubating period, we determined—for each egg—whether it hatched, failed (a non-broken, non-hatched egg), or other (that is, the egg was broken or disappeared due to predation or accident).

Once identified, the following variables were measured: (1) laying date (for the first egg); (2) laying sequence for each egg (with a letter A, B, C); (3)

clutch size; (4) egg size (length [L], width [W], measured with a digital caliper with ±0.01 mm accuracy); and (5) hatching success. Thereafter, egg volume was calculated as: $0.000476 \times L \times W^2$ (Bolton *et al.* 1992).

Apart from breeding parameters, the type of substratum around each nest was also recorded. Specifically, we assessed the proportion of the total area within a 1-m radius around each nest that was covered by bare soil (rock) or vegetation.

Statistical modelling

First, we removed five nests (*n* = 1 in 2018, *n* = 2 in 2019, *n* = 2 in 2020) from the data set that had a clutch size of one egg, which were otherwise very rare within the colony.

To assess whether the habitat type at a micro-scale level affected the breeding parameters listed above, we built Generalized Linear Models (GLM), with year as a factor and the laying date and proportion of vegetation cover as covariates. The type of link-function and the errors' distribution used in each GLM varied in relation to the nature of each object variable. Thus, we used a linear-link function with normal errors for the following dependent variables: laying date, hatching date, and mean and maximum egg volume. By contrast, we used a logit-link function with negative binomial errors' distribution for hatching success (0 = no hatching, 1 = one or more eggs hatched). The analyses were carried out with the software R (R Core Team 2014) and the package "lmerTest" (Kuznetsova *et al.* 2017); post-hoc tests (Tukey test and Chi square test) were applied.

RESULTS

We sampled 267 nests (see Table 2), containing 752 eggs in total, all of which were measured. Mean clutch size (± standard deviation, SD) was 2.81 ± 0.38 eggs/nest, and the mean and maximum egg volume per nest was, respectively, 72.1 ± 5.48 cm^3 and 75.6 ± 6.25 cm^3 (Table 2). Overall, 233 (87.3%) nests had at least one hatched egg (Table 2), and of the 752 eggs laid, 486 (64.6%) hatched successfully; 171 eggs (22.6%) did not hatch, and 95 eggs (12.6) were broken or disappeared (Table 2). The mean percentage of vegetation cover around nests was 71.10% ± 34.18% (Table 2).

Table 2. Breeding parameters of a Yellow-legged Gull Larus michahellis colony in the Bay of Biscay, Spain, 2018-2020[a].

	2018	2019	2020	Total - mean values
Sample size (nests)	172	50	50	272
EGGS				
Sample size (eggs)	494	134	129	757
Clutch size (eggs)	2.88 ± 0.32	2.75 ± 0.43	2.64 ± 0.48	2.81 ± 0.38
Mean volume (cm^3)	73.05 ± 5.13	71.037 ± 6.56	70.32 ± 4.95	72.19 ± 5.48
Max. volume (cm^3)	76.21 ± 5.82	75.39 ± 7.92	73.60 ± 5.57	75.60 ± 6.25
HATCHING				
Hatching success: % (count)	89.5% (153)	93.8% (45)	64.6% (31)	87.3% (233)
Hatched eggs: % (count)	66.5% (326)	74.4% (99)	47.6% (64)	64.6% (486)
Hatched eggs	326	99	64	486
Hatched egg by nest	1.89	1.96	1.26	1.78
Non-hatched eggs: % (count)	20.0% (99)	25.6% (35)	32.6% (39)	22.7% (171)
Broken/Disappeared eggs: % (count)	13.5% (69)	0.0% (0)	19.8% (26)	12.6% (95)
VEGETATION				
Vegetation cover (%)	70.09 ± 35.67	73.12 ± 33.65	72.70 ± 29.44	71.10 ± 34.18

[a] *Mean values have been provided ± standard deviation; units shown in parentheses*

Clutch size, egg volume, and hatching did not vary in relation to vegetation cover (Table 3). A later laying date showed a positive effect on egg volume, but a negative effect on hatching success and hatched eggs (Table 2). Finally, egg volume tended to be smaller in 2020 compared to 2018 and 2019, and hatching success decreased through the sampling period, with the lowest values occurring in 2020 (Table 3).

Table 3. Beta-parameter estimates obtained from a set of models used to determine the effect of vegetation cover and other parameters on breeding performance of a Yellow-legged Gull Larus michahellis population from the Bay of Biscay, Spain.

	VC	DT	YR: 2019	YR: 2020
Clutch size	+0.00 ± 0.00	-0.00 ± 0.01	-0.06 ± 0.12	-0.09 ± 0.10
R^2 = 0.06	(0.893)	(0.837)	(0.607)	(0.374)
Mean volume	+0.01 ± 0.01	+0.21 ± 0.08	-0.26 ± 1.08	-2.04 ± 0.91
R^2 = 0.07	(0.483)	(0.006)	(0.813)	(0.025)
Max. volume	+0.01 ± 0.01	+0.26 ± 0.09	+1.28 ± 1.25	-1.80 ± 1.04
R^2 = 0.06	(0.667)	**(0.004)**	(0.304)	(0.086)
Hatching success	+0.00 ± 0.00	-0.23 ± 0.90	-1.80 ± 0.90	-2.71 ± 0.54
R^2 = 0.20	(0.723)	**(<0.001)**	(0.045)	**(<0.001)**
Hatched eggs	-0.00 ± 0.00	-0.04 ± 0.01	-0.20 ± 0.14	-0.47 ± 0.14
	(0.452)	**(0.001)**	(0.149)	**(<0.001)**

a Independent variables: VC, vegetation cover; DT, laying date; YR, year.

b Beta parameter estimates are values ± standard error; P-values are in parentheses; significant values (P <0.05) are in bold font).

DISCUSSION

This study analysed, for the first time, the influence of vegetation cover on the breeding performance of a Yellow-legged Gull population of the *L. m. lusitanius* subspecies. It adds to the findings of other studies on the effect of vegetation cover on gulls' breeding performance (see Ellis 2005), including in the Yellow-legged Gull (Bosch & Sol 1998).

We found no evidence of an effect of vegetation cover on any of the analysed breeding parameters (clutch size, egg volume, hatching). This result agrees with a previous study in which vegetation had no or unclear effects on breeding performance in another Yellow-legged Gull colony (Bosch & Sol 1998), but it is in contrast to studies in which dense vegetation was found to improve hatching success (Bosh & Sol 1998, Rodway & Regehr 1999). The role of vegetation on breeding performance could vary not only between species, but also geographically, with ecological drivers such as intra- and interspecific interactions and local climate playing key roles in observed differences. In theory, the amount of vegetation cover could be more critical in denser colonies, in colonies with higher predation pressure, in colonies that are more susceptible to disturbance, or in colonies that are subject to harsher climatic conditions (such as high temperatures and insolation; With & Webb 1993, Miyazaki 1996, Kim & Monaghan 2005b). None of these circumstances are applicable to our survey colony.

The effect of vegetation through phases of the breeding period (i.e., during the chicks rearing period) is unclear. Presumably, chicks that are surrounded by more vegetation would benefit from greater protection against bad weather and predation (Saliva & Burger 1989, Kim & Monaghan 2005b), as well as attacks from conspecifics (Burger 1977, Krause & Ruxton 2002, Kim & Monaghan 2005b). Overall, our breeding colony is characterized by vegetation cover that grows in parallel with the breeding season—it is relatively small during the first weeks of the breeding period (i.e., egg laying, incubation) and grows larger as the breeding period progresses. In this scenario, it is possible that in other, similar colonies, vegetation may have a null or marginal (undetectable) effect over the breeding period.

The mean values of the analysed breeding parameters show a clutch size similar in value to other large gulls, in which size varied from 2.5 to 2.9 (Monaghan *et al.* 1991, Bosh & Sol 1998, Baaloudj *et al.* 2014, Hammouda *et al.* 2014). Mean egg volume for each nest was also similar in this study to some other gull colonies (Monaghan *et al.* 1991), although it was lower than in others (Isenmann 1976, Baaloudj *et al.* 2014, Hammouda *et al.* 2014). Hatching success in this colony was greater than in several other colonies (Isenmann 1976, Bosh &

Sol 1998, Moulaï *et al.* 2006, Baaloudj *et al.* 2014), yet it was similar to hatching success in northern European populations (Oro *et al.* 1995; Bosch *et al.* 2000, Duhem *et al.* 2002).

In general, seabird colonies show a decrease in hatching success and overall breeding performance with a later laying date (Davis 1975, Bosman 2014, Galarza & Arizaga, 2014), an outcome that was also observed in our study. However, unexpectedly, we observed that a later laying date had a positive effect on egg volume. Egg volume is correlated with eventual greater chick mass and body condition, increasing chick survival (Bolton 1991, Kubelka *et al.* 2020). For late lay nests or replacement eggs, the egg-size is typically lower in laying producing a small-volume egg (Birkhead & Nettleship 1982).

We detected significant yearly variation in egg volume and hatching. More specifically, all parameters tended to decrease during the three study years and were lowest in 2020. From a meteorological standpoint, the spring of 2020 was not colder or more rainy than previous springs (Table 4). Therefore, weather may not be a direct factor explaining these inter-annual variations, but it could affect changes in diet and food access. Thus, our colony, and the entire Yellow-legged Gull population from the southeastern Bay of Biscay, may be experiencing a change in its trophic ecology because of the effects of landfill closures and changes in the management of fish discards (Arizaga *et al.* 2018, Zorrozua *et al.* 2020a, 2020b). Access to food from open-air landfills within the study region has been decreasing; the only site open during 2018–2020 was at Zaluaga (43°22'N, 01°34'W), 22 km away in France. The processing and waste treatment on this landfill changed in October 2019, decreasing food availability for opportunistic species.

It is noteworthy that the number of chicks ringed in the Ulia colony in 2019 and 2020 (*n* = 43 and 76, respectively; JA unpubl. data) was remarkably lower than in 2018 (*n* = 170), even though we invested a similar ringing effort. These data indicate that egg volume and/or hatching success may be used as a proxy for the colony’s breeding output.

Despite finding no effect of vegetation cover on gulls' breeding performance, we uncovered interesting annual variation in parameters such as egg volume, hatching success, and the number of hatched eggs, which are worth studying in the future.

Table 4. Meteorological parameters obtained from the Euskalmet Santa Clara station, Spain, 2018-2020.

	2018	2019	2020
April			
Precipitation (l/m^2)	225.5	115.7	106.7
Mean temperature (C°)	14.4	12.1	14.5
Minimum mean daily temperature (C°)	11.1	10.5	12.0
May			
Precipitation (l/m^2)	114.3	164	104.2
Mean temperature (C°)	14.5	15.7	16.53
Minimum mean daily temperature (C°)	12.0	13.5	13.8
June			
Precipitation (l/m^2)	94.2	67.1	108.7
Mean temperature (C°)	18.3	17.7	16.8
Minimum mean daily temperature (C°)	15.83	14.4	14.48

Monitoring long-term changes in these parameters could provide a pool of variables that could be used to evaluate the relationship between breeding performance and factors such as landfill management, fishing, or climate change (Belant *et al.* 1993, Oro *et al.* 1995, Real *et al.* 2017). Other breeding parameters, such as productivity, are much more difficult to measure because gulls can occupy colonies having an intricate topography, making measurement problematic.

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ARTÍCULO 2

No evidence supporting sex-dependent differential movements or survival in Yellow-legged gulls.

Abstract

Important biological parameters, such as movement and survival, can differ substantially between sexes, so sex can be one of the factors driving demographic patterns of a population. Sex-dependent spatial segregation at the foraging grounds can evolve as a mechanism to reduce competition for resources between sexes, among other causes. Investigating such segregation, especially in scenarios of decreasing food availability, can contribute to understand how birds adapt to a fast-changing world. The aim of the present study was to determine whether year-round movements and survival varied between sexes within a resident Yellow-legged gull *Larus michahellis* population. To this end we used live encounter data on previously colour-ringed Yellow-legged Gulls in northern Spain. We used Generalized Linear Models with distance to the colony of origin and multi-state models to estimate both survival and year-round movements, which did not provide evidence supporting an effect of sex on movement and survival. In contrast, distances travelled from the natal colony varied between seasons and results supported a higher probability to move to places situated at more than 100 km from natal sites than to return to places closer to natal sites. Juveniles showed lower survival rates (0.55) than older birds (0.90). Future research with GPS data might help to reveal, if existing, the occurrence of local spatial segregation between sexes that otherwise remains undetected.

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Biological parameters, such as movement and survival, can differ substantially between sexes, so that sex can be one of the main factors driving the demographic patterns of a population (Newton 2013). When breeding, individual birds in a colony generally share the same foraging areas, although minor differences in foraging strategies are seen in some gull (Laridae) populations (Camphuysen *et al.* 2015). Once breeding ends, spatial segregation between sexes can increase (Cristol *et al.* 1999, Nebel 2005, Bosman *et al.* 2011, Lundblad & Conway 2020). There are several ultimate factors explaining this sexual segregation and these can vary between species. Size-dimorphic competition at an intra-specific level might force one sex (usually the smaller) to look for food elsewhere (e.g. Cristol *et al.* 1999, Catry *et al.* 2005, but see Bosman *et al.* 2011). Sex-dependent geographic segregation can evolve therefore as a mechanism to reduce competition for resources (mostly food; Wearmouth & Sims 2008). When both sexes feed on similar resources, spatial overlap between the sexes may be reduced when food is a limiting factor (Cristol *et al.* 1999, Catry *et al.* 2005). When sexes share a space, it indicates either the absence of competition behaviour or of food limitations. Competition can be reduced by exploiting different trophic resources or different foraging habitats, even at a very small spatial scale (Catry *et al.* 2005, Forero *et al.* 2005, Alves *et al.* 2013, Kazama *et al.* 2018). Spatial segregation between sexes, however, can be explained by alternative hypotheses (Nebel & Ydenberg 2005, Lundblad & Conway 2020), including (1) arrival time, when males stay closer to the colony during the non-breeding season to enable them to occupy a good site within the colony as soon as the breeding season approaches, (2) thermal tolerance, where the larger sex is able to winter in colder places, (3) fasting endurance, where the larger sex has greater capacity to tolerate episodes of food shortage (4) resource partitioning, where sexual spatial segregation exists when sexes vary in their morphology, which is optimized to exploit a feeding resource that is spatially partitioned, or (5) predator escape performance, where the sexes can reduced their risk of predation in different areas due to spatially segregated predators.

Sexual segregation in seabirds (suborder Lari), is receiving increasing attention among scientists owing to the effects that different behaviours could have at a population level (Camphuysen *et al.* 2015, Kazama *et al.* 2018). For instance, studies with Lesser Black-backed Gulls *Larus fuscus* have revealed that during the breeding season females forage closer to the colony and perform shorter foraging trips than males (Camphuysen *et al.* 2015), highlighting some degree of spatial segregation between sexes during the breeding season.

The Yellow-legged Gull *Larus michahellis* is one of the most common large gulls in the southwestern Palaearctic. In Spain, the latest census assessed a population of c. 125,000 pairs (Molina 2009). Recent landfill closures led to food shortages for some mainland colonies, not only causing a population decline but also changes in movement patterns of local gulls (Arizaga *et al.* 2014b, Egunez *et al.* 2017). The population breeding along the coast of northern Iberia is mostly resident, with c. 70% of the individuals remaining within 50 km from their natal colonies year-round (Egunez *et al.* 2017). However, long-distance movements, up to more than 1000 km, also happen, especially in first-year birds (Arizaga *et al.* 2010, 2015a). If there is spatial segregation between the sexes, we could test evidence supporting different hypotheses. If segregation is detected, one sex is likely to move further. Under the competition and arrival hypotheses, females should winter in more distant regions, because they are smaller and do not need to arrive at the colonies as soon as winter ends. Other hypotheses, however, would not fit with such findings when following the scenarios of a favourable food distribution and availability close to the colony where females, which wintering close by, have a higher breeding success (Lundblad & Conway 2020). Additionally, survival analyses could allow us to find evidence in favour of some previously mentioned hypotheses.

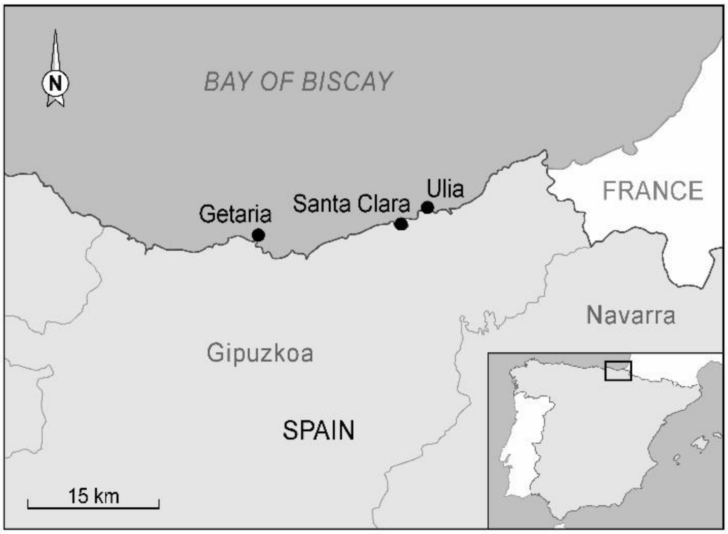
The aim of the present study was to determine whether movement and survival varied between sexes within a resident Yellow-legged Gull population.

MATERIAL AND METHODS

Sampling area and data collection

The data used in this study were obtained from birds marked in three Yellow-legged Gull colonies situated along the coast of Gipuzkoa, northern Spain, in the south-eastern part of the Bay of Biscay (Figure 2); from east to west: Ulia (43°33'N, 01°95'W; with 660 breeding pairs in 2017), Santa Clara (43°32'N, 1°99'W; 100pairs) and Getaria (43°31'N, 2°20'W; 165 pairs). Adult Yellow-legged Gulls start to occupy their breeding sites in the colony around February and the laying period is in April. With an incubation period of c. 26 days, the first hatchlings appear from mid-May onwards. The parents feed their offspring within the colony until the first half of July. During the breeding season of 2009,2010, 2011 and 2013, 155 chicks were marked when they were close to fledging, with both a metal ring (Aranzadi ringing scheme) and a PVC ring with an alphanumeric code, allowing the identification of each individual bird from a distance (Fernández *et al.* 2017).

Figure 2. Locations of the three study colonies in the Gipuzkoa province, Spain.



Dorsal feathers were taken from each bird in order to determine its sex using molecular techniques. Feathers kept in paper envelopes until they were sent to the laboratory (Department of Zoology and Ecology, University of Navarra). Gulls were sexed using DNA-analyses (Griffiths *et al*. 1998). For each bird, a sample from the base of the rachis from one to two feathers was taken and stored in a 1.5 ml vial, filled with 99% ethanol. The DNA fragment related to the CHD-protein, present in both Z and W chromosomes, was amplified by means of a Polymerase Chain Reaction (PCR) technique. PCR fragments were separated by electrophoresis on a 2.5% agarose gel: a single band of DNA on the gel showed that a bird was male (corresponding to CHD-Z gene), while two bands were present in females (corresponding to both CHD-Z and CHD-W). From the 155 gulls sampled, 80 were males and 75 females.

Once these birds left their natal colonies, they were observed by birdwatchers and our own research team, this data was used to estimate survival as well as distance from the natal colony for the two sexes. Records of re-sighted birds were taken up to 31 December 2017. Overall, from 155 ringed chicks, 84 (54.2%) were seen at least once after they left their colonies. In total, these 84 gulls provided 671 sighting occasions.

Data analysis

Our main goal was to determine whether distances travelled from the colonies and survival varied between the two sexes. With that aim, we divided the year into four seasons (Egunez *et al*. 2017): Jan–Mar (pre-breeding), Apr–Jun (breeding), Jul–Sep (post-breeding), Oct–Dec (wintering). The age of each gull was classified as juvenile (here, from the hatching year to June of the next year), immature (from June of their second year of life to June of their fourth year of life) or adult (older birds, when gulls reach sexual maturity).

First, we conducted Generalized Linear Mixed Models (GLMM) to test whether distances travelled varied between the sexes. We $\log_{10}$-transformed distance to the colony of origin to improve normality. The following explanatory variables were included as factors: season (using the four seasons considered above), age class and sex, and the 2- and 3-way interaction between these three factors; bird identity was included as a random factor. Age was included in the model since the movement patterns in this population vary between age classes (Egunez *et al.* 2017). For each individual bird, we only considered the maximum distance achieved per season and age category in order to remove pseudo-replicates. We used an identity link function with a Gaussian error structure. The analysis included 336 sightings of 84 individuals (for details see Table 5) and was performed in R v. 3.6.3 (R Core Team 2020) using the packages 'lme4' (Bates *et al.* 2015) and 'lmerTest' (Kuznetsova *et al.* 2017). We started with a global model with 3-way interactions and performed a model selection procedure using the 'dredge' function of the package 'MuMIn' (Barton 2014). Models with AICs differing less than 2 were considered to fit the data equally well (Burnham & Anderson 1998).

Table 5. Mean (± quartile ranges) distance to the natal colony and sample size of colour-ringed Yellow-legged Gulls seen alive in the Bay of Biscay, in relation to their age class and season.

	Juvenile	Immature	Adult
Male			
Jan–Mar	26.5 (31.6–18.3; 12)	34.9 (35.1–30.5; 19)	75.1 (30.5–15.7; 11)
Apr–Jun	32.2 (30.5–30.5; 14)	12.3 (26.52 –3.32; 10)	5.8 (3.3 –1.7; 7)
Jul–Sep	60.6 (53.0–30.5; 19)	69.1 (30.5 –30.5; 19)	81.6 (65.0 –3.3; 10)
Oct–Dec	41.0 (38.6–1.7; 22)	32.9 (30.5 –14.7; 20)	78.6 (50.1 –13.8; 12)
Female			
Jan–Mar	75.7 (56.6–16.9; 14)	32.5 (30.5–30.5; 17)	17.7 (30.5 –3.4; 11)
Apr–Jun	59.5 (30.5–16.8; 14)	72.5 (8.5–2.4; 7)	14.6 (27.6 –1.7; 4)
Jul–Sep	130.8 (114.3–30.5; 19)	42.2 (64.3–30.5; 13)	71.7 (47.7 –30.5; 11)
Oct–Dec	101.0 (69.4–26.4; 21)	113.5 (112.4–30.5; 17)	74.5 (39.1 –13.4; 13)

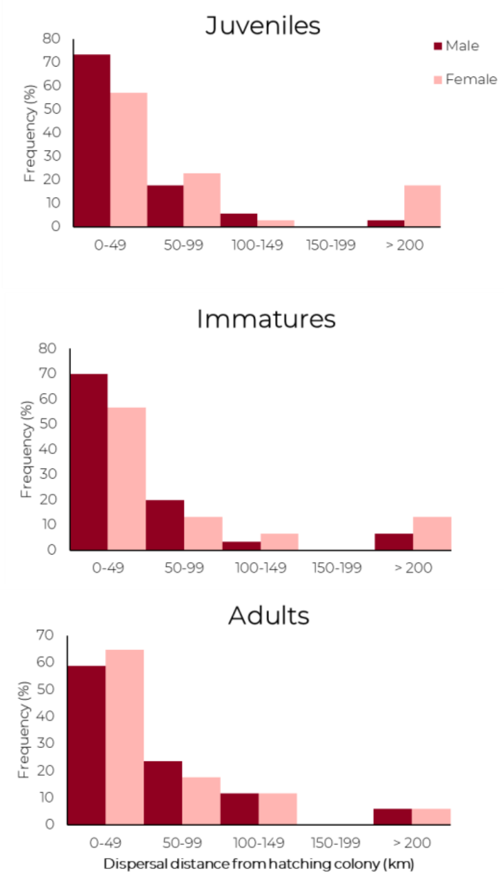
Second, we used capture-mark-recapture (CMR) models in MARK (White & Burnham 1999) in order to estimate whether survival and presence in the breeding area (<100 km) varied in relation to sex and time of the year. We used multi-state models with live encounters, which allow the estimation of survival (φ: probability that a bird survives from t to $t + 1$), recapture probability (p: probability that a bird that survives from t to $t + 1$ is seen in $t + 1$) and movement probability (ψ: probability that a bird in site 1 moves to site 2). Since such models need relatively long-time intervals between capture sessions, we only considered two seasons per year in this case: the breeding and the winter period, i.e. data collected from April to June and from October to December Overall, the design matrix had 18 columns (two seasons per year during a period of nine years, 2009 to 2017), 155 rows (individuals) with two distance categories (sightings more or less than 100 km from the natal colonies) and the two sexes. Before testing the fit of the data to candidate models, we tested if the data met the CMR assumptions. For this we used the global goodness-of-fit test performed in U-CARE (Choquet *et al.* 2009) and found no significant effects for either sex (females: $c_2 = 27.61$, $df = 22$, $P = 0.189$; males: $c_2 = 14.91$, $df = 21$, $P = 0.828$) was found, indicating that our data fitted the CMR assumptions regarding trap-dependence (encounter probability biased) and occurrence of transients (emigrant individuals marked).

We constructed a number of candidate models that were then subjected to a selection procedure modified for small-sample sizes (AICc), taking into account both the number of parameters of the model and its deviance (Burnham & Anderson 1998). We started by building models assuming either constant or time dependence effects on φ, ψ and p. After that we also ran all the possible combinations that included an effect of sex, period and age on φ and ψ for the two distance categories, while for p, we only considered models that also included sex and distance category. Overall, we built 101 candidate models. Models that differed less than 2 AICc units compared to the model with the lowest AICc were averaged.

RESULTS

Overall, the mean maximum distance achieved by each individual bird was 60 ± 6 km (±95% CI). Mean distance travelled from the natal colony was less than 50 km in all age classes (Figure 2). According to our global GLMM, the distance was affected by age and season, but not by sex (sex: $F_{1,82} = 0.78$, $P = 0.38$, age: $F_{1,82} = 13.13$, $P < 0.001$, season: $F_{1,230} = 21.06$, $P < 0.001$; interactions: all $P > 0.05$, except age × season: $F_{1,230} = 4.80$, $P<0.001$).

Figure 2. Distribution of distances to the natal colony between sex classes in relation to age categories: juveniles (from hatching year to June of the next year), immatures (from June of their second year of life to June of their fourth year of life)



Model selection procedure provided two models that fitted to the data equally well: a first model, considering an effect of age, season, and the age-season interaction on distance, and a second model, which also included an effect of sex. These models were averaged, and the result only provided a significant effect of season, and the age season interaction on distance, indicating a weak and non-statistically significant effect of sex on this variable (Table 6). On average, distances travelled tended to decrease during the breeding period (Apr–Jun), with a significant interaction between age class and season, due to juveniles being observed at larger distances from the colony in the breeding season than the other age categories (Table 6).

Table 6. Parameter estimates obtained from model-averaged Generalized Linear Mixed Models testing for the effect of sex, age and season on distance to the natal colony of Yellow-legged Gulls in the Bay of Biscay. Reference values: Sex: female = 0; Age

	Beta	SE (*Beta*)	*P*
Parameters			
Sex: male	–0.07	0.17	0.655
Age: juvenile	+0.25	0.25	0.312
Age: immature	+0.35	0.25	0.164
Season: Jan–Mar	–0.45	0.28	0.106
Season: Apr–Jun	–2.13	0.36	<0.001
Season: Jul–Sep	–0.16	0.29	0.571
Age-season interaction			
Juvenile×Jan–Mar	+0.19	0.37	0.614
Immature×Jan–Mar	+0.11	0.35	0.758
Juvenile×Apr–Jun	+1.95	0.43	<0001
Immature×Apr–Jun	+0.44	0.45	0.334
Juvenile×Jul–Sep	+0.33	0.36	0.359
Immature×Jul–Sep	+0.09	0.37	0.800

CMR analyses provided four best-ranking models that fitted the data equally well (Table 7). These models were very similar, and included age-dependence on survival (with one value for the juvenile birds and one for immature plus older birds), an effect of distance and season on ψ, and an effect of distance and age on p.

Table 7. Ranking of the best ten models used for testing effect of age, season and distance travelled on survival (φ*), displacement (*ψ*) and re-sighting (p). The reference model, with constant *φ*, *ψ* and p, is shown at the bottom. ΔAICc indicates the difference with the model with the lowest AICc. '2ages' indicates a difference between juveniles and older birds, while '3ages' indicates variation among the three age classes (juvenile, immature and adult birds). 'season' stands for a difference between the winter and summer period. 'dist' means that a parameter varies between gulls close (<100 km) or far away (>100km) from the colony. 'Sex' indicates a difference between males and females.*

Model	AICc	ΔAICc	AICc weight	*Np*	Deviance
φ_{2ages}, $\psi_{season+dist}$, $p_{zone+2ages}$	1164.31	0.00	0.33	9	589.87
φ_{3ages}, $\psi_{season+dist}$, $p_{zone+2ages}$	1165.01	0.70	0.23	10	588.45
φ_{2ages}, $\psi_{season+dist+sex}$, $p_{zone+2ages}$	1165.98	1.67	0.14	13	582.96
φ_{3ages}, $\psi_{season+dist+sex}$, $p_{zone+2ages}$	1166.19	1.88	0.13	14	581.00
φ_{3ages}, $\psi_{season+dist}$, p_{zone}	1168.11	3.80	0.05	8	595.78
φ_{2ages}, $\psi_{season+dist}$, p_{zone}	1168.17	3.86	0.05	7	597.94
φ_{3ages}, $\psi_{season+dist+sex}$, p_{zone}	1169.12	4.81	0.03	12	588.27
φ_{2ages}, $\psi_{season+dist+sex}$, p_{zone}	1169.65	5.34	0.02	11	590.95
$\varphi_{2ages+season+dist}$, ψ, p_{zone}	1176.18	11.87	0.00	6	608.04
φ_{3ages}, ψ_{3ages}, p_{dist}	1176.21	11.89	0.00	8	603.87
φ, ψ, p	1228.14	63.82	0.00	3	666.17

The averaged model provided the following parameter estimates. For φ (±SE), adult birds had a similar apparent survival (0.90 ± 0.04) to immatures (0.87 ± 0.03), while that of juveniles was significantly lower (0.55 ± 0.05). However, the overlap between the two oldest categories was high and, therefore, only the juvenile fraction had a significantly lower survival value. In the case of ψ, the four best models included an effect of season (pre-breeding and breeding period vs. post-breeding and winter period), distance (less or more that 100 km from the breeding colony) and sex.

Looking at Table 8, however, the effect of sex was not significant, with much overlap in parameter estimates between sexes. Finally, for p, we obtained higher values when gulls were within a radius of less than 100 km from their natal sites. Within that radius, the juvenile birds had a probability of detection that was lower (0.76 ± 0.16) than the older birds (0.99 ± 0.01). At distances >100 km from their natal sites, the probability of detection for juveniles increased (0.26 ± 0.08) when compared to older birds (0.09 ± 0.02).

Table 8. Movement probabilities (mean ± SE) obtained after model-averaging (best four models from Table 7) of a Yellow-legged Gull population, in relation to season, distance and sex. As example, a value of 0.5 means that there is a probability of 50% to move from site A to site B.

	Male	Female
Breeding to winter season		
Move close (<100 km)	0.56 ± 0.070	0.57 ± 0.075
Move away (>100 km)	0.32 ± 0.047	0.25 ± 0.053
Winter to breeding season		
Move close (<100 km)	0.67 ± 0.059	0.64 ± 0.059
Move away (>100 km)	0.12 ± 0.036	0.15 ± 0.042

DISCUSSION

Using data from a Yellow-legged Gull population breeding in the North of Spain, we did not find evidence supporting sex-dependent variation in either distance travelled from the natal colony or survival. Our results suggest that there is limited or no competition for feeding resources, perhaps because there is enough food available (Zorrozua *et al.* 2020a) or because females feed on different food items than males (Camphuysen *et al.* 2015). Even though a number of landfills within the region have been closed, food ecology studies suggest that there is still enough food available, in part because some landfill sites remained open during the study period, and also because the gulls are able to shift to other resources (Zorrozua *et al.* 2020b). Furthermore, the lack of spatial segregation between the sexes during the non-breeding season suggests that the Yellow-legged Gull population in the Bay of Biscay may not be influenced by natural and sexual selection processes that promote sexual segregation in winter (Cristol *et al.* 1999, Catry *et al.* 2005, Nebel & Ydenberg 2005, Lundblad & Conway 2020).

Our findings are also compatible with some other explanations. For example, females might tend to disperse more (longest distance records were detected in females; Table 5), but the fraction of birds doing so is small. Furthermore, the relatively small sample size (155 birds in total), makes it difficult to detect small differences due to limited statistical power. However, such small differences may have marginal biological meaning. Additionally, it can be also mentioned that the sighting effort at distances >100 km was notably smaller than at distances <100 km, and this may cause a bias in the number of long-distance dispersers (reducing sample size), although there was no difference in the proportion of male-females found outside this radius of 100 km. Thus, the uneven observation effort across the distribution range of these birds should not have a significant impact on the fact that we did not find differences in distances travelled between the sexes.

In the near future other landfills will be closed (e.g. Arizaga *et al.* 2014b, Steigerwald *et al.* 2015) and, at the same time, fish discards will be reduced in northern Spain. Therefore, the availability of food is expected to decrease substantially within the region and it will be interesting to test whether competition between sexes will change, and if this may lead to spatial segregation between the sexes.

We detected a significant effect of age on distance to the natal colony. As expected (Munilla 1997), adults and immatures remained closer to the colony than juveniles during the breeding season. Greater distances to the colony were detected for juveniles (from the hatching year to June of the next year) compared to older individuals in the breeding season. This is a very common phenomenon in seabirds, and it can be due to either the saturation of the area around the colonies by breeding adult birds (Gaston 2004) or simply because adults are restricted in their range as they must return to the colony for reproduction. Immatures could benefit from staying close to the colonies, since this would allow them to learn where to find food and gain social experience when prospecting areas for breeding (Dittmann *et al.* 2005). Our findings indicate that this population was very faithful to its natal area. This finding is in contrast with the Mediterranean Yellow legged Gull population, that is partially migrant (Galarza *et al.* 2012), but not with the Macaronesian population, that is also sedentary (Romero *et al.* 2019).

The multi-state CMR models did not reveal different movements patterns between the sexes. Overall, our gulls had a higher probability of moving further than 100 km away than to return to areas near their natal colonies, with such a pattern being more marked in winter. This pattern, however, does not mean that our population acts as a source from a demographic view (i.e. emigration exceeding return rate considering birds born within the region), because immigration was not measured. Finally, survival was found to be lower among first-year birds (c. 0.55) than in older birds (around 0.90). These apparent survival estimates are similar to those estimated in previous studies, indicating that survival has remained rather constant in the last decade (Arizaga *et al.* 2015b, Juez *et al.* 2015). Future research with GPS-tracking will be key to learning when and where juvenile mortality rates is at its highest, leading to a better understanding of their risks and vulnerable periods.

Such new technologies may also reveal otherwise undetectable small-scale niche partitioning between the sexes (e.g. Camphuysen *et al*. 2015).

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ARTÍCULO 3

High philopatry rates of Yellow-legged gulls in the southeastern part of the Bay of Biscay.

Abstract

Philopatry rate is one of the main factors shaping population dynamics in colonial seabirds. Low rates of philopatry are linked to populations with high dispersal, while high rates are linked to populations with a very high spatial structure pattern (i.e., metapopulations). The Cantabrian Yellow-legged gull (*Larus michahellis*) population is considered to be resident, with relatively low dispersal rates. Precise estimations of its philopatry rates are however still lacking. Here, we aimed to estimate philopatry rates in the main Yellow-legged gull colonies of the province of Gipuzkoa, in the southeastern part of the Bay of Biscay. We analysed 734 resightings, during the breeding season at the colonies of Getaria, Santa Clara and Ulia, relative to a total of 3245 individuals ringed at birth in these same colonies during a period of 13 years. These data were analysed using Multi-State Recapture models in MARK. After controlling survival and resighting probability, the average dispersal rate among colonies was 4% (±SD = 2%) when individuals are immature, decreasing to 1 ± 1%) for adult breeding gulls (i.e., philopatry rate was 99%). Annual survival rates were assessed to be 0.27 ± 0.02 for birds in their first year of life and 0.87 ± 0.01 for older individuals. The probability of observing immature birds in the colonies was 0.08 ± 0.01, as compared to 0.21 ± 0.02 in adult birds. We obtained evidence of extremely high local philopatry rates, clearly within the upper limit found in gulls. A high philopatry favour a speciation in these species who are vulnerable to obtain the main food source (landfills and fishing discard) which are transforming under new ecological process.

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In population dynamics, philopatry is the habit by which individuals show fidelity to a site, hence natal philopatry would refer to those breeding in or close the place where they were born. Recruitment, however, refers to new individuals joining a population, either by birth or by immigration, thus involving philopatric specimens, together with those that, coming from abroad, also settle (recruit) in a place to breed (e.g., Croxall & Rothery 1991). Note, therefore, that this definition approximately fit to the one provided by Pradel (1996), where recruitment is shown to be equivalent to survival in reverse, and can be carried out by inverting capture histories. In colonial birds, therefore, we can calculate the degree of philopatry to natal sites (i.e., to origin colony), i.e., the extent (or rate) to which birds from given colonies return (recruit) to their natal colony (Pradel *et al.* 1996, Hafner *et al.* 1998, Oro *et al.* 2013). If we call this last concept 'local' recruitment, it can be then stated as a synonym of philopatry.

The local recruitment among close-by well-established colonies has been one of the least known aspects on seabird biology (Gaston 2004), though the body of studies dealing with this subject is increasing (Bosch *et al.* 2019, Davis *et al.* 2019, McKnight *et al.* 2019). The analysis of local recruitment rates in colonial seabirds is called to play a very relevant role in our comprehension of the dynamics and demographic factors driving seabird populations (Spear *et al.* 1998, Cam *et al.* 2002, Payo-Payo *et al.* 2015).

Recruitment rates in gulls can differ greatly both inter- and intra-specifically. In general, local recruitment rates will be higher in colonies that grow fast (Oro & Pradel 2000) and that have not reached their maximum carrying capacity (Newton 1998, 2013). Or when birds find cues indicating good conditions, such as a high reproductive output (Cadiou *et al.* 1993, Cadiou *et al.* 1994), high colony size (Oro & Pradel 2000), or high food availability (Spear *et al.* 1995). Abnormally high mortality rates in adult birds would promote local recruitment as well (Votier *et al.* 2008) due to the occurrence of vacant places within the colonies. By contrast, these rates will decrease when a colony shows density-dependence (e.g., saturation of breeding places) (Coulson & Coulson 2008) or other external factors that will promote natal dispersal, such as culling programs carried out in the colonies (Chabrzyk & Coulson 1976, Parsons & Duncan 1978).

The Yellow legged gull *Larus michahellis* is the most abundant gull in the southwestern Palaearctic (Olsen & Larson 2004). Its abundance is linked to the high availability of food subsidies from landfills or fish discards. Overall, this superabundance of food, that can be very high at a local scale, is considered to foster sedentary habits and reduce dispersal processes to a minimum (Gilbert *et al.* 2016). In such a scenario, gulls would be expected to show high local recruitment rates, as they would tend to show a high degree of philopatry to those colonies where they find cues indicating good conditions (Gaston 2004). Previous research with the species has assessed local recruitment rates under scenarios where culling programs were active (Bosch *et al.* 2000, Bosch *et al.* 2019) or using reference estimates from other gull species of similar ecology (Brooks & Lebreton 2001), but where, again, the colonies were subject of culling (Chabrzyk & Coulson 1976, Parsons & Duncan 1978). Therefore, there is still a gap of knowledge on this topic.

Our aims in this work were to: (1) estimate natal philopatry rates in a Yellow-legged gull population non-intervened by culling consider the local recruitment and the philopatry of adults (with the immigration rate), and (2) survival, in order to test whether high philopatry rates might be linked with low survival in adults since these low rates could facilitate vacant places within the colony which would be available for new breeders(Votier *et al.* 2008). A priori, we hypothesize high local philopatry rates, since previous works on this population have shown very low dispersal and short-range movements from breeding colonies (Arizaga *et al.* 2010, Egunez *et al.* 2017). To address this work, we used data collected over a period of 13 years in three Yellow-legged gull colonies from the Bay of Biscay (Spain).

METHODS

Study area and data collection

This study was carried out in the most important Yellow-legged gull colonies from Gipuzkoa (North of Spain; from east to west): Ulia (43°20'N 01°57'W, ha:1.25), with 660 adult breeding pairs (last census from 2017); Santa Clara (43°19'N 01°59'W, ha:3.28), with 100 pairs; and Getaria (43°18'N 02°12'W, ha:0.7), with 165 pairs. The distance from Ulia to Getaria (the two colonies situated in the two furthest coordinates) is roughly 20 km.

A variable number of chicks were ringed within each colony on a yearly basis since 2005 (Ulia and Santa Clara) or 2006 (Getaria). Overall, up to 2018, we ringed 3285 chicks (for details see Table 9). The chicks were ringed at the age of ca. 20 days (range: ca. 15 to >30 days), with both a metal ring and a Darvic ring with an alphanumeric code (red ring with four white numbers/letters), with no differences among years or colonies. Previous study in Ulia breeding colony shows a pre-fledging survival of 60% per chicks, conditioned to hatching date (Delgado & Arizaga 2017). Together with this task, we also compiled all the re-sighting data (ring-readings) obtained across the entire year, both by our teams and by fellow birdwatchers., However we only selected those obtained in the colonies during the breeding period and comprised a three-month period from April (eggs' laying) to June (grown chicks almost ready to fledge). We assumed that adult individuals found inside the colonies were breeding. Overall, up to 2018, we collected 853 readings of 398 individual, but 111 gulls had more than one sighting in a year, hence we removed 119 readings from the data set, resulting in 734 resightings (Table 9). In four occasions (0.46%), an individual bird was seen in the same year in different colonies, we selected the resighting from the original colony or the one closer to it.

Statistical analyses: philopatry models

We used the software Mark (White & Burnham 1999) to estimate local recruitment rates in each of the three study colonies. In particular, we used multi-state recapture models (Lebreton *et al.* 2009). Before starting to build any model, we tested for the fit of the data to Arnason-Schwartz assumptions.

Table 9. Number of chicks ringed and subsequently number of different encounters (resightings) per colony along the period 2005 to 2018. The number of chicks ringed in total is 3285.

	Getaria		Santa Clara		Ulia	
	Ringed	Resightings	Ringed	Resightings	Ringed	Resightings
2005	0	-	23	-	17	-
2006	30	0	69	0	129	2
2007	10	12	85	6	202	0
2008	38	0	55	5	194	5
2009	20	27	50	12	265	7
2010	59	21	42	18	221	38
2011	32	6	37	0	129	17
2012	63	9	86	4	130	22
2013	50	10	54	16	68	66
2014	50	11	59	10	151	31
2015	50	0	35	0	141	12
2016	33	15	40	12	162	35
2017	33	5	27	9	81	37
2018	49	31	52	28	164	195

We used for that a goodness of fit test ran in the software U-CARE (Choquet *et al.* 2009). The global GOF test was non-significant ($\chi^2 = 44.14$; $df = 80$; $P > 0.05$), test 3G for trap-dependence ($\chi^2 = 29.59$; $df = 58$; $P > 0.05$) was not significant and test M for transients ($\chi^2 = 14.56$; $df = 12$; $P > 0.05$). Our models provided three different parameter estimates: (1) movement rate among sites (states), ψ; probability of moving from a site (colony) a to b, contingent on survival. Applied to our colonies the parameter ψ may be the probability of a bird ringed as a pullus in colony a to be subsequently seen in colony b; (2) survival, S; probability of surviving from year t to year t+1; S can be calculated for each site (colony); (3) encounter rate, *p*; probability of detecting a bird which is alive in a site. Note that, concerning *S*, what we estimate here is the apparent local survival, because dispersal outside the colonies will appear as mortality.

In Mark, the models were built under the 'multi-state recaptures only' framework. For that, 'recaptures' (i.e., resightings) from Ulia were coded as 1, Santa Clara as 2, and Getaria as 3. The distance between colonies was: Ulia-Santa Clara, 3.54 km; Ulia-Getaria, 19.60 km; Santa Clara-Getaria, 16.32 km.

First, we built a model where all the parameters (abbreviated as ψ, *S*, *p*) were constant (.) or time dependent (*t*). Afterwards, we ran a set of alternative models considering an effect of age, colony, and age and colony on the parameter estimates. We considered up to 3 age categories: 2a, 3a and 4a. 2a refers to models assuming a difference between two age categories (birds in their first year of life and older birds); 3a refers to models assuming a difference between birds in their first and second year, and older birds; 4a refers to models assuming a difference between birds up to their third-year of life (immatures), and older birds (adults). Given that most birds hatch by the end of May or early June (Arizaga *et al.* 2012), a year was considered to last from June to the end of May of the subsequent year. To run such models: (1) we started by fixing ψ and *S* as constant and time dependent, and testing for the best options fitting on *p*. Alternative models were ranked in relation to their small sample size-corrected Akaike values (AICc; Burnham & Anderson 1998); models differing in less than 2 AICc values were considered to fit to the data equally well (Burnham & Anderson 1998). (2) Once we found the model best fitting on p, we repeated a similar process on *S*, and, finally, on ψ.

RESULTS

Out of 3285 chicks ringed from 2005 to 2018, 398 (12%) were seen in the colonies one or more years after-fledging (Table 9). Most resightings of ringed gulls were detected when the gulls were adults (*n* = 653; note here than many individuals were seen more than once); by contrast, a small fraction of the birds were seen in the colonies as immatures (*n* = 93) (Table 10). The proportion of resightings of adults in any colonies was higher in Ulia (95.9%) as compared to Santa Clara (77.6%) or Getaria (77.5%) (for further details see Table 10). Out of these 398 birds, 205 (51.5%) were observed only once, whilst 193 (48.5%) were seen in more than a single breeding period. The number of gulls found in a colony different from the natal one was just 18 (4.6%) (Table 10); these birds were seen in 64 occasions. Only one individual was observed to visit the three sampling colonies, and just four out of these 398 birds were found in more than one colony.

Table 10. Number (percentage in brackets) of resightings within each colony (columns) in relation to the origin colony (rows). Immature gulls comprise birds of up to three years of life; older birds have been considered as adults.

Age class		Immatures			Adults	
Origin colony	Getaria	Santa Clara	Ulia	Getaria	Santa Clara	Ulia
Getaria	25 (92.6%)	6 (18.7%)	9 (26.5%)	107 (97.3%)	16 (13.1%)	15 (2.8%)
Santa Clara	2 (7.4%)	26 (81.3%)	3 (8.8%)	3 (2.7%)	100 (82.0%)	4 (0.7%)
Ulia	0	0	22 (64.7%)	0	6 (4.9%)	509 (96.4%)

From 28 models tested in total, only one was detected to fit the data better than the rest (Table 11). The second model was found to have an AICc value 15 units higher than the first one, as also the third-ranked model had

(Table 11). The probability of moving between nearby colonies was age-dependent, it differed among colonies (Figure 3), and was very low overall (0.025, 2.5%). In immatures, such flow rates ranged from 0.01 (1%) to 0.09 (9%) (for details see Fig. 3). In adults, such rates were even smaller, being virtually close to zero (Figure 3), indicating that, once established in a colony, an adult was very unlikely to move to another one.

Table 11. Best-ranked models used to estimate a total of three demographic parameter estimates: local recruitment rates, ψ; annual survival, S; encounter rate, p. For comparison, we also add the model with constant and time-dependent parameter estimates. Abbreviations: AICc, small sample sizes-corrected Akaike values; ΔAICc, difference of AICc values in relation to the first model; Np, number of parameters.

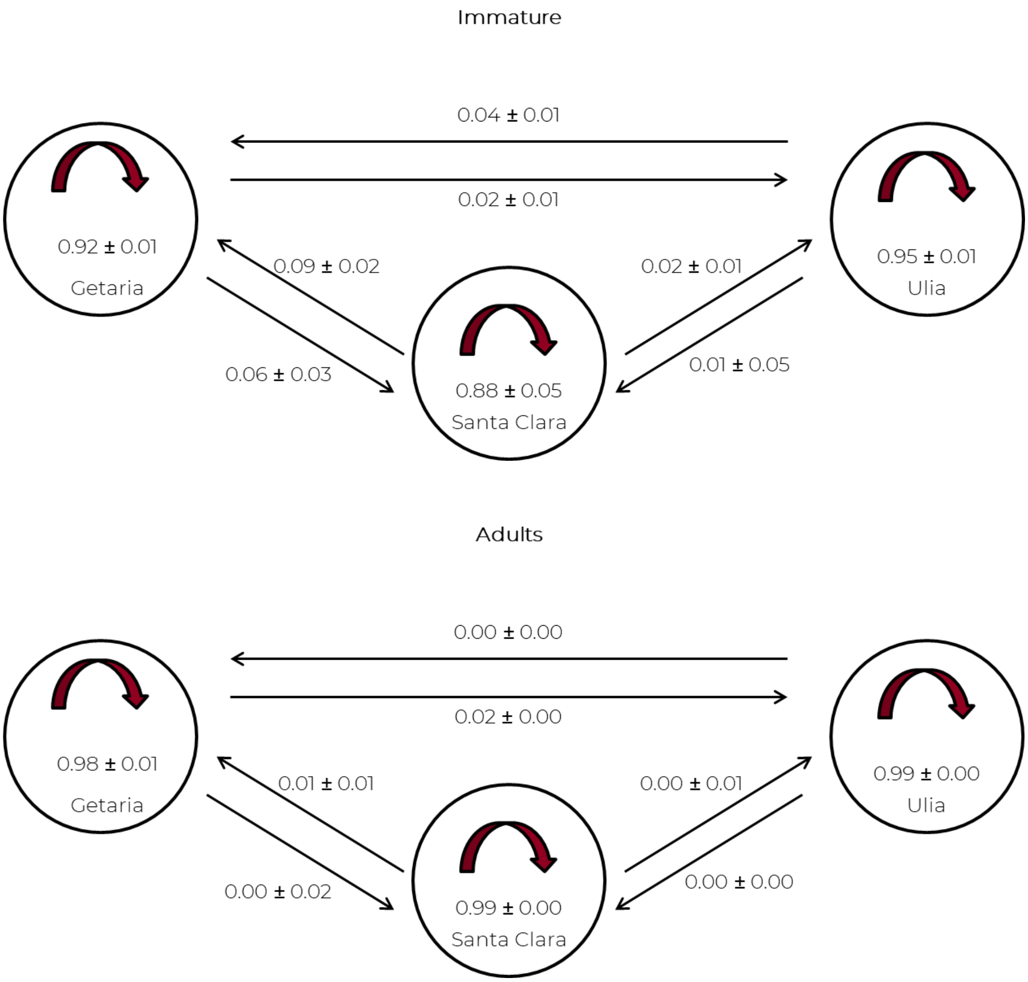
Models	AICc	ΔAICc	AICc weight	*Np*	Deviance
Ψ(colony+4a),*S*(2a), *p*(colony+3a)	5665.27	0	0.9995	18	1735.92
Ψ(4a), *S*(2a), *p*(colony+3a)	5680.45	15.18	0.0005	10	1767.22
Ψ(colony), *S*(2a), *p*(colony+3a)	5696.30	31.08	0.0000	14	1775.02
Ψ(.), *S*(*t*), *p*(*t*)	6359.11	693.84	0.0000	54	2356.37
Ψ(.), S(.), *p*(.)	6367.65	704.52	0.0000	7	2460.45

Conducted models consider either a constant, colony and/or age-dependent effect on parameter estimates. Age categories have been lumped into 2a, 3a or 4a. 2a refers to models assuming a difference between two age classes (birds in their first calendar year and older birds); 3a refers to models assuming a difference between birds in their first and second year, and older birds; 4a refers to models assuming a difference between birds up to their third-year of life, and older birds.

Regarding survival estimation, we observed that this parameter varied between two age classes, whilst it was not colony-associated. Annual survival in the first-year gulls was 0.27 (95% CI =: 0.24, 0.31), increasing up to a mean of 0.86 (95% CI = 0.84, 0.88) in older birds (> second-year gulls, Fig. 4). Finally, encounter rates differed between age classes and colonies (Fig. 5), being lower in immatures (<15%), and in the Santa Clara colony, and higher in adults and the colony of Ulia (Getaria showed intermediate values; Fig. 5).

In the first one, local fidelity rates varied among colonies and in relation to age classes. Overall (birds older than 4 years), these rates ranged from 0.98 (±SE = 0.014) to 0.99 ± 0.041 in adults, being only slightly smaller in Getaria (Fig. 3). In immature gulls, local philopatry rate varied more markedly among colonies, from 0.88 ± 0.023 in Santa Clara to 0.92 ± 0.010 in Getaria or 0.95 ± 0.001 in Ulia.

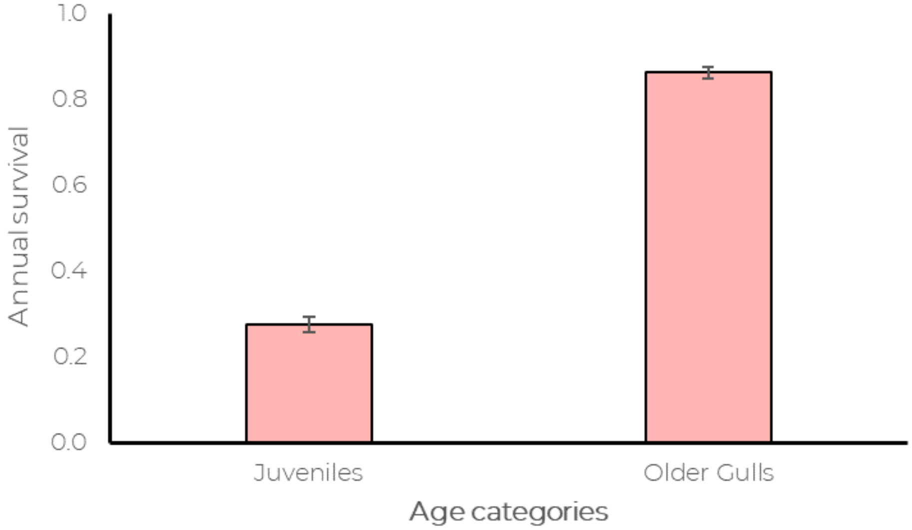
Figure 3. Movement rates in three Yellow-legged gull colonies from Gipuzkoa, northern Spain, as obtained from the model one of Table 11. Estimates have been accompanied ±SE.



DISCUSSION

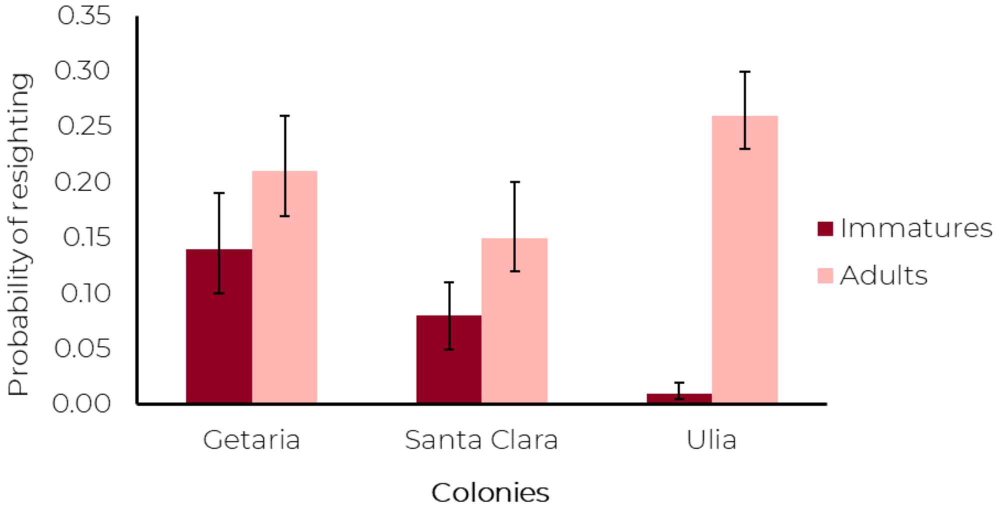
This study is one of the few ones analysing local philopatry rates in a Yellow legged gull population. Our colonies showed extremely high philopatry, with rates, close to 100%. It is true that this result might partly be biased by the fact that our sampling colonies were located close between each other, hence ignoring those birds which might recruit in colonies located at further distances (Coulson & Coulson 2008).

Figure 4. Annual survival probabilities (S in models and ± SE) of the two main age categories with all the breeding colonies pooled.



However, during the period of 14 years in which this study was carried out, we were notified to have only 21 birds breeding in colonies found at further distances, and all of them located to the west of the western-most colony of Getaria: Lekeitio (25 km), Bermeo (44 km), Santander (130 km). Furthermore, even within our small sampling range in Gipuzkoa, an extremely low number of birds were found to breed in a colony different from the one where they hatched. Overall, evidence supports that the philopatry rates are truly high within the population. In this research we were not able to estimate the first age of reproduction due to two main reasons: the orography and the fact that most gulls fly when we approach to the colony, which hampers us from confirming whether a certain bird is breeding or not. Age of recruitment is an aspect that remains for future studies (Porter & Coulson 1987, Cadiou *et al.* 1994, Spear *et al.* 1995, Coulson & Coulson 2008, Oro *et al.* 2013). Therefore, we had to assume that adults were breeding, so by default the first age of reproduction was the 4th year of life, which is the age when the species reaches its sexual maturity (Olsen & Larson 2004).

Figure 5. Resighting probability (p in models from Table 11; bars: 95% confidence interval) of the two main age categories within in each colony.



This assumption, however, is rather irrelevant for the conclusions made in this study, since many gulls were sighted several years after their 4th year of life, and the flow among colonies was extremely low.

What we were able to confirm is that immatures were rare within the colonies and this is due to the fact that these birds do not have the need to breed, hence they remain in other areas where they can feed and rest without getting in conflict with adults. The observation of these immature birds within the colonies may be associated to prospecting processes (Dittmann *et al.* 2005). Even though our data set comprised mostly adult birds, since immatures avoid entering the colonies, therefore, they are much more unlikely to be seen, we still had data to estimate their movement among colonies. During prospection, seabirds visit several sites in order to evaluate the quality and suitability of each colony for future breeding (Dittmann *et al*. 2005). It is interesting to note here that in Gipuzkoa most immatures were found in their hatching colonies, i.e. the philopatry was high even for these immature birds, suggesting that the prospection was small or very fast, until most birds decided to settle in their origin colony. This suggests that, if prospection exists, our birds may decide at relatively early age recruiting in their natal colony; otherwise, the proportion of resightings at their natal sites would be remarkably lower.

Ultimate causes underlying such a high philopatry in these colonies still remain to be determined, so we cannot do more than advancing some plausible explanations. (1) The familiarity with hatching site may have played a role in this phenomenon (Greenwood & Harvey 1982), but not the colony size (as found by Oro & Pradel 2000), since local recruitment was not higher in Ulia, by far the largest colony within the region. (2) Differences in feeding patterns between the colonies can also influence these high rates of fidelity to natal sites (Enners *et al*. 2018). In spite of their proximity, our colonies are well known to depend on different feeding resources (Zorrozua *et al*. 2020), and maybe the juveniles from a given colony would tend to specialize on feeding on those resources exploited by their parents which, consequently, would favour to settle in their hatching sites. (3) The density of the colony has been reported to have a negative effect on recruitment: in areas with high rates of mortality in adults (Duncan 1978), the recruitment for immature individuals is possible in greater number. However, this does not seem to be the case for our colonies. Adults' annual survival was assessed to be almost 90%, which might be considered a 'normal' value for a *Larus* gull (Gaston 2004). After applying this estimate in a simple population model, we found that our population was stable or even experiencing a slight, moderate increase (Supporting information 1). In this scenario, it could also be stated that the colonies may have a higher carrying capacity, which would still allow a high philopatry (Duncan 1978). (4) High recruitment rates would also be possible in contexts of very high availability of food subsidies, both due to the existence within the region of a number of fish harbours with high activity (Arizaga *et al.* 2011, Zorrozua *et al.* 2020), as well as some still open-air landfills (Egunez *et al.* 2017, Arizaga *et al.* 2018). However, the landfills still remaining in the region are expected to be closed in a very short-term period, which is expected to have direct consequences on the dynamics and trophic ecology of this population (Steigerwald *et al.* 2015, Zorrozua *et al.* 2019). A sudden food shortage should result in a decreasing reproductive output or survival (Oro *et al.* 1995), and may also increase dispersal (Arizaga *et al.* 2014b), and it's likely to generate population declines relatively fast (Galarza 2015). As a consequence, local recruitment rates would be lower because natal dispersal would be expected to increase (Oro & Pradel 2000). This incoming new scenario will offer us an excellent opportunity to test for the effect of landfill closure on local recruitment rates.

Globally, our results are within the upper limit of other species where very high rates of local recruitment have been also detected, e.g., in colonies of Audouin's gull (Oro & Pradel 2000). By contrast, other smaller gulls, such as the black-legged kittiwake (*Rissa trydactyla*), were shown to have much smaller local recruitment rates, ranging between 35% and <10% (Porter & Coulson 1987, Coulson & Coulson 2008, McKnight *et al.* 2019). Estimates for the Herring gull (*Larus argentatus*) have also reported considerably low local recruitment rates (<40%) (Chabrzyk & Coulson 1976, Parsons & Duncan 1978), though in this case the cull of breeding birds could have promoted philopatry rates values abnormally high (Bosch *et al.* 2000, Bosch *et al.* 2019). The effect of external factors like culling programs on recruitment is important, and the values from some colonies may not be extrapolated to other colonies since this would lead false premises in demographic models (Brooks & Lebreton 2001). Even though the use of mean values is a common practice when building population models, we highlight here the need to be very cautious, and to estimate values obtained from the surveyed colonies always that this is possible.

Local apparent survival estimation values did not vary substantially as compared to previous works (Juez *et al.* 2015), providing a relatively low value for the first year of life (where the first weeks after fledgling are the most critical ones; Genovart *et al.* 2017). The real survival value for these first-year birds after fledging could indeed be slightly higher, since all these birds were ringed when they were chicks, so pre-fledging mortality should be considered. It is also true, in addition, that pre-fledging survival varies with the chick age, being lower for those birds ringed at an age closer to fledging time (Delgado & Arizaga 2017). Overall, if we assume that that daily mean survival rate from hatching to fledging is ca. 0.98, and that the chicks were ringed when they were 20 days, their survival from ringing to fledging is roughly 0.668 (Delgado & Arizaga 2017). Thus, survival from fledging to next year could be about 0.40. As we assessed apparent local survival rates, low values in first-year birds might be also interpreted as an artefact associated to dispersal outside our three-colony system. This is possible, and it is probably the case for some individuals, but our population is resident, and the majority of birds remain close to their natal colonies even in their first year of life (Arizaga *et al.* 2010). Therefore, it can be stated that local survival rates must be rather close to true survival.

The survival estimation of adults felt within the range found for other large gulls (Chabrzyk & Coulson 1976, Pons & Migot 1995). Our models did not detect differences among colonies hence suggesting that, within such a small geographic range, the factors driving the survival of our population may operate at spatial scales larger than a very few kilometres around each colony.
Encounter probabilities were, overall, relatively low (0.15 to 0.26 in adult birds), showing that finding ringed individuals in the colonies was relatively difficult. A higher sampling effort should help to enhance these results, though, as deducted from the relatively low confidence intervals obtained for the other two parameter estimates, we consider that this higher effort may not have a statistical effect on recruitment or survival. Encounter probabilities were found to be still much lower in immature gulls. Causes underlying this result must focus on the fact that immature specimens tend to remain outside the

colonies, often exploiting different foraging areas (Pettex *et al*. 2019). The colony-dependent variation in these encounter rates is attributed by us to the topography and the accessibility.

In conclusion, the Yellow legged gull population breeding in the southeaster part of the Bay of Biscay showed remarkably high philopatry, indicating a high dependency to natal sites. Such rates might be expected to change in a nearby future, where the still existing open landfills will be closed, hence promoting higher dispersal, probably lower survival rates and also decreasing local recruitments.

ACKNOWLEDGEMENTS

This research was partly funded by the Basque Government and the Gipuzkoa Regional Council. This last administration provided the permits to capture and ring the gulls. We are grateful to the volunteers and birdwatchers that, during all these years, have reported sighting data. SD benefited from a pre-doctoral fellowship from the Basque Government. Two referees provided valuable comments that helped us to improve an earlier version of this work

ARTÍCULO 4

Demographic impact of landfill closure on a resident opportunistic gull

Abstract

The management of mixed municipal waste can have an impact on wildlife and ecosystems. Previous studies have investigated how opportunistic species like gulls can react very fast to new landfills, however, the impact of landfill closure on bird populations is less investigated. Yet, there is a need to understand how fast and to what extent, animal populations can be adapted to new scenarios where the waste will not be deposited in landfill sites anymore. The aim is to determine the influence of landfill closures on apparent survival of a resident Yellow-legged gull (*Larus michahellis*) population, used as a model species showing short-distance foraging movements, and with a high dependence on local food subsidies. Complementarily, we built some basic population growth models in order to determine how potential changes in survival (before/after landfill closure) will impact on population growth rate. Using a data set of 4437 Yellow-legged Gull chicks ringed in four colonies over a period of 13 years, we obtained evidence supporting that the apparent survival was affected by landfill closure, especially if the landfill was located within a buffer of 10 km around the colony. Landfill closure affected the survival of first-year gulls (with a mean decrease of ca. 0.5 to 0.36), but not of older birds. Consequently, we did not detect a remarkable effect of landfill closures on population growth rate, probably due to the lack of effect on adult survival rates except for one of the surveyed colonies, where we found an annual decline of 7%.

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The management of mixed municipal waste can have a very strong impact on wildlife and ecosystems (Hobson *et al.* 2015, Oro *et al.* 2013, Seif *et al.* 2018). Waste concentration produced by humans in landfill sites opens new opportunities for those species which are able to exploit this feeding resource. This superabundant food subsidy is clearly advantageous for them in terms of increasing reproductive outputs and survival prospect (Real *et al.* 2017, Weiser & Powell 2010), but also has some, very critical disadvantages, including the ingestion of plastics (Seif *et al.* 2018, Witteveen *et al.* 2017) and the exposure to higher concentrations of pollutants or diseases (Monaghan *et al.* 1985, Ramos *et al.* 2010, Roscales *et al.* 2016), among other threats. The presence of a landfill does not only change the diet of animals feeding on it, but also can alter their decision to breed in some places and not in others. Thus, several opportunistic species worldwide move to or concentrate at abnormally high densities near landfills (Belant *et al.* 1998, Duhem *et al.* 2008, Gilbert *et al.* 2016, Hidalgo-Mihart *et al.* 2004, Tortosa *et al.* 2002). Thus, landfill sites produce great impacts on the diet, demography and spatial distribution pattern.

The relationship between opportunistic species and landfills is particularly well documented in gulls, probably as a consequence of conflicts with the humans (Vidal *et al.* 1998, but see Belant 1997, Brown *et al.* 2001, Oro & Martinez-Abrain 2007, Rock 2005). Thus, several previous studies have investigated how landfills influence gulls' diet (Arizaga *et al.* 2013, Duhem *et al.* 2003, Ramos *et al.* 2009) or reproduction (Belant *et al.* 1998, Real *et al.* 2017, Steigerwald *et al.* 2015, Weiser & Powell 2010). In general these birds respond rather fast to new landfills and their populations, therefore, grow exponentially in relative short-time periods (Arizaga *et al.* 2009, Duhem *et al.* 2008). At the same time, however, the populations can become very landfill-dependent, so very vulnerable to threats related to feed in such places (e.g., exposure to diseases, pollutants, or being hit by the heavy machinery). The impact of landfill closure on bird populations is, however, much less investigated than impacts when these sites are open (Payo-Payo *et al.* 2015, Steigerwald *et al.* 2015). Yet, there is a need to understand how fast and to what extent, animal populations can be adapted to new scenarios where the waste will not be deposited in open landfill sites anymore (for instance see the European laws Directive 1999/31/UE, Directive 2008/98/CE).

The closure of a landfill would have then both short- and long-term consequences in a gull population. First, those individuals which forage on this resource will have to find alternative food (Zorrozua *et al*. 2020a). In principle, it can be stated that this alternative food will be scarce or, directly, will not exist, since quite often the system had a carrying capacity much smaller than the one existing when the landfill was open (Duhem *et al*. 2008). Therefore, three potential scenarios might be possible: (1) if there is an alternative feeding resource in sufficient amount, the change may affect the diet but, presumably, may not have demographic consequences (unless the new resource has a different nutritional value with consequences in the reproduction and survival); (2) if there is an alternative feeding resource but not in sufficient amount, hence competition would increase/appear and only a fraction of the population may be able to change to this new resource and benefit from it; the remaining individuals, however, would be expected to leave the population, either because they disperse to other areas to survive or perish given the lack of food; (3) if there is not an alternative food, then the population may be expected to collapse through density-dependent processes (Newton 2013). All these scenarios should be more critical in resident populations, that might show a much higher dependence on given local resources than populations that inhabit a given area only during part of the year.

In the scenarios 2 and 3, a critical aspect would be to estimate how survival is affected by landfill closure. In scenario 2, the apparent survival (including survival and emigration) would be expected to decrease in those individuals with a subordinate status within the population, e.g. the immature fraction compared to adults when both age classes feed on same feeding sources. In scenario 3, however, the apparent survival would be expected to decrease markedly in all age classes.

Another, also critical question is to quantify the area of influence of a landfill. Landfills attract individuals from the surroundings and, even though this attraction will depend on species' movement capacity, there will be a maximum distance from which it would not be advantageous for an individual to travel and feed on a landfill (Egunez *et al*. 2017). Determining this landfill area of influence is important as it has direct consequences on wildlife management and conservation measures, i.e. which populations or individuals would be affected if a landfill is closed.

The aim of the present article is to determine the influence of landfill closures on the apparent survival of a resident Yellow-legged gull (*Larus michahellis*) population, used here as a model species showing short-distance foraging movements, and with a very high dependence on local food subsidies, including landfills (Zorrozua *et al*. 2020b). The specific hypotheses that we tested here were: 1) Closure of those landfill sites located closer to the breeding colonies will have a higher impact on survival as compared to landfills located further away; 2) If there are no alternative and sufficient feeding sources, the impact on survival will decrease with the age of the individuals (i.e., from first-year birds to adults) if the older birds are able to compensate for the lack of landfill food, or alternatively all age classes will suffer similar effects on survival if all of them are not able to compensate for this food shortage. Complementarily, we built some basic population growth models in order to determine how potential changes in survival (before/after landfill closure) will impact on population growth rate (presumably, our population may pass from a stable/increasing status to decreasing). This last exercise was carried out with the aim of determining the demographic consequences of landfill management on our avian model.

MATERIAL AND METHODS

Model species

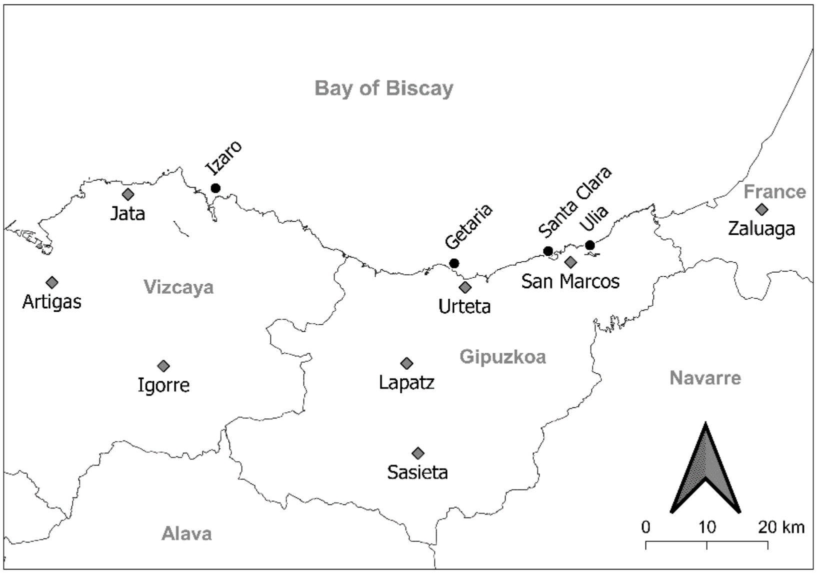
The Yellow-legged gull is the most abundant gull within the southwestern Palearctic (Olsen & Larson 2004). It is a partial migrant species, with the Atlantic populations that breed both along the coast of Iberia and Africa and in Macaronesia being resident, and the ones breeding in the Mediterranean showing partial migration (Cramp & Simmons 1983, Galarza *et al.* 2012, Martínez-Abraín *et al.* 2002; Olsen & Larson 2004; Rodríguez & Muntaner 2004; Romero *et al.* 2019). As other large gulls, the species has been adapted to feeding on landfills, a phenomenon that permitted a very fast population growth and, possibly, the colonization of new areas and the broadening of its distribution range (Castège *et al.* 2016, Skorka *et al.* 2005, Yésou 1991).

We worked here with a resident Yellow-legged gull population situated in the Bay of Biscay (Spain). Individuals from this population move a mean distance of less than 50 km from their natal colony sites, and they had/have are highly dependent on landfill food (Arizaga *et al.* 2013, Egunez *et al.* 2017, Zorrozua *et al.* 2020b,). During the last decades between the last and the current century, the existence of several landfills located near the main breeding colonies of this species within the region favoured the increase of its population (Arizaga *et al.* 2009). However, the landfill use decreased with distance to the colony and temporal landfill closures were associated with increasing movement distances (Arizaga *et al.* 2014b, Egunez *et al.* 2017). Furthermore, one of the colonies within the region was observed to decrease fast when a landfill situated nearby was closed (Galarza 2015). There was, therefore, previous evidence supporting that the area of influence of the landfills was relatively local.

Study area and data collection

This research was carried out in the main four Yellow-legged gull colonies in the Basque coast, southeastern part of the Bay of Biscay (from east to west): Izaro island (with ca. 400 adult breeding pairs censused in 2017), Getaria (165 pairs), Santa Clara (100 pairs) and Ulia (660 pairs) (Fig. 6). Overall, these colonies host >80% of the Yellow-legged gull population within the region (Arizaga *et al.* 2009).

Figure 6. Location (circles) of the four-sampling Yellow-legged gull colonies in the Bay of Biscay and all the landfill sites situated at 50 km or less from each colony (diamonds). These remained either open or were closed along the study period (2006-2018; for details see Supporting Information 3).



Every year between 2006 and 2018, chicks of ca. 20 days old were ringed by the end of June with both an official metallic ring in one of leg and a second ring with an alphanumeric code (to be read at distance) in the other leg (provider: R. Juvaste). After fledging, these ringed birds were seen during the study period by multiple observers (‘gull-watchers’), in many zones. We retained the observation made from April to June, from 2007 to 2019 and coded them into capture histories of individual birds (Table 12). We considered sighting data obtained in as well as outside the colonies.

Table 12. Number of chicks ringed during the breeding period in four Yellow-legged gull colonies in the Bay of Biscay during the period 2006–2018. In parenthesis, we also show the percentage of individual birds that were seen after they fledged from their natal colony.

Year	Izaro	Getaria	Santa Clara	Ulia
2006	232 (3.9)	30 (33.3)	69 (34.8)	147 (37.4)
2007	103 (6.8)	10 (40.0)	85 (30.6)	202 (37.1)
2008	49 (22.5)	38 (31.6)	55(38.2)	208 (41.4)
2009	30 (36.7)	20 (35.0)	50 (22.0)	258 (41.1)
2010	105 (33.3)	59 (39.0)	43 (18.6)	221 (29.9)
2011	90 (33.3)	32 (18.8)	37 (29.6)	185 (31.4)
2012	58 (12.1)	109 (12.8)	86 (20.9)	168 (28.6)
2013	38 (2.6)	50 (32.0)	52 (32.7)	68 (27.9)
2014	45 (15.6)	50 (12.0)	59 (23.7)	151 (22.5)
2015	62 (16.1)	50 (14.0)	32 (21.9)	141 (15.6)
2016	68 (42.7)	54 (24.1)	40(30.0)	158 (17.7)
2017	54 (11.1)	52 (15.4)	27 (14.8)	81 (13.6)
2018	56 (1.8)	49 (4.1)	51 (19.6)	170 (8.2)

Landfill management

For this resident Yellow-legged gull population (Egunez *et al.* 2017), we considered the eight landfill sites situated within 50 km from the four colonies (Fig 6.; Supporting Information 2). During the research period six landfills were closed gradually (Supporting Information 3); the first one to be closed was S. Marcos (in October of 2008), followed by Igorre, Jata, Urteta, Sasieta, Lapatz. Jata, however, was re-opened in 2016. Zaluaga and Artigas remained open during this study. Overall, therefore, we obtained a combination of open/closed landfill sites situated at different distances from the four colonies where the birds were ringed. This distance matrix was used as external covariates when modelling gull transitions.

Statistical modeling

To assess the effect of landfill management on survival we built Cormack-Jolly-Seber (CJS) models in MARK (White & Burnham 1999). These models permit to estimate apparent survival (φ, probability that a bird survives from t to $t+1$) and the probability of recapture (p, the probability that a bird estimated to be alive at time t is seen at $t+1$) separately. Basic assumptions of capture-mark-recapture analysis (CMR) were evaluated with a goodness-of-fit (GOF) test of a general model assuming all parameters time-dependent (the Cormack–Jolly–Serber, CJS) using U-CARE 2.3 (Choquet et al. 2009). The global GOF test was not statistically significant ($\chi^2 = 82.009$, $p = 0.999$, $df = 142$), nor the specific Z test used to detect trap dependence ($p > 0.636$) or transients ($p > 0.464$) indicating that the CJS model explained fitted to the data adequately. We contrast this model with simper ones assuming time-dependent φ and p, alternatively and a set of models with an effect of the colony of origin (colony-dependence). Note here that in this case we assigned each individual bird to one out of the four categories of origin (i.e., ringing colony). Thereafter, we built more models to test for the effect of landfill management (open vs close).

Landfill management, we considered for each colony if it had had an open landfill within a radius of 10, 20, 30, 40 or 50 km each year (spanning from July of year t-1 to June of year t). If one landfill was closed along a given year, that year was considered to have an open landfill. Landfill management, therefore, was treated as a binary variable (0, closed; 1, open landfill) within an original matrix assuming that φ was time- and colony-dependent. Despite GOF tests did not indicate a statistically significant effect of age, we considered up to 3 groups of age, assuming that φ (or p) showed different values in first year birds (i.e., annual survival from hatching year to the next year), subadults (annual survival from the second year of life to the fourth one), and adults (annual survival in older birds). Colonies were lumped into categories of (1) colonies with negative *versus* stable or positive population trends (Izaro *vs.* rest of colonies), and (2) colonies with a higher dependence on marine prey

(Getaria), landfill food (Santa Clara, Ulia) or an intermediate contribution of these two types of prey (Izaro). All the alternative models were ranked based on their small-sample sized corrected Akaike values (AIC_c; White & Burnham 1999). Models with AICc values differing in less than 2 in relation to the top-ranked one (i.e., the one with the smallest AICc) were considered to create an averaged model with which to obtain the survival and recapture probability estimates.

To estimate a long-term population growth rate (**λ**) based on the previously assessed survival values, we built a 5×5 post-breeding population model as shown by Caswell (2001) (Eq. 1): where S_{FY} $S_{SY,}$ S_{TY} and S_{AD} are the apparent survival rate of first-, second-, third-year birds and adults, respectively, *F* is the mean number of females fledged in relation to each breeding female (it is calculated as half of mean clutch size) multiplied by breeding success (Hiraldo *et al.* 1996).

Eq. 1: $$\begin{bmatrix} 0 & 0 & 0 & FS_{fY} & FS_{AD} \\ S_{FY} & 0 & 0 & 0 & 0 \\ 0 & S_{SY} & 0 & 0 & 0 \\ 0 & 0 & S_{TY} & 0 & 0 \\ 0 & 0 & 0 & S_{FY} & S_{AD} \end{bmatrix},$$

The mean clutch size (2.8 number eggs/nest) and the breeding success (0.60) were inferred from one of our colonies (Ulia), where intense field work to estimate these basic breeding parameters was carried out in 2018 and 2019 (S. Delgado, unpubl. data). Our population models were run using the library for R 'PopBio' (Stubben & Milligan 2007). Positive **λ** values indicate population increase; negative values, population decrease, and when **λ** = 1 the population is stable.

RESULTS

Overall, we ringed 4437 chicks (Table 12): Ulia, 2158 chicks (48.7%); Izaro, 990 (22.3%); Santa Clara, 686 (15.4%); Getaria, 603 (13.6%). These ringed birds provided 2245 resightings along the study period. The proportion of individual birds that were seen at least once after they left the colony ranged from 18.34% (Izaro) to 27.10% (Ulia; for further details see Table 12).

We obtained a total of three top-ranked models differing in less than 2 AICc values in relation to the first one (Table 13). These models shared that the apparent survival was influenced by landfill management at a distance of less than 10 km from the colonies and by the age of the individuals (first year birds, immature and adults). For the colonies of Ulia, Santa Clara and Getaria there was just one landfill within a radius of 10 km from each colony (S. Marcos for the first two colonies; Urteta for Getaria). Izaro, however, did not have a landfill within such buffer area (Supporting Information 2). Moreover, the first model included an effect of the main prey consumed within each colony on survival, whilst the second one included an effect of population trends on survival. The third model included an effect of colony on survival. The parameter estimates of these last effects, however, had an associated 95% confidence interval that included zero, so it can be concluded that the effect of main prey consumed, population trends and colony on survival were statistically negligible. Apart from this, we also detected that survival was age-dependent; particularly it varied between first-year, immature and adult birds (Table 13), increasing gradually from the first age category to adults (Fig. 7).

Regarding *p*, the best ranked models considered time-dependence on this parameter, ranging between 0.21 (in 2012) to 0.47 (in 2011; for further details see Supporting Information 4). Given the survival values obtained for each colony after model-averaging (for details see Table 13), we did not detect a remarkable effect of landfill closures on assessed population growth rate, except for the Izaro colony, where we would obtain an annual decreasing rate of 7% (Table 14). For the colonies of Ulia and Santa Clara, models predicted a decrease of 0.5 (5%) and 0.2, respectively, which still was not as high as to generate a negative growth rate in Ulia (which would reach a value of 1.02, i.e. a positive annual growth rate of 2%). In Santa Clara, that growth rate would fall up to 0.98 (i.e., a negative annual growth rate of 2%). In Getaria, interestingly, the growth rate even increased slightly after landfill closure, passing from 0.97 to 1.00.

Table 13. Ranking of the best models used to assess the effect of landfill management on the apparent survival (φ) of a resident Yellow-legged gull population. Survival varied between age classes (φFY, annual survival in first year birds, i.e., from hatching year to the next one; φIM, annual survival in immature birds; φAD, annual survival in adult birds), as well as in relation to landfill management (open/closed) at a distance of 10 km from each colony (10KM), colony (COLO), main prey consumed within a colony (DIET), demographic trends (DEMO). Other abbreviations: AICc, small sample sizes-corrected Akaike Information Criterion; ΔAICc, difference in AICc between each model and the first one; np, number of parameters.

Models	AICc	ΔAICc	AICc Weights	*np*	Deviance
Model 1:					
φ_{FY}(10KM+DIET), φ_{IM}(10KM+DIET), φ_{AD}(10KM+DIET) *p*(t)	13095.85	0.00	0.49	27	2881.93
Model 2:					
φ_{FY} (10KM+DEMO), φ_{IM}(10KM+ DEMO), φ_{AD}(10KM+ DEMO), *p*(t)	13096.97	1.12	0.28	22	2893.13
Model 3:					
φ_{FY}(10KM+COLO), φ_{IM}(10KM+COLO), φ_{AD}(10KM+COLO), *p*(t)	13097.31	1.46	0.23	32	2873.30
Other models:					
φ(time), *p* (time)	13618.52	425.49	0.00	25	3408.64
Φ, *p*	13810.84	617.81	0.00	2	3647.16

In these three colonies, however, if we consider the confidence interval associated to survival estimation, all estimates may fall within a range which would include 1, indicating that the size of these colonies would have remained stable independently of landfill management. In Izaro, however, we detected a substantial decline of this colony (95%, CI: 3-10%).

DISCUSSION

Using a data set of 4437 Yellow-legged Gull chicks ringed in four colonies over a period of 13 years, we obtained evidence supporting that the apparent survival was affected by landfill closure, especially if the landfill was located within a buffer of 10 km around the colony. Models considering an effect of landfill management at longer distances from the colonies had weaker support. These results suggest that the effect of landfills on the population dynamics of an opportunistic species like the Yellow-legged Gull is manifested within a relatively small geographic range.

Figure 7. Apparent survival estimation (± SE), obtained after model-averaging (Models 1 to 3 from Table 13) in relation to age, colony and the existence of an open/closed (i.e., before/after) landfill within a buffer of 10 km around each colony.

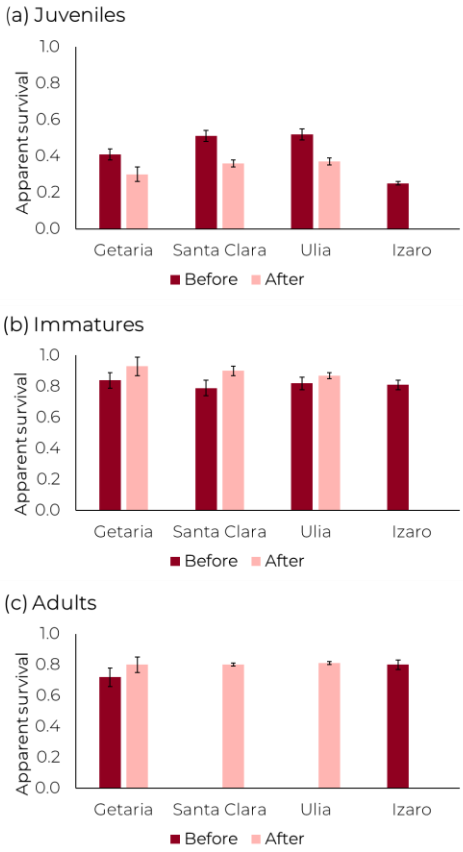


Table 14. Lambda values and 95% confidence interval (representing the global population trends) obtained for the fitted models including scenarios before/after the closure of open-air landfills at a distance of 10 km of less from the colonies.

	Ulia	Santa Clara	Getaria	Izaro
Before	1.07 (1.00, 1.09)	1.00 (0.95, 1.05)	0.97 (0.87, 1.00)	0.93 (0.90, 0.97)
After	1.02 (0.99, 1.03)	0.98 (0.93, 1.03)	1.00 (0.94, 1.04)	

Previous research within the region demonstrated that the proportion of landfill food in the diet had a strong relationship with the landfill-colony distance, indicating that the exploitation of a food resource is heterogeneous across the landscape, existing distance-dependency even at very small geographic scales (Zorrozua *et al.* 2020b).

When age classes were considered, we observed that, interestingly, landfill closure affected the apparent survival of first-year gulls (with a mean decrease of ca. 0.5 to 0.36), but not of older birds. Chicks were ringed at a mean age of ca. 20 days, so at least partially, this survival would include a period before fledging. Mortality by landfill management on first-year birds, therefore, may not be only associated to after-fledging survival. Also, we should consider survival between the ringing and fledging date may be impacted, if food shortage affects parents provisioning their offspring. It is true, in addition, that most landfills were closed during the first years and, therefore, we were not able to properly estimate the impact of landfill management on adult survival in some colonies (F7). Anyhow, the fall in survival estimation in adults would never be as high as in first-year birds, so it is clear that it is in this age class where the impact of landfill closure was greater. Likely, the experience and knowledge of the territory has a key role in this process. Our results suggest that (1) the carrying capacity of the region has probably reached its maximum, because landfill closure reveals a significant fall of the survival prospect of first-year gulls; (2) there might be alternative food (e.g. fish discards, etc.) (Arizaga *et al.* 2013, Zorrozua *et al.* 2020a,), available in limited amounts, hence this food would be exploited mostly by more experienced birds and those maybe being able to specialize in very particular feeding sources (Tyson *et al.* 2015, Van Donk *et al.* 2020). First-year gulls, however, would probably lack enough experience to exploit these scarcer, probably also less predictable resources, which might result in the observed decreasing survival prospect in this age class. Therefore, our results may support the Scenario 2 stated in the Introduction. To what extent this lower apparent survival rate was due to true mortality or to an increasing emigration rate is an aspect that deserves further studies. Previously, we observed that gulls increased their spatial range when landfills were closed (Arizaga *et al.* 2014b), so it is likely that a fraction of the observed lower apparent survival rates would indeed be caused by emigration from our survey region, but higher mortality cannot be excluded either. Detailed analyses on the territory use and spatial ecology of first-year birds will be crucial to disentangle the behaviour and determining the potential mortality of juvenile Yellow-legged Gull once they fledge and leave their natal colonies (Hake *et al.* 2003, Klaassen *et al.* 2014).

Gulls are long-lived species whose population dynamics are not so dependent on juvenile, first-year survival, but on the survival of adult birds (Newton 2013). As landfill closure did not seem to have a remarkable impact on adult survival rates, models estimating the population growth rate using these survival rates were unable to detect a clear fall in the growth rates after landfill closure.

The dynamics observed for the Izaro colony requires special mention. As compared to the other three colonies, Izaro did not have a landfill within a buffer of 10 km. This forced us to estimate a single survival value for the whole period considered in this work (for details see Supporting Information 2 and 3). Previous works, however, suggested a very high dependence of this colony to Jata (Galarza 2015), a landfill located at 15 km from Izaro. This colony, in addition, was by far the largest Yellow-legged Gull colony of the Basque coast, reaching a size of ca. 1300 adult breeding pairs in 2007 (Arizaga *et al.* 2009). The closure of Jata in 2013 triggered a cascade of effects on Izaro, where the productivity was found to crash and the number of adult breeding pairs in the colony was observed to decrease year after year (Galarza 2015); in 2013/2014, the size of the colony was 795 pairs (Galarza 2015), and in 2017, 409 pairs (A. Galarza, pers. comm.). In this particular case, even the Scenario 3 hypothesized in the Introduction may be also possible. The Scenario 2, anyhow, would be fully supported by the data: due to low productivity and also low juvenile survival, in an environment with a presumably very high competition for food. Very few birds may recruit in the colony and this may explain the steep decline found in the colony, at least in part.

Apart from landfill management, we also explored whether survival was influenced by aspects related to each colony, more particularly the degree of dependence on main feeding sources (trophic ecology), the colony size growth rate and the colony itself. Alternative models assuming in each case one of such factors were equally supported, so overall it can be concluded that apart from landfill closure apparent survival was also influenced by colony. The variance explained by the colony may probably respond to several factors that, overall, conform the particular characteristics of a colony, including its dependence on certain key feeding resources (e.g. landfill vs. fishing harbour), its dynamics and degree of density-dependent effects on such dynamics.

Landfills have a clear impact on animal populations able to exploit this food subsidy. We demonstrate that landfill closure affected the apparent survival of a resident Yellow-legged Gull population, especially by reducing survival in first-year individuals. This impact, in addition, seemed to have a relatively local influential range, suggesting strong landfill-colony distance relationships. Landfills can increase the survival prospect of gulls during the non-breeding season, especially if animals can move long distances to exploit this kind of food subsidies (Jordi *et al.* 2014). However, the dynamics of focal breeding colonies, and particularly the survival of first-year birds, was influenced by landfills at a rather local spatial scale. From a management standpoint, the closure of the open-air landfills existing within the region is having a positive effect on the surveyed population, in the sense that the carrying-capacity of the system is reduced and, probably through density-dependent processes. The population may return to a scenario which would reflect dynamics more typical of what would be a Yellow-legged Gull population in a more natural scenario without food subsidies of artificial origin. Note, however, that discarded fish is also a food subsidy that still remains within the system. In a nearby future, the only landfill still open will reduce the amount of food available to gulls up to zero. Under such scenario, we expect to find a steeper decrease of survival expectancy in young gulls, and also maybe in adult individuals. Overall, we expect that even those colonies which currently show stable trends will likely start to decline, probably suffering a process similar to the one observed at Izaro colony.

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ARTÍCULO 5

Models projections reveal negative growth rates for the first time ever recorded in a Yellow legged-gull population in the Bay of Biscay.

ABSTRACT

Many populations of opportunistic species have increased considerably in the last decades by feeding on food subsidies derived by human activities. Yellow-legged gull has been favoured by human-related food discards, but recent laws on sustainable management have cause local population declines. The aim of this study is evaluated possible change in the survival and the population trend over the last decades and provide population projection at short- and medium-term. We used observations on 3346 Yellow-legged Gull chicks ringed at three colonies over a period of 15 years in Northern Spain. The landfill closures in the region and partially the banned of fishing discards led to a gradual reduction in the annual survival probability especially in juveniles (from 53% to 23%), but also in older birds (from 86% to 78%). In the current scenario, the positive population growth observed in the last decades has reversed. The population trajectories hare expected to be affected by the reduction of the carrying capacity. An effective management of landfills closures which contain most of their feeding resources will allow to opportunistic species not to suffer a drastic population decline in a short time, hence the population regulates adequately.

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Predictable Anthropogenic Food Subsidies (PAFS, Oro *et al.* 2013), such as waste at open-air landfills of fishery discards, can have a very strong impact on the ecology of opportunistic species with cascading effects at individual and population, influencing the structure and functioning of entire ecosystems (Oro *et al.* 1995; 1996, González-Solís *et al.* 1997a, 2003; Votier *et al.* 2004; Hobson *et al.* 2015). Understanding how and to what extent PAFS affect wildlife is crucial to know and quantify the impact of human activities, toward a sustainable management of those ones that make subsidies available to wildlife. Evidence indicates that the successful exploitation of PAFS by animals improves their breeding investment and output (Bosch *et al.* 1994; Oro *et al.* 1995; Belant *et al.* 1998; Duhem *et al.* 2002; Tortosa *et al.* 2002; Steigerwald *et al.* 2015, Real *et al.* 2017), body condition (Auman *et al.* 2008, Steigerwald et al. 2015) or survival (Weiser & Powell 2010; Plaza & Lambertucci 2017). At the same time decreases dispersal (Arizaga *et al.* 2014b; Gilbert *et al.* 2016; Oro *et al.* 2008) and even promotes sedentary versus migratory strategies (Hebblewhite & Merrill 2011; Bonnet-Lebrun *et al.* 2020). PAFS have a direct positive effect on population dynamics (but see Carrete et al. 2006); Rideout *et al.* 2012), promoting the growth and even the geographic expansion (Duhem *et al.* 2008) but they can reduce individual breeding success in territorial species by promoting competitions (Oro *et al.* 2008). The role of anthropogenic food subsidies, and the establishment of open-air landfills in particular, has been extensively studied (Duhem *et al.* 2008). Much less interest, however, has been devoted to the effect of PAFS removal (Payo-Payo *et al.* 2015; Steigerwald *et al.* 2015; Real *et al.* 2018; Delgado *et al.* 2021b). In Europe, the ecology of many opportunistic birds, including gulls, raptors and storks among others (Plaza & Lambertucci 2017), is tight to open-air landfills and the managing of human-derived humid waste. However, the open-air landfills are progressively being closed or changed into incinerators plants following EU directives for sustainable waste management (Directives 1999/31/EU and 2008/98/CE).

Most large gulls (genus Larus) behave as opportunistic foragers (Duhem *et al.* 2003, Ceia *et al.* 2014, Steigerwald *et al.* 2015, Garthe *et al.* 2016). Their extensive use of open-air landfills makes them a good avian model with which to analyse the effects of PAFS on animal populations (Noreen & Sultan 2021). From a population dynamics perspective, and more particularly under a model building standpoint, it can be stated that landfill closure may affect the reproductive output (from laying to fledging) and/or after-fledging survival. This last, in addition, may affect one or more age categories (juveniles, immatures, adults). Subsequently, these parameters could be used to estimate population growth models in order to assess the sign and magnitude of food removal effects. More complex scenarios e.g. including some degree of emigration, could be also perfectly integrated in such models (emigration, indeed, is reflected as an increase of apparent survival rate) (Lebreton *et al.* 1992; Pradel *et al.* 1997).

Gulls are long-lived organisms which, therefore, prioritize survival in adults as compared to reproduction investment (Gaston 2004; Newton 2013). Depending on how fast the closure is and the availability of alternative feeding sources, several scenarios are possible: (1) impact on reproduction, but not on after-fledging survival, with no or weak impact on population growth rate, unless the impact on reproduction is strong (and the immigration rate is zero or close to zero, as shown in Delgado *et al.* 2021a); (2) impact on survival, with weak to strong impact on population growth rate, depending on the age and magnitude of this impact; (3) impact on reproduction and survival, with presumably strong impact on population growth rate. In other words, the impact of landfill closure on the population may be expected to progressively increase whether this closure would progressively affect only reproduction, only survival or both parameters.

The Yellow-legged gull (*L. michahellis*) is an opportunistic gull which was shown to experience very high population growth rates after the generalization of the open-air landfill sites since the decade of 1980 (Morais *et al.* 1998; Bosch *et al.* 2000; Skorka *et al.* 2005; Tavecchia *et al.* 2007; Duhem *et al.*

2008; Arizaga *et al.* 2009). In zones, most of its diet can be comprised by anthropogenic food waste taken from these landfill sites (Duhem *et al.* 2005; Neves *et al.* 2006; Ramos *et al.* 2009; Moreno *et al.* 2010; Arizaga *et al.* 2013a; Egunez *et al.* 2018, Catarina *et al.* 2021). The last census in Spain estimated ca. 125,000 adult breeding pairs of Yellow-legged gull (Molina 2009). The Basque coast along the south-eastern part of the Bay of Biscay hosts a Yellow-legged gull population of ca. 2,300 pairs (JA, unpubl. data). This population is resident (Arizaga *et al.* 2010; Arizaga *et al.* 2014b; Egunez *et al.* 2017), with very high philopatry and very small exchange rates among even very nearby colonies (Delgado *et al.* 2021a), and has a relatively high dependence on landfills (Arizaga *et al.* 2013; Egunez *et al.* 2017; Zorrozua *et al.* 2020a).

Since 2008, five landfills have been progressively closed within the region; in 2021, still three others remained open. The most recent studies carried out in the area show some signs that could indicate the starting of a possible population collapse. In this scenario of PAFS removal, (1) the population has declined in part of this region up to 46% in a period of 6 years (Galarza 2015), and (2) the first models based on survival estimation before and after landfill sites were closed reveal an impact on first-year birds, but not on adults, especially when the colony was at less than 10 km from a landfill (Delgado *et al.* 2021b). In parallel, we also have evidences hampering us to infer a clear negative population growth rate. Thus, some colonies seem to keep very stable (Arizaga *et al.* 2014a), and the overall population size today seems to have remained constant from 2017 (JA, unpubl. data). Also, we did not find strong evidence supporting a remarkable reproductive decrease (e.g., in terms of clutch size or hatching success) (Delgado *et al.* 2021c). Further research with a data set from individual gulls ringed as chicks during a period of 15 years and together with the test of different potential scenarios with effects of different magnitude on reproduction and survival may help to understand the clue mechanisms underlying the demographic consequences of landfill closure on this avian model population. In particular, we aimed to quantify to changes in which parameters would the population be more sensible, so closure effects on which parameters would have a faster impact on the population.

METHODS

Study area and data collection

The data used to feed up the models used in this paper were obtained from a total of three Yellow-legged gull breeding colonies situated in the Bay of Biscay, more in particular along the coast of Gipuzkoa (Basque Country, Spain): Getaria (43°18´N 02°12´W), Santa Clara (43°19´N 01°59´W) and Ulia (43°20´N 01°57´W). Colonies were located on the coast within a maximum distance of 20 km and the sizes of breeding colonies were approximately 165, 100 and 660 breeding pairs, respectively (last census 2017). These colonies represent 92% of the Yellow-legged gull population in Gipuzkoa (Arizaga *et al.* 2009).

The data sets used in this paper come from two main field procedures which were designed to estimate (1) survival, and (2) breeding output.

Survival was assessed using data based on a PVC-ringing program of chicks at the three colony sites mentioned above, and the subsequent search of these ringed birds elsewhere. Overall, 3645 chicks were ringed from 2005 (2006 in Getaria) to 2019. The ringing was carried out in late June, when chicks were around 20 days old. In order to reduce the impact inside the colonies, ringing was carried out in a single or a few days at each breeding colony and season. The chicks were ringed with both an official metallic ring (Aranzadi scheme) in one tarsus and a PVC ring with a code to be read at distance in the other tarsus (Fernández *et al.* 2017) (for details see Table 18).

To estimate the breeding output, we used a data set from the Ulia colony. Compared to other known, surveyed colonies (e.g., Isla de Tarifa or Delta del Ebro), our colonies are very difficult to sample due to the very accidental topography. Clutch size was deduced with the mean of data from Ulia colony during 2018-2020 breeding periods. The Yellow-legged gull chicks leave their nest very soon after hatching, and the attempts used previously to estimate productivity revealed to be inadequate or be very time-costly. Thus, we tried to build fenced sites with the idea of counting the pairs breeding there and, subsequently, the number of chicks alive before fledging (Delgado *et al.* 2016).

Table 18. Number of ringed chicks of Yellow-legged gull in the three study breeding colonies and the total number of individual birds seen after fledging (resight) each year during the months of April to June. Individuals seen in a year were ringed in the year before (i.e., resightings of first-year birds) or previously (older birds).

Year	Getaria	Santa Clara	Ulia	Total ringed	TotaL resight
2005	0	23	17	40	-
2006	30	69	147	246	1
2007	10	85	202	297	126
2008	38	55	208	301	79
2009	20	50	263	333	102
2010	59	42	221	322	225
2011	32	37	200	269	175
2012	61	86	178	325	397
2013	50	54	68	172	214
2014	50	59	151	260	335
2015	50	35	141	226	167
2016	54	40	163	257	212
2017	52	27	81	160	152
2018	49	52	170	271	427
2019	78	45	43	166	389
2020	-	-	-	-	345
Totals	633	759	2253	3645	3346

After this initial work, we compiled resighting data of 1855 (50.9%) individuals, producing 3342 resighting events over the study's period. These resightings were provided by birdwatchers during breeding periods (April, May and June) from 2005 to 2020 (16 years) for alive individuals (support information 1). These resightings were obtained mainly at the breeding colonies, feeding areas (landfills, harbours) and resting areas (rivers, intertidal flats, roof).

Data analyses

We used Cormarck-Jolly-Serber (CJS) models to estimate gull survival for different age classes. The program MARK (White & Burnham 1999) allows assessing survival (φ; it is the probability that an individual survived from t to t+1) and resighting probability (p probability that an individual seen at t and still alive in t+1 is seen at t+1) (Lebreton 1992). Prior to starting to run models the data set was tested for fitness to the CJS assumptions using U-CARE software (Chequel *et al.* 2009). We used for that a matrix of matrix of 16 columns (16 years) by 3645 rows (individuals) (for details see Supporting Information 5). Nota that the data used for each year were constrained to a period of 3 months, from April to June.

The global goodness-of-fit (GOF) on a CJS model was not significant (χ2 = 155.40, df = 149, P = 0.34), nor was the specific test to detect transients (Z = 0.800, P = 0.211) or trap-dependence (Z = 1.574, P =0.115). Model notation followed the one proposed by Lebreton *et al.* *(*1992) in which effects are noted in brackets after the parameter. We used ‘t’ for time, when we considered a different parameter for each year, ‘.’ to note a constant value (no effects) and ‘g’ to note a difference between colonies (groups). Our starting model in which all parameters were assumed to vary over time is noted φ(t*g), p(t+g).. Even though the GOF did not detect the presence of transients, we were interested in testing to what extent survival can vary across age classes. Yellow-legged gulls reach their sexual maturity in their 4th year of life (4Y) (Cramp & Simmons 1983) as a consequence we considered age from 1 to 4. In model notation we used xY for a parameter referring to age x, with x from 1 to 4+ year old. Age structure was modelled contrasting different groups of age classes. A model with only two age classes on survival is noted φ1Y(.)φ2+Y(.), one with three would be φ1Y(.)φ2Y(.)φ3+Y(.) and the one with a full age structure would be φ1Y(.)φ2Y(.)φ3Y(.)φ4+Y(.) A model in which two- and three-years old share the same parameters as suggested in Juez *et al.* (2015) will be noted φ1Y(.)φ2_3Y(.)φ4+(.). A possible lineal time trend was checked for different age categories (Table 19). All possible models were ranked following the AICc values (Burnham & Anderson 1998). The model with the lowest value should be considered as the best compromise between residual deviance and model complexity. Models differing less than 2 AICc points should be considered equivalent (Burnham & Anderson 1998).

Table 19. Ranking of CJS models used to assess the effect of colony, year and age categories on the annual survival rate (φ) of Yellow-legged gull population from northern Spain. Abbreviations: AICc, small-sample sizes corrected Akaike values; ΔAICc, difference in AICc between each model and the first one; np, number of parameters. Model notations: φ, constant survival; φ_{time} , time-dependence on survival, year as a factor; φ_{linear}, linear time effect on survival, i.e., linear trends on survival; φ(1Y/2Y/3Y or 4Y), survival varies in relation to age (from 1Y to 4Y; for details see Methods; 1Y = first-year birds… 4Y = birds older than three years.); φ(colony), survival varies for each colony site; p, resighting probability.

Model		AICc	ΔAICc	AICc weight	*Np*	Deviance
φ1Y(T), φ2_3_4+Y(t)	p_{time}	12400.02	0.00	0.76	19	2707.92
φ_{linear}(1Y), φ(2Y-3Y-4Y)	p_{time}	12402.39	2.38	0.23	18	2712.31
φ_{time}(1Y), φ(2Y-3Y-4Y)	p_{time}	12409.86	9.84	0.01	31	2693.55
Φ_{time}(1Y), φ_{time}(2Y-3Y-4Y)	p_{time}	12409.99	9.97	0.01	41	2673.41
φ(1Y), φ(2Y-3Y-4Y)	p_{time}(colony)	12426.87	26.86	0.00	45	2682.17
φ(1Y), φ(2Y), φ(3Y), φ(4Y)	p_{time}	12440.56	40.55	0.00	19	2748.47
φ(1Y), φ_{linear}(2Y-3Y-4Y)	p_{time}	12448.20	48.19	0.00	18	2758.12
φ(1Y), φ_{time}(2Y-3Y-4Y)	p_{time}	12452.00	51.99	0.00	28	2741.76
φ(1Y), φ(2Y-3Y), φ(4Y)	p_{time}	12459.02	59.01	0.00	18	2768.94
φ(1Y), φ(2Y-3Y-4Y)	p_{time}	12459.72	59.70	0.00	17	2771.65
φ(1Y, colony), φ(2Y-3Y-4Y, colony)	p_{time}	12461.49	61.47	0.00	21	2765.36
$\varphi_{time*colony}$	$p_{time*colony}$	12933.30	533.29	0.00	81	3114.95
φ_{time}	p_{time}	12973.75	573.73	0.00	29	3261.48
φ(colony)	p_{time}	12997.92	597.91	0.00	18	3307.84
φ	p_{time}	12998.56	598.55	0.00	16	3312.51
φ_{time}	p(colony)	13198.30	798.28	0.00	18	3508.22
φ_{time}	p	13203.38	803.37	0.00	16	3517.33
φ(colony)	p(colony)	13241.16	841.15	0.00	6	3575.19
φ	p(colony)	13251.14	851.13	0.00	4	3589.18
φ(colony)	p	13252.09	852.07	0.00	4	3590.13
φ	p	13253.74	853.73	0.00	2	3595.78

Population projections

To investigate the long-term effect of landfill closure we explore different scenarios of long-term population growth rate (λ) using a 5×5 post-breeding deterministic population model (Caswell 2001)

$$\text{(Eq.2) : } \begin{bmatrix} 0 & 0 & 0 & FS_{IM} & FS_{AD} \\ S_{JU} & 0 & 0 & 0 & 0 \\ 0 & S_{IM} & 0 & 0 & 0 \\ 0 & 0 & S_{IM} & 0 & 0 \\ 0 & 0 & 0 & S_{IM} & S_{AD} \end{bmatrix}$$

where SJU SIM and SAD is, respectively, annual survival of juvenile (1Y), immature (2Y and/or 3Y) and older birds (adults; 4+Y) as obtained from the best ranked model shown in Table 19; F is the average number of fledged female chicks per breeding female (half of mean clutch size) multiplied by the average breeding success (Hiraldo *et al*. 1996).

All such parameters were used to obtain a λ representing the actual population status. Sensibility and elasticity values were obtained for each age group. This model allowed us to calculate the parameter sensibility and elasticity (Caswell, 2001). To evaluate the response of this population to potential environmental changes, we recalculated λ by changing SJU, SIM, SAD and F (clutch size and breeding productivity) assuming different potential scenarios (Table 20), where at least one of such parameters was forced to decrease to biologically reasonable values (for details see Supporting information 5). Overall, the values were decreased by 5% in each step up to a maximum of 50%. Decreases of more than 50% were ignored because they were considered to be very unrealistic (Supporting information 5). In a second step, we built a model assuming some degree of environmental stochasticity. In particular, we forced survival to vary from a normal distribution truncated at 0 and 1 with a mean (±SD) as obtained from the top-ranked model from table 19, and for breeding success from a normal distribution truncated at 0.226 and 1.460.

Table 20. Scenarios considered for the effect of hypothetical reductions of feeding sources on the population growth rate of a Yellow-legged gull population. Each scenario assumes changes in one or more parameter estimates, affecting the breeding investment or/and after-fledging survival. Changes for the parameters ranged from -5% to -50% in relation to the actual values (model 0).

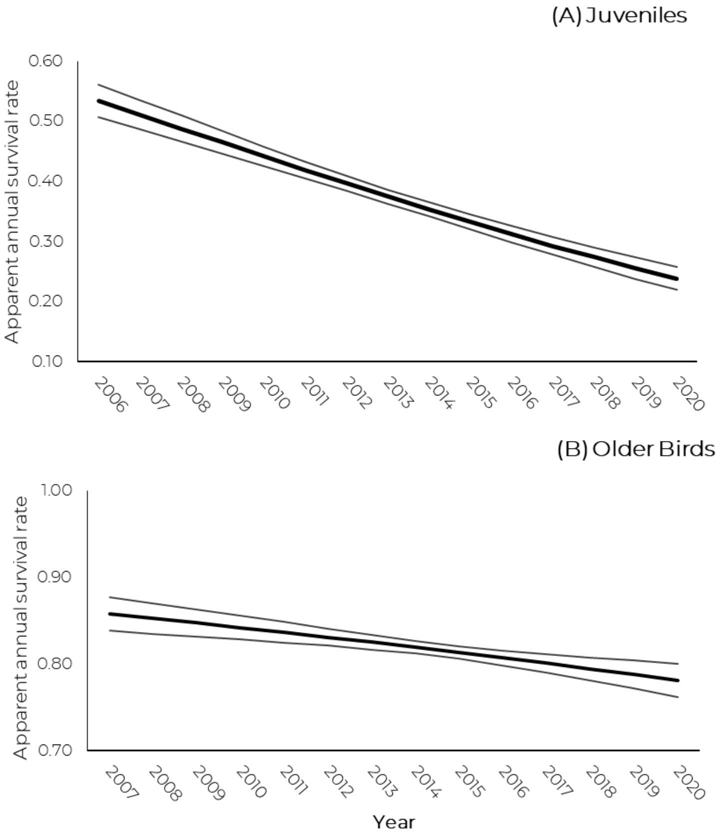
	Affected parameters:				
Population models - Environmental changes affect to:	Clutch size	Pre-fledging survival	1Y survival	2Y-3Y survival	≥4Y survival
0. Current scenario (no change)					
1. Breeding investment A	▨				
2. Breeding investment B		▨			
3. Survival A			▨		
4. Survival B				▨	
5. Survival A+B			▨	▨	
6. Survival C					▨
7. Breeding investment	▨	▨			
8. Breeding investment; Survival C	▨	▨			▨
9. Survival			▨	▨	▨
10. Breeding investment B + Survival		▨	▨	▨	▨
11. Breeding investment + Survival	▨	▨	▨	▨	▨

We simulated 1000 populations, each with a random value of these two parameters, and then calculated the mean and variance of lambda (population growth rate). Statistical analyses were done using popbio package (Stubben & Milligan, 2007) and truncnorm package (Geweke, 1991) in RStudio 1.2.5 software (RStudio, 2019).

RESULTS

The first ranked model (Table 19) included different survival values for two age categories, with also a liner effect of time on survival (i.e., linear trends on survival), without any obvious (significant) colony effect. Survival values for each age category declined over time, though the effect was particularly clear for the first-year fraction of gulls (Fig. 8). Survival values for older than one-year individuals showed just a slight decrease, with a high overlap of the 95% confidence interval between the extreme survival estimations. Thus, survival values (±95% CI) in first-year gulls ranged from 0.54 ± 0.03 in 2006 to 0.24 ± 0.02 in 2020. In older birds, these values ranged from 0.86 ±0.02 to 0.78 ± 0.02 (Fig. 1).

Figure 8. Apparent annual survival estimation (± 95% confidence interval) of first-year gulls (A) and older gulls (B) of the Yellow-legged gull in the Gipuzkoa colonies during the period of 2006 to 2020. Linear trends obtained from the top-ranked model in Table 19.



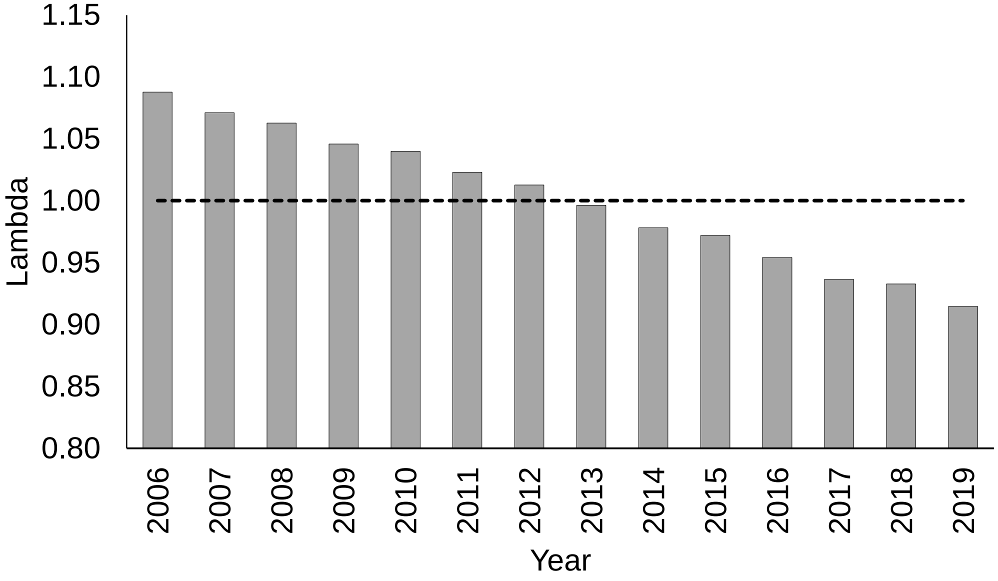
Taking survival values obtained in 2020 (Fig. 8), which would better fit with the actual scenario, we obtain a population decrease rate of 0.913. If we go to a more conservative scenario, taking survival values in 2013 (middle of time series in Fig. 8), this growth rate would rise to 1.006. Deterministic lambda estimation in each year in relation to survival values from Fig. 8 shows that the population may pass from a positive to a negative growth rate in 2012 (Fig. 9). The stochastic approach provided a mean lambda of 1.018 with an 95% confidence interval ranging between 0.97 and 1.06. Sensibility and elasticity values showed that the parameter with a highest effect on the population growth rate was survival in adult birds (Table 21).

Table 21. Perturbation analysis with the sensitivity and elasticity for different age classes obtained for our gull population. Survival notation as in Table 20.

Parameters	Estimates	Sensibility	Elasticity
Clutch size	2.78		
Breeding success	0.65		
Survival A	0.38	0.31	0.12
Survival B	0.82	0.12	0.10
Survival C	0.82	0.52	0.42

To go further we now considered the survival values obtained in 2013 (Fig. 9) as a starting scenario (i.e., first-year gull, 0.38 ± 0.01; older gulls, 0.82 ± 0.006), and then forced parameters from Table 20 to decrease from -5% to -50%. Overall, lambda values fell up to more than a 50% in relation to the original lambda in the worst scenarios which affect both breeding parameters and survival in all age categories (Fig. 10). A detailed look at Fig. 10 revealed that not all parameters had effects of similar magnitude on lambda. Overall, even high decreases (of up to -50%) in breeding investment or survival in non-adult birds had small effects on lambda (scenarios 1 to 5, of up to -11.8%, equating to a lambda = 0.88) (Fig. 10). By contrast, changes in adult survival would show a much stronger impact of lambda, even with the rest of parameters remaining unchanged (with reductions of up to -31%, equating to a lambda = 0.69). The worst scenario which would consider both reductions of a 50% in the breeding and all survival parameters, would result in a decrease of a 56%, equating to a lambda of 0.44 (Fig. 10; see also for details Supporting information 6).

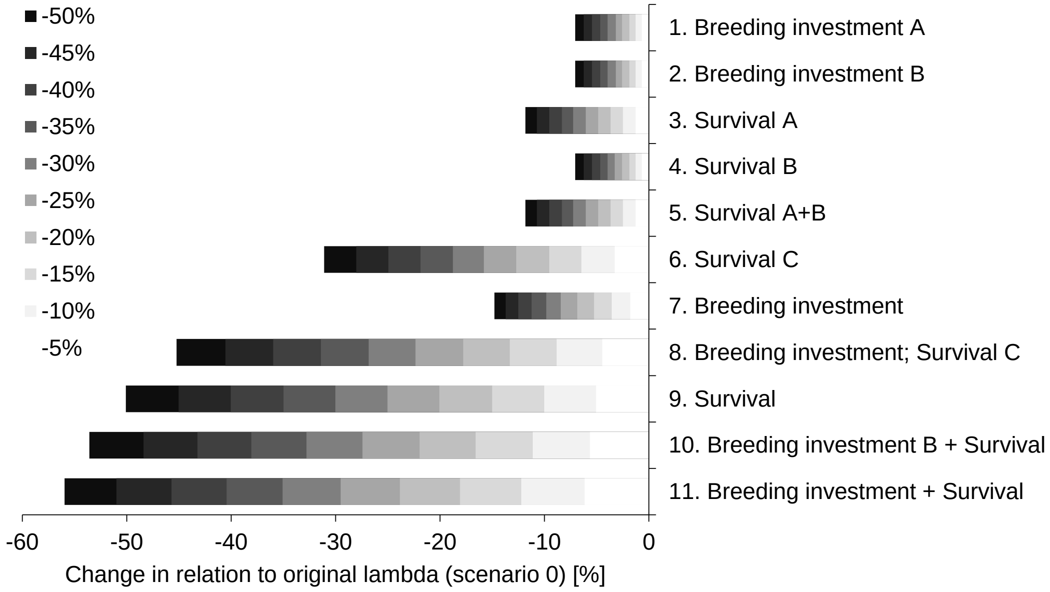
Figure 9. Deterministic population growth rate (lambda) of a Yellow-legged gull population where apparent annual survival rates varied annually as shown in Fig. 1. The dashed line shows the point where the population growth rate is zero; values above that point indicate a positive growth rate, while values below that point indicate a negative population growth rate.



