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God did not save the kings: Environmental consequences of the 1982 Falklands War

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God did not save the kings: Environmental consequences of the 1982 Falklands War*

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Abstract

Warfare has been found to have detrimental impacts on biodiversity due to its long-lasting economic and social consequences. Yet, much less is known about the amount of biodiversity loss directly resulting from the use of military technology. This paper analyzes the environmental consequences of one of the largest aerial and naval conflicts of the late 20th century, namely the 1982 Falklands War. As an indicator of the marine ecosystem status, we analyze population trends of king penguins breeding on the Falkland Islands over the period 1963-1997. Using interrupted time series analysis, we find that the war significantly slowed the growth rate of king penguins' population. To take better account of time-varying confounders, we complement this analysis using a synthetic control group based on data from other Sub-Antarctic colonies and find similar results.

JEL codes: C22, F51, Q57.

Keywords: ecological warfare, Falklands War, king penguins, seabirds.

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1 Introduction

There is still considerable uncertainty about the effects of warfare on biodiversity. Current scholarship has come to the conclusion that most war-induced damages to biodiversity are indirect effects (such as poaching or deforestation) resulting from war-related economic deprivations (Dudley et al., 2002; Gaynor et al., 2016): to date, there is scant evidence that biodