



# Model projections reveal a recent decrease in a yellow-legged gull population after landfill closure

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## Abstract

The food available in open-air landfills, one of the most common predictable anthropogenic food subsidies (PAFS), can have a profound impact on animal biodiversity. Understanding how and to what extent PAFS affect wildlife is crucial for a sustainable management of resources. Most large gulls behave as opportunistic foragers and constitute a good avian model to analyze the effect of PAFS reduction on animal populations. Using individual data from a yellow-legged gull population of the Basque coast (northern Iberia) collected over a 15-year period, we estimated survival and reproductive parameters and used them to parameterize an age-structured population model to explore the effects of the local landfill closure. Local survival probability declined with time as a consequence of the progressive closure of the local landfill sites. The top-ranked models included a quadratic function of time, suggesting an acceleration of mortality during the later years, especially in juveniles, while survival in adults was linear. An effect more pronounced in first year birds than in older birds. Population models predict a decrease of the population and confirmed a greater sensitivity of the population growth rate to adult survival probability. Overall, our results suggest that the reduced carrying capacity of the system resulted after landfill closures have caused a population decline which is expected to continue in the near future.

**Keywords** Demography · Breeding output · Food subsidies · Population dynamics · Landfill · Seabirds · Survival

## Introduction

Predictable anthropogenic food subsidies (PAFS, Oro et al. 2013), like the food available in open-air landfills, can have an impact on local biodiversity, from individual to population levels, and on the structure and functioning of entire ecosystems (Oro et al. 1995; González-Solís et al. 1997; 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 activity toward a sustainable management of resources. The exploitation of PAFS by opportunistic animals improves their breeding output through an increase in breeding investment, chicks growth rates, hatching success (Bosch et al. 1994; Oro et al. 1995; Belant et al. 1998; Duhem et al. 2002; Tortosa et al. 2002;

Steigerwald et al. 2015), body condition (Auman et al. 2008), or survival (Weiser and Powell 2010; Plaza and Lambertucci 2017). Predictable food sources also contribute to reduce dispersal (Arizaga et al. 2014b; Gilbert et al. 2016) or migration (Hebblewhite and Merrill 2011; Bonnet-Lebrun et al. 2020). In some cases, the general positive effect on population growth (Rideout et al. 2012) promotes geographic expansion (Duhem et al. 2008). The role of PAFS worldwide has been studied extensively (Duhem et al. 2008), but there is much less research on population response after PAFS removal (but see Payo-Payo et al. 2015; Steigerwald et al. 2015; Delgado et al. 2021b). In Europe, many opportunistic bird species, including several species of gulls, raptors, storks, or herons among several other (Plaza and Lambertucci 2017), depend, to different degrees, on landfills.

The recent EU directive on sustainable waste management establishes the closure of open-air landfill (theoretically, they should have closed by 2020; Directives 1999/31/EU and 2008/98/CE) and it is expected to have an impact on the ecology of landfill foragers. Most large gulls (genus *Larus*) behave as opportunistic foragers (Duhem et al. 2003; Ceia et al. 2014; Steigerwald et al.

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2015; Garthe et al. 2016) and they constitute a good avian model to analyze the effects of PAFS closure on animal populations (Noreen and Sultan 2021). Gulls are long-lived organisms that prioritize investment on adult survival to the one on reproductive output (Gaston 2004; Newton 2013). The closure of a landfill can lead to several possible scenarios with different demographic consequences: (1) it can impact only on gull reproductive output, which will have no or little consequence on population growth rate; (2) it can decrease survival, with a weak to strong impact on population growth rate, depending on the age-class affected; (3) it can simultaneously influence reproduction and survival. These three scenarios correspond to a progressively stronger impact on population dynamic.

In southern Europe the gull more closely linked to landfill is the yellow-legged gull (*L. michahellis*), a medium-large gull with an opportunistic diet. Since the 1980s, yellow-legged gull numbers increased all over the species distribution area due to the availability of food in open-air landfill (Morais et al. 1998; Bosch et al. 2000; Skorka et al. 2005; Tavecchia et al. 2007; Duhem et al. 2008; Arizaga et al. 2009) as shown by the large waste items in the diet (Duhem et al. 2005; Neves et al. 2006; Ramos et al. 2009; Moreno et al. 2010; Arizaga et al. 2013; Egunez et al. 2017; Lopes et al. 2021). In Spain, the last census at country level estimated ca. 125,000 adult breeding pairs (Molina 2009); however, the population has recently declined all around the country due to a progressive closure of open-air landfills (Oliveira et al. 2022). The south-eastern part of the Bay of Biscay, in northern Spain, hosts ca. 2300 pairs (Arizaga et al. 2022). This population made an extensive use of landfills (Arizaga et al. 2013; Egunez et al. 2017; Zorrozueta et al. 2020a) and in contrast to the Mediterranean populations, it is resident showing high philopatry and little exchange even among nearby colonies (Arizaga et al. 2010, 2014b; Egunez et al. 2017; Delgado et al. 2021a). Since 2008, five landfills have been progressively closed within the region with three remained open in 2021. Recent evidence shows signs of a population decrease in some colonies up to 46% in a 6-year period (Galarza 2015). Similarly, the annual survival probability of first-year birds, but not of adults, decreased in colonies located less than 10 km away from a landfill (Delgado et al. 2021b). In contrast, the population at regional level and the number of breeding pairs in other colonies seem to be stable or just (Arizaga et al. 2014a).

Here we used individual ringed-based data collected during a 15-year period to estimate survival and detailed information on breeding output to estimate which parameter and with which intensity is affected by landfill closure (for details see Appendix 1, Table 5). We subsequently build a population model to explore different scenarios considering effects of different magnitudes on reproduction and survival (considering different survival estimates for different age classes).

## Methods

### Study area and data collection

Parameters were estimated using data obtained at three yellow-legged gull colonies situated along the coast of Gipuzkoa region (Bay of Biscay, Basque Country, Spain): Getaria (43°18'N 02°12'W), Santa Clara (43°19'N 01°59'W), and Uliia (43°20'N 01°57'W). Colony sizes were approximately 165, 100, and 660 breeding pairs, respectively (last census 2017) and they represent about 92% of the yellow-legged gull population within the region (Arizaga et al. 2009). The maximum inter-colony distance was of 20 km.

Landfill management is detailed in Appendix 1, Table 5. According to previous knowledge (Arizaga et al. 2013, 2014b, 2018; Delgado et al. 2021b; Egunez et al. 2017), the species has been exploiting up to 6 landfill sites within the region where this study was carried out. Overall, the scenario has passed from years when these six landfills were open to years, especially during the last part of the temporal series, when practically all the landfills were closed.

The annual local survival probability of gulls was estimated through capture-mark-recapture models (Lebreton et al. 1992 see below) using observations of gulls ringed as chicks (~ 20 days old) at the three colonies. Overall, 3645 chicks were ringed from 2005 to 2019. Chicks were ringed with a metallic ring (Aranzadi scheme) and a PVC ring with an alphanumeric code to be read from the distance (Fernández et al. 2017) (Table 1). Ethics approval was not required. We compiled 3342 resightings of 1855 (50.9%) individuals made by birdwatchers from April to June between 2005 and 2020 (Supporting Information 1). The majority of these observations were made at the breeding colonies and feeding or resting sites, e.g., landfills, harbors, rivers, intertidal flats, roofs, of the Gipuzkoa region.

Breeding output was difficult to estimate at each colony due to precocial habit of the species and the accidental topography of the sites. As a consequence, we used the data collected at Uliia colony during the period 2018–2020 when clutch size and hatching success (proportion of nests in which at least one egg hatched) were estimated at given zones of the colony in which all nests were monitored during the breeding season (Delgado et al. 2021c).

### Data analyses

We estimated age- and colony-dependent (1) apparent annual survival probability ( $\varphi$ ), which was the probability that an individual survived from  $t$  to  $t + 1$ , and (2) resighting probability ( $p$ ), which was the probability that

**Table 1** 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	Ulía	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
<b>Totals</b>	<b>633</b>	<b>759</b>	<b>2253</b>	<b>3645</b>	<b>3346</b>

an individual that survived from  $t$  to  $t + 1$  was seen in  $t + 1$ , using program MARK (White and Burnham 1999).

We began the analysis by assessing the goodness of fit of a model in which all parameters were time and colony dependent using the software U-CARE (Choquet et al.

2009; Appendix 2, Table 6). The global goodness of fit (GOF) on this model was not significant ( $\chi^2_{149} = 155.40$ ,  $P = 0.340$ ), nor were the directional tests specifically designed to detect the presence of transients ( $Z = 0.800$ ,  $P = 0.211$ ) and trap dependence ( $Z = 1.574$ ,  $P = 0.115$ ; see Choquet et al. 2009 for details). We included a two-age structure (first-year birds and older birds) in this general model, so that our starting model was considering that both  $\varphi$  and  $p$  varied among years and colonies and among first-year and older birds. Alternative nested models assumed different combinations of these effects.

Overall, we contrasted 14 models (Table 2) and ranked them using the Akaike’s information criterion corrected for the effective sample size (AICc (Burnham and Anderson 1998)). The model with the lowest AICc values was considered the best compromise between model deviance and complexity. Models differing by less than 2 AICc units should be considered equivalent (Burnham and Anderson 1998).

### Population projections

To investigate the long-term demographic effects (Peery and Henry 2010) of landfill closure, we explored different scenarios by using a  $5 \times 5$  post-breeding deterministic population model (Caswell 2001). The model assumes a population divided into 5 age classes and first reproduction at age 4 and includes the respective survival and fertility parameters (Eq. 1):

**Table 2** Ranking of models used to estimate annual local survival ( $\varphi$ ) and detection probability ( $p$ ) as a function of the age, time, and colony

Models		AICc	$\Delta AICc$	AICc weight	np	Deviance
$\varphi_{1Y}(\text{time.quad}), \varphi_{2Y}(\text{time})$	$p(\text{year} + \text{colony})$	12380.73	0.00	0.58	22	2682.59
$\varphi_{1Y}(\text{time.quad}) \times \varphi_{2Y}(\text{time.quad})$	$p(\text{year} + \text{colony})$	12382.72	2.00	0.22	23	2682.57
$\varphi_{1Y}(\text{time.quad}) + \varphi_{2Y}(\text{time.quad})$	$p(\text{year} + \text{colony})$	12384.27	3.54	0.10	21	2688.15
$\varphi_{1Y}(\text{year}), \varphi_{2Y}(\text{year})$	$p(\text{year} + \text{colony})$	12385.18	4.45	0.06	32	2666.85
$\varphi_{1Y}(\text{year}) \times \varphi_{2Y}(\text{year})$	$p_{1Y}(\text{year} + \text{colony}), p_{2Y}(\text{year} + \text{colony})$	12387.75	7.02	0.02	46	2641.01
$\varphi_{1Y}(\text{time}) \times \varphi_{2Y}(\text{time})$	$p(\text{year} + \text{colony})$	12388.83	8.10	0.01	21	2692.71
$\varphi_{1Y}(\text{year}) \times \varphi_{2Y}(\text{year})$	$p(\text{year} + \text{colony})$	12389.03	8.30	0.01	43	2648.40
$\varphi_{1Y}(\text{year}), \varphi_{2Y}$	$p(\text{year} + \text{colony})$	12399.36	18.63	0.00	33	2679.00
$\varphi_{1Y}(\text{year}) \times \varphi_{2Y}(\text{year})$	$p_{1Y}(\text{year} \times \text{colony}) \times p_{2Y}(\text{year} \times \text{colony})$	12,401.64	20.92	0.00	112	2519.11
$\varphi_{1Y}(\text{year}) \times \varphi_{2Y}(\text{year})$	$p_{1Y}(\text{year}) \times p_{2Y}(\text{year})$	12,411.61	30.88	0.00	56	2644.51
$\varphi_{1Y}, \varphi_{2Y}(\text{year})$	$p(\text{year} + \text{colony})$	12,443.60	62.87	0.00	30	2729.31
$\varphi_{1Y}(\text{year} \times \text{colony}) \times \varphi_{2Y}(\text{year} \times \text{colony})$	$p_{1Y}(\text{year}) \times p_{2Y}(\text{year})$	12,449.92	69.19	0.00	112	2567.38
$\varphi_{1Y}, \varphi_{2Y}$	$p(\text{year} + \text{colony})$	12,451.52	70.79	0.00	19	2759.43
$\varphi_{1Y}(\text{year} \times \text{colony}) \times \varphi_{2Y}(\text{year} \times \text{colony})$	$p_{1Y}(\text{year} \times \text{colony}) \times p_{2Y}(\text{year} \times \text{colony})$	12,454.40	73.67	0.00	163	2464.68

AICc small sample size-corrected Akaike values,  $\Delta AICc$  AICc difference in relation to the top-ranked model;  $np$ , number of parameters,  $1Y$  first-year birds,  $2Y$  older birds—birds in their second year of life or older,  $year$  year effect as a factor,  $time$  linear effect of year—trend,  $time.quad$  quadratic linear effect of year,  $\times$  interaction between factors/covariates,  $+$  additive effect

$$\begin{bmatrix} 0 & 0 & 0 & FS_{IMi} & FS_{ADi} \\ S_{JUi} & 0 & 0 & 0 & 0 \\ 0 & S_{IMi} & 0 & 0 & 0 \\ 0 & 0 & S_{IMi} & 0 & 0 \\ 0 & 0 & 0 & S_{IMi} & S_{ADi} \end{bmatrix} \quad (1)$$

In Eq. (1)  $S_{JUi}$ ,  $S_{IMi}$ , and  $S_{ADi}$  are, respectively, annual survival of first-year (1Y), immature (2Y and/or 3Y), and older birds (adults; 4Y/+4Y). The parameter  $F$  is the average number of fledged female chicks per breeding female. Assuming 1:1 sex-ratio at fledgling, this parameter is calculated as the average breeding success probability multiplied by the average clutch size and divided by 0.5 (Hiraldo et al. 1996).

The matrix of Eq. (1) is also called transition matrix because it allows to project the population from an initial state through future time steps. The maximum real eigenvalue of the matrix in Eq. (1) is the asymptotic population growth rate,  $\lambda_i$  (Caswell 2001). The relative importance of each parameter of Eq. (1) in influencing  $\lambda$  can be calculated through sensitivity and elasticity analyses (Caswell 2001). These quantities measure the change in  $\lambda$  corresponding to an absolute and a relative change of the parameter, respectively.

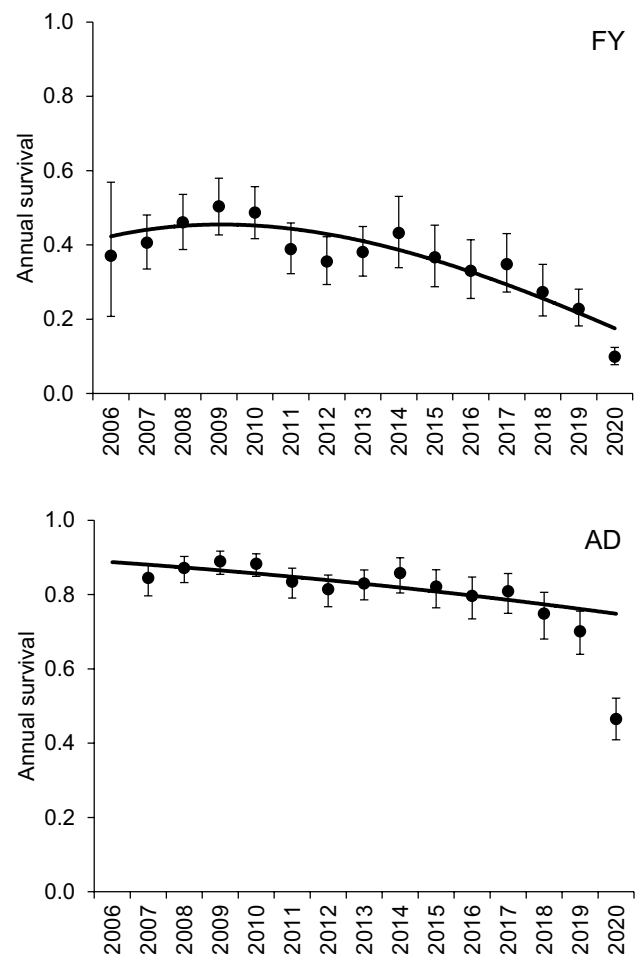
### The present and future of gull population

We first calculated  $\lambda$  related to a constant matrix, in which all parameters were taken by the model assuming a constant survival probabilities (Table 2). This would deliver an “average” population growth rate that although unrealistic will allow comparisons with other system. We included uncertainty in this model by considering survival estimates derived from a normal distribution truncated at 0 and 1. The mean and standard error of the distribution were set as the survival estimates and its standard error as estimated by the model. Similarly, breeding success followed a normal distribution with mean and standard error as those estimated from the data and truncated at 0.226 and 1.460 as indicated in Delgado et al. (2021a, b, c). By simulating 1000 population matrices, each with a random value of survival and breeding parameters, we obtained a distribution of  $\lambda$ . Statistical analyses were done using the packages “popbio” (Stubben and Milligan 2007) and “truncnorm” (Geweke 1991) in RStudio 1.2.5 software (RStudio Team 2019).

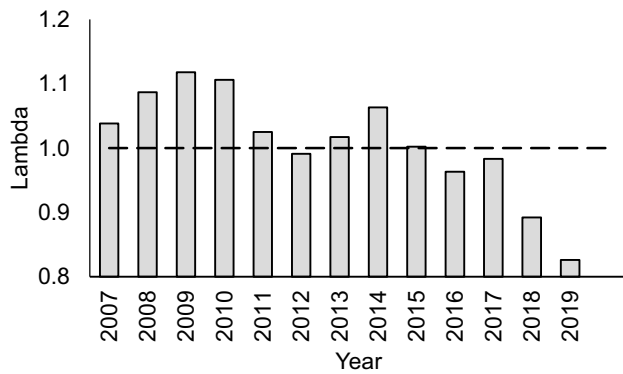
Subsequently, we calculated  $\lambda$  under different scenarios where one or more parameters in Eq. (1) ( $S_{JU}$ ,  $S_{IM}$ ,  $S_{AD}$ , and  $F$ ) were decreasing as suggested by the retained model (for details see Appendix 3, Table 7). To measure the expected changes in population growth rate, we used Eq. (1) with values of 0.38 and 0.83 for first-year and adult birds, respectively (these were the mean values of survival for the temporal series considered in this work), and force parameters to decrease from  $-5$  to  $-50\%$ .

## Results

The interaction terms between the effects of colony, age, and time were dropped at an early stage of the analysis (Table 2). Model selection theory indicated that resighting probability differed among colonies, while annual local survival did not. Both parameters changed over time (Table 2, Fig. 1). In particular, survival of first-year gulls ranged from 0.54 in 2006 to 0.24 in 2020. In older birds, these values ranged from 0.86 to 0.78 (Fig. 1). The first three ranked models included a quadratic function of time on survival, suggesting an acceleration of mortality during the last years for first-year birds but not for the older ones, with these last showing a linear relationship. Indeed, including a quadratic term for



**Fig. 1** Apparent annual survival estimation ( $\pm 95\%$  confidence interval) of first-year gulls (FY) and older gulls (AD) of the yellow-legged gull in the Gipuzkoa colonies during the period of 2006 to 2020, obtained from the fourth model shown in Table 2. Additionally, we add the linear trends obtained from the top-ranked model in Table 2, where a quadratic effect of year on survival is considered for first-year birds, but a linear one for older birds



**Fig. 2** 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 0 (lambda=1); values above that point indicate a positive growth rate, while values below that point indicate a negative population growth rate

adult birds did not produce a relevant change in model deviance (Table 2; Fig. 1).

The annual population growth rate from the matrix population model showed positive values up to 2015 (with the exception of 2012) and negative values from 2016 to 2019 (Fig. 2). The stochastic approach provided a mean population growth rate,  $\lambda$ , of 1.02 (95% CI: 0.97–1.06). Sensitivity and elasticity values showed that the parameter with the greater influence on population growth rate was survival of adult birds (Table 3).

Models showed that a change in breeding output, survival of subadult birds, or both parameters at the same time (scenarios 1–5 and 7 in Fig. 3) showed a weak impact on

lambda (max. –15%). However, even small changes in adult survival rates had a very high impact on lambda, resulting in a decrease of > 50% (Fig. 3).

### Discussion

In this research it is analyzed for the first time for the Bay of Biscay not only the temporal trend of survival of a yellow-legged gull population throughout a relative long time series (15 years) but also its impact on the population growth rate as provided by demographic models. The first chief finding of the study was that apparent survival rate tended to decrease along the study period, though differentially between age classes. Overall, this decrease in survival is compatible with a reduction of predictable anthropogenic food subsidies (PAFS), which would probably be to our knowledge the main demographic driver that more drastically has changed through the last years within the region. Though probably less likely (according to experience), other possible causes of a lowering survival cannot be rejected and then deserve some discussion. Reductions in fish discards or natural sources of food could have similar effects on survival if these feeding sources are relevant for the population. The estimation of fish consumption by our population can be relevant (>40% of the diet for some colonies), but it seems that these fish are taken not from discards offshore, but chiefly from fishery waste which is produced in harbors (Zorrozueta et al., in prep.). Fishery activity in the harbors did not change substantially during the period in which this study was carried out (Zorrozueta et al. 2020c). Other factors which could also affect survival like, e.g., a progressive change in the climate, remain obscure to us since

**Table 3** 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)

Population models - Environmental changes affect to:	Affected parameters:				
	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	■	■	■		



**Table 4** Perturbation analysis with the sensitivity and elasticity for different age classes obtained for our gull population. Sensitivity and elasticity in some parameters have two values due to have reference to new breeders' individuals and breeders. Survival notation as in Table 3

Parameters	Estimates	Sensitivity	Elasticity
Clutch size	2.78		
Breeding success	0.65		
Survival A	0.42	0.31	0.12
Survival B	0.83	0.12	0.10
Survival C	0.83	0.52	0.42

this type of studies are still in their infancy (Zorrozueta et al. 2020b), but, if existing, they probably should have a weaker effect as compared to a sudden closure of a landfill site.

Annual survival was decreasing much more markedly, following a quadratic function, in first-year birds. This suggests that young gulls are more affected than adults by a reduction of anthropogenic food subsidies. Older gulls would then be able to buffer, at least partially, the dietary change, possibly thanks to a better knowledge of the environment and of alternative food supplies (Van Donk et al. 2020). Indeed, direct observations in other areas suggest that gulls are increasingly visiting urban areas (Méndez et al. 2020), though the use of the urban habitat still remains relatively marginal in the Basque coast (Zorrozueta et al. 2020b). Survival of first-year gulls was lower than older birds, an expected result given that this value includes pre-fledging mortality referred to the period between ringing and full fledgling, but also the first weeks after fledging, when mortality still remains high (Delgado and Arizaga 2017).

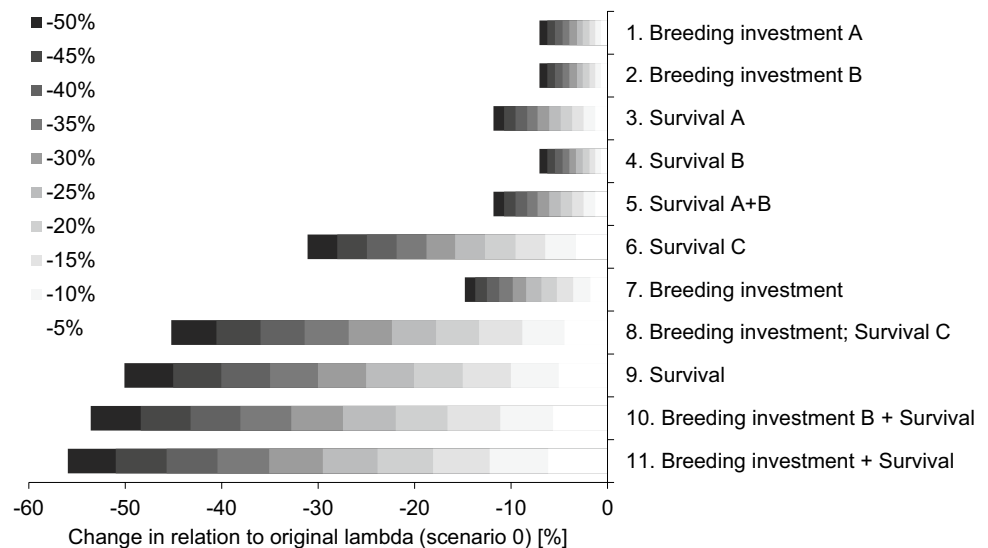
Models showed a clear decrease in survival since 2015. Although capture-mark-recapture models do not distinguish between mortality and permanent emigration, the

low survival during the last years of the study did likely corresponded to a real increase in mortality. As compared to other zones from the Mediterranean and even along the coast of northern Iberia, gulls inhabiting the Basque coasts are highly philopatric (Arizaga et al. 2017, 2018; Zorrozueta et al. 2020b) and their emigration might be considered to be virtually equal or close to 0. Following this, we should also mention that re-sightings from outside our study region are rare and did not increase recently (S. Delgado, unpubl. data).

This is the first study that provides population projections of the yellow-legged gull within the Bay of Biscay. Deterministic models indicate that the population is nowadays decreasing, a phenomenon which would be compatible with a reduction in the availability of predictable anthropogenic food subsidies (PAFS), landfills in particular, within the region. Recent evidence from studies on gulls' diet confirms the reduction of food of anthropogenic origin (Arizaga et al. 2018; Zorrozueta et al. 2020a). The census of the colonies run in 2017 and 2021 showed a stable population in this period, with ca. 1100 adult breeding pairs (Arizaga et al. 2022). Since 2000, however, the population has shown a clear decrease of ca. 6% (Arizaga et al. 2022). The stochastic population approach provided an annual growth rate of 1.02 (i.e., stability). Note, however, that this lambda estimation was obtained with the mean survival values for the entire temporal series (Fig. 1)—a conservative approach—and that survival tended to decrease during the later years within the series, especially for birds in their first year. Projections calculated by using time-dependent estimates suggested a clear decrease, which would be a theoretical estimation closer to the observed 6% decrease.

In the near future new closures of landfills are expected to occur. The most plausible scenario is a further decline in the breeding investment, the breeding output, and the survival probability. Recent studies carried out in the Uliia colony reveal a decrease in clutch size during the last

**Fig. 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



3 years (Delgado et al. 2021c). The population model shows a small elasticity to these parameters but a greater one to adult survival, something expected in long-lived species (Gaston 2004). This parameter is also decreasing in our population. Moreover, the observed decline could be more pronounced in denser colonies due to density-dependent processes (Newton 2013; Galarza 2015). Therefore, due to the decrease in survival and breeding parameters, we will expect a population decline through the next years. In this context landfill closure will probably return yellow-legged gull populations to demographic values (in terms of breeding output, survival, or population growth rate) closer to what we may expect for a “natural” scenario with no or little input of anthropogenic food subsidies. “No managing,” therefore, would probably be the best way to manage this species from a conservation standpoint. This probably entails avoiding any intervention (at least in natural colonies), either in the

direction of trying to reduce the breeding output or survival, but also in the lack of sense of trying to recover the very high values that were habitual when landfills were open.

In conclusion, we obtained evidence supporting a decrease in survival in a resident yellow-legged gull population from the Bay of Biscay (northern Iberia) during a 15-year temporal series (2005–2020). Although this decrease was much more marked in first-year birds than in older gulls, and in spite of the fact that the population was more sensible to a survival decrease in adults, model projections still show a negative population growth rate if we consider the survival values of the later years within the series. Such projections are compatible with the observed population change during the period 2000–2021. Likely, landfill closures are the main factor explaining this decline.

### Appendix 1

**Table 5** Management (open, O/closed, C) of those landfill sites situated within the home range area used by the studied population through the study period. We also indicate the month in which the landfill was closed or open (if re-open). Note that 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
2019	C	O	C	C	C	C
2020	C	O	C	C	C	C

### Appendix 2

**Table 6** 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	Bosch and Sol (1998); Baaloudj et al. (2014); Hammouda et al. (2016)
Pre-fledging survival	0.65	0.51–0.79	Bosch and Sol (1998); Baaloudj et al. (2014)
1Y survival	0.38	0.28–0.83	Chabrzyk and Coulson (1976); Juez et al. (2015); Bosman et al. (2016)
2Y–3Y survival	0.82	0.63–0.80	Chabrzyk and Coulson (1976); Reid (1988); Bosman et al. (2016); Kralj et al. (2018)
≥4Y survival	0.82	0.59–0.94	Chabrzyk and Coulson (1976); Coulson and Butterfield (1985); Camphuysen and Gronert (2012); Rock and Vaughan (2013); Bosman et al. (2016); Kralj et al. (2018)

## Appendix 3

**Table 7** Deterministic lambda estimation when population parameters (breeding, survival) changed according to different scenarios (for details see also Table 4)

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

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## Declarations

**Conflict of interest** The authors declare no competing interests.

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