DISCUSSION

This is the first study where the population growth rate of a Yellow-legged gull population within the Bay of Biscay is modelled assuming different scenarios where, overall, descends of both/either breeding investment and survival are considered. Such decreases are justified under a scenario of a general descend in the availability of food subsidies due to the closure of open-air landfills within the region.

Figure 3. Maximum absolute lambda values for each model developed when one or more variables decrease up to 50%. Zero values show the current situation and values decrease in relation it.



Survival models revealed that the population had two main after-fledging annual survival values: one for the first-year birds and another one for older birds. It is true that the first one was assessed for the time interval spanning from ringing date (when chicks were still within the colony) to June of the subsequent year. So, this survival value would also include some pre-fledging mortality (Delgado & Arizaga 2017). Anyhow, we observed that apparent annual survival in first-year birds tended to decrease more deeply than for older birds along the study period, dropping from a mean rate slightly higher than 0.5 in 2006 to less than 0.25 in 2020. This result suggests that it is this age group the one suffering more strongly a presumably decrease of food resources within the system. This apparently decrease in food from landfill is confirmed by trophic ecology studies (Arizaga *et al.* 2018; Zorrozua *et al.* 2020a). Older gulls, however, did not seem to experience such a strong decrease in survival, maybe because these individuals have a better knowledge of the environment and are able to compensate for such a decrease by exploiting other resources which could be new in the system and/or that were less used until then. Individual specialization in using resources which are scarce but that can be abundant at a fine-scale level could be part of a process that would allow older gulls to resolve their previous dependence on landfills (Van Donk *et al.* 2020). In this scenario, the use of the urban areas has been appealed to be an increasingly used habitat for the species (Méndez *et al.* 2020). As compared to other zones from the Mediterranean and even along the coast of northern Iberia, however, our surveyed population from the Basque coast seems to use the urban habitat still relatively marginally, at least as revealed by GPS tracking data (Arizaga *et al.* 2017; Arizaga *et al.* 2018; Zorrozua *et al.* 2020c).

According to our models the population started to decline in 2013, with an annual negative population growth rate of -8.6% in 2020. The census of the colonies run in 2017 and 2021 showed a stable population, with ca. 1,100 adult breeding pairs (SD and JA, unpubl. data), which would contradict our previsions. It is true, however, that this census only counts adult birds, thus ignoring the youngest fraction of the population. It is this fraction, indeed, the one where the decrease in survival is stronger (Fig. 9), and where the decline may be more apparent. Specific counts of these subadult birds are, however, lacking for the species within the region, so we have no way to test whether this part of the population may already be in decline. Whether this negative trend would continue, however, our prediction is that we should soon detect declines in colonies, due to the lack of a sufficient amount of recruits, especially if we take into account that our population has a very high philopatry rate (Delgado *et al.* 2021a). Furthermore, we must also consider that our lambda estimation was deterministic but, of course, some degree of stochasticity is plausible (Garcias & Tavecchia 2018). Thus, given the still closeness of lambda values to 1, we cannot reject that our models may still fit with a stable population, at least for the values provided around 2013.

Due to the fact that in the near future new closures of landfills will happen and the management of the few remaining ones will be also improved (with tools to deter gulls from accessing to these sites), the most reliable scenario is one where the population parameters such as breeding investment, breeding output and survival will decrease to a more or less extent. After testing for several potential scenarios where these parameters were forced to decrease from -5% to -50% in relation to the actual observed values, we obtained that the most critical parameter estimate was survival in adult birds. This is also corroborated by the elasticity estimation from population growth models. Models show, therefore, that our population was particularly sensible to changes in adult survival which is something expectable in a k-strategy bird species like the Yellow-legged gull is (Gaston 2004). Rapid declines in the population may be expected, therefore, only or if together with other parameters adult survival is affected. Our survival models reveal a slight, though continuous decrease of this parameter along the study period. This decline could be still more pronounced in zones with dense colonies, where due to density-dependent processes (Newton 2013), adults could show high mortality, as suggested in a nearby colony (Galarza 2015). Apart from survival, a decrease in the breeding investment is also expected. Recent studies carried out in the Ulia colony reveal a decrease in clutch size during the last three years (Delgado *et al.* 2021C). Therefore, the most possible demographic scenario is one in which both survival and breeding parameters will be compromised by the actual and expected decrease of food availability in the system. In such circumstances, we expect a population decline through the next years.

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SUPPORTING INFORMATION

Supporting Information 1.

A simple population model to check whether our population was stable, decreasing or increasing (i.e., showed a lambda, λ, equal, below or above 1). With that goal we used the package for R ‘popbio’ (Stubben and Milligan 2007), that calculates a λ value for given values of survival and productivity. In this case, we used the apparent survival values obtained in this work for both first-year birds and older age categories (0.27 and 0.86, respectively). Productivity was obtained for the colony of Ulia (SD, unpubl. Data), and it was calculated to be 0.91 ((2.05 hatching eggs/2)*0.65 survival before ringed). The script built for the model considered four age categories (the first age of reproduction was considered to be at the 4th year of life).

```
breeding=c(0,0,0.91,0.91) #each female produces 0.91 females, and the first age of reproduction is year 4th.
survival=c(0.27,0.86,0.86,0.86) #pre-breeding census
B1=breeding[1]
B2=breeding[2]
B3=breeding[3]
B4=breeding[4]
B5=breeding[4]
S1=survival[1]
S2=survival[2]
S3=survival[3]
S4=survival[4]
S5=survival[4]
M=matrix(0,4,4)
M[1,]=c(0,0,0,S1*B4)
M[2,]=c(S2,0,0,0)
M[3,]=c(0,S3,0,0)
M[4,]=c(0,0,S4,S5)
eigen.analysis(M)$lambda
```

With a λ = 1.0111, it can be stated that the population was increasing at an annual rate of 1.11%. This is a moderate increase, which indeed might be close to zero if we consider the 95% confidence interval (± 0.08) associated to the assessed apparent survival values.

Supporting Information 2.

Distance (km) from each colony to each landfill, for those landfill sites situated at less than 100 km from the four sampling colonies in the Bay of Biscay.

Colony	Getaria	Ulia	Santa Clara	Izaro
S. Marcos	22	5	6	63
Urteta	7	19	15	45
Lapatz	14	29	25	44
Sasieta	28	38	35	56
Zaluaga	52	32	35	90
Jata	55	73	70	15
Igorre	58	74	74	30
Artigas	61	82	78	31

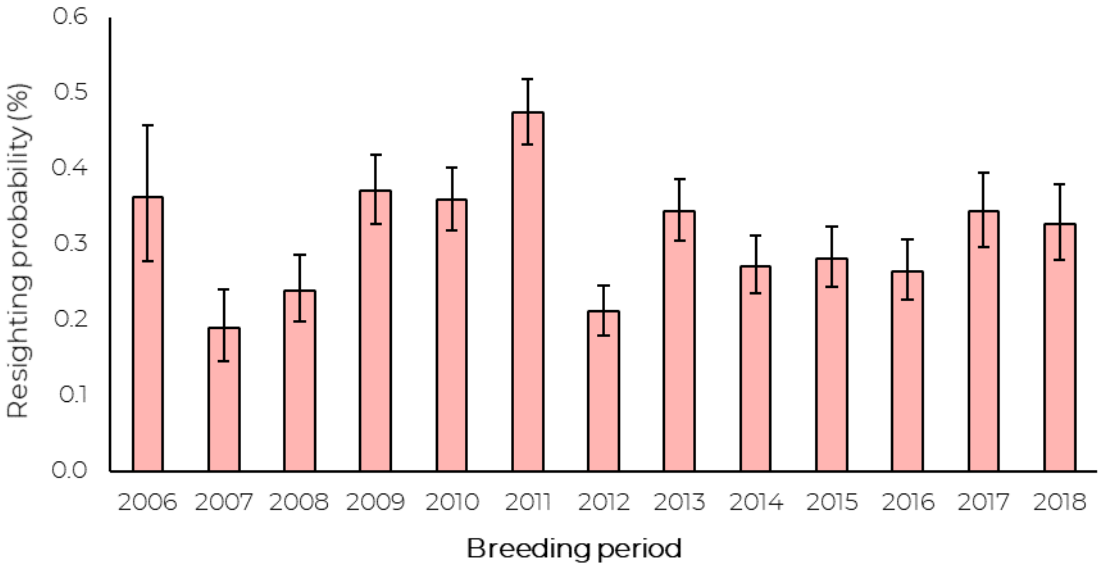
Supporting Information 3.

Landfill management (open, O/closed, C) along the study period. We also indicate the month in which the landfill was closed or open (if re-open). Note than some landfills were re-opened after being closed for a while. Artigas, Zaluaga remained always open.

Year	S. Marcos	Jata	Urteta	Sasieta	Lapatz	Igorre
2006	O	O	O	O	O	O
2007	O	O	O	O	O	O
2008	C(Oct)	O	O	O	O	O
2009	C	O	O	O	O	O
2010	C	O	O	O	O	O
2011	C	O	O	O	O	O
2012	C	O	O	O	O	O
2013	C	C(Dec)	O	O	O	C(Dec)
2014	C	C	C(Dec)	O	O	C
2015	C	C	C	C(Jan)	O	C
2016	C	O(Feb)	C	C	C(Dec)	C
2017	C	O	C	C	C	C
2018	C	O	C	C	C	C

Supporting Information 4.

Annual variation of the resighting probability (± 95% CI) obtained after model averaging (models 1 to 3 from Table 2) of a Yellow-legged Gull population in the Bay of Biscay.



Supporting Information 5.

Demographic parameters referenced in several bibliographic sources, relative to the Yellow-legged gull or other white-headed Larus gulls, used to roughly assess the observed biological range within which these parameters can vary as compared to the ones seen in our survey colonies in Gipuzkoa.

Variable	This work	Other studies (range)	References
Clutch size	2.78	2.14-2.90	Bosh &Sol 1998, Baaloudj *et al.*, 2014, Hammouda *et al.*, 2016.
Pre-fledging survival	0.65	0.51-0.79	Bosh & Sol 1998, Baaloudj *et al.*, 2014.
1Y survival	0.38	0.28-0.83	Chabrzyk & Coulson 1976, Juez *et al.*, 2015 Bosman *et al.*, 2016.
2Y-3Y survival	0.82	0.63-0.80	Chabrzyk & Coulson 1976, Reid 1988, Bosman *et al.*, 2016 Kralj *et al.*, 2018.
≥4Y survival	0.82	0.59-0.94	Chabrzyk & Coulson, 1976, Coulson & Butterfield 1985, Camphuysen & Gronert 2012, Peter & Vaughan 2013, Bosman et al., 2016, Kralj *et al.*, 2018.

Supporting Information 6.

Deterministic lambda estimation when population parameters (breeding, survival) changed according to different scenarios (for details see also Table 20).

Decrease	1	2	3	4	5	6	7	8	9	10	11
0%	1.006	1.006	1.006	1.006	1.006	1.006	1.006	1.006	1.006	1.006	1.006
-5%	0.999	0.999	0.993	0.999	0.993	0.973	0.988	0.961	0.955	0.949	0.944
-10%	0.993	0.993	0.981	0.993	0.981	0.941	0.970	0.917	0.905	0.894	0.883
-15%	0.987	0.987	0.969	0.987	0.969	0.91	0.953	0.872	0.855	0.839	0.824
-20%	0.980	0.980	0.957	0.980	0.957	0.878	0.937	0.827	0.804	0.785	0.766
-25%	0.974	0.974	0.945	0.973	0.945	0.847	0.921	0.781	0.754	0.730	0.709
-30%	0.966	0.966	0.933	0.966	0.933	0.817	0.907	0.736	0.704	0.676	0.653
-35%	0.959	0.959	0.922	0.959	0.922	0.786	0.893	0.69	0.654	0.623	0.599
-40%	0.951	0.951	0.910	0.951	0.91	0.755	0.88	0.644	0.603	0.571	0.546
-45%	0.943	0.943	0.898	0.943	0.898	0.724	0.868	0.598	0.553	0.519	0.493
-50%	0.935	0.935	0.887	0.935	0.887	0.693	0.857	0.551	0.502	0.467	0.443

CONCLUSIONES

1. El tamaño de puesta, el tamaño de los huevos y el éxito de eclosión de los huevos de gaviota patiamarilla en la colonia de Ulia no depende del nivel de cobertura vegetal en el entorno del nido y sí muestra variaciones interanuales. Tales variaciones podrían mostrar un ligero descenso en la inversión reproductiva, tal vez asociado al cierre de vertederos, si bien es pronto para confirmar tal conclusión.

2. La distancia de dispersión de la población estudiada no varía significativamente entre sexos y sí entre clases de edad y a lo largo del año. Tal distancia es máxima durante el primer año de vida y en los meses que suceden al final de la época reproductora.

3. La supervivencia tampoco varía entre sexos para ninguna de las clases de edad analizadas.

4. Los valores de filopatría son muy altos (99%) en cualquiera de las tres colonias estudiadas en Gipuzkoa, lo que sugiere un flujo mínimo de individuos entre colonias y cierto aislamiento reproductivo entre estas. Esto podría ser debido a que los individuos de cada colonia dependen de recursos tróficos muy concretos (o de zonas de alimentación específicas) y evitan la competencia con los ejemplares de otras colonias.

5. El valor de supervivencia anual de los ejemplares juveniles se estima en 0.27 ± 0.02 y no varía sustancialmente entre las tres colonias de Gipuzkoa. Este valor se incrementa considerablemente (0.82 ± 0.01) a partir del primer año de vida y no varía significativamente a partir de entonces.

6. Los individuos juveniles son los más afectados cuando se clausura un vertedero que se localiza a menos de 10 km de la colonia reproductiva,

puesto que su supervivencia anual desciende considerablemente (de 0.50 a 0.36). En el caso de inmaduros y adultos, la supervivencia se mantiene similar una vez que se produce dicho cierre. Las colonias dependientes de vertederos una vez que estos se clausuran sufren una disminución en el tamaño poblacional de hasta un 7% anual.

7. Los valores de supervivencia de las poblaciones de gaviotas patiamarrillas del Pais vasco y con ello el valor de tendencia poblacional se ha reducido a lo largo de estos últimos años. Adultos e inmaduros han reducido su supervivencia de valores iniciales de 0.86 a 0.78. En los juveniles esta disminución ha pasado del 0.53 al 0.23 en 15 años. La tendencia poblacional actualmente presenta unos valores de estabilidad, pero diferentes modelos futuros realizados ponen de manifiesto que una pequeña disminución en las diferentes supervivencias o valores reproductivos provocara una disminución de las diferentes poblaciones.

PRODUCCIÓN CIENTÍFICA Y FORMACIÓN

En este apartado se detalla todo el material científico elaborado durante el periodo que ha durado la tesis doctoral (2018-2021) y la diferente formación científica desarrollada:

1. Publicaciones científicas

Primer autor

Delgado, S., Zorrozua, N., & Arizaga, J. (2020). Marginal presence of plastic in nests of Yellow–legged gulls (*Larus michahellis*) in the southeastern Bay of Biscay. *Animal Biodiversity and Conservation*, *43*(2), 191-195.

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2. Conferencias

Asistencia al II Congreso de aves de Euskadi en Vitoria-Gasteiz. Febrero - 2019

“¿Influye el sexo en la distancia de dispersión de poblaciones sedentarias de gaviotas patiamarillas?".

Participación en las Jornadas Doctorales del G9. Abril - 2019.

"¿Cómo afecta el cierre de vertederos en los parámetros demográficos de la gaviota patiamarilla en el País Vasco?".

Participación en las jornadas de Pint of Science. Mayo-2019.

“Hazte con todos”

Asistencia y participación en el International Gull Meeting realizado en Donostia . Noviembre – 2019.

"Recruitment rates of Yellow-legged gulls in the southeastern part of the Bay of Biscay".

Asistencia al Simposio Internacional "Fauna y edificios, manejo y conservación" en Vitoria. Febrero - 2018.

3. Proyectos de colaboración

Participación en las diferentes campañas EMAN desarrolladas a lo largo de los años 2018-2021.

Promotor de las campañas de anillamiento de Golondrina común *Hirundo rustica* a lo largo de los años 2019-2021.

Responsable de la campaña de anillamiento anual en Zorroaga.

Gestor del seminario de diferenciación de gaviotas argénteas llevado a cabo en Zarautz en el año 2018.

Responsable del proyecto de anillamiento de gaviotas invernantes de País Vasco durante las temporadas 2019-2021.

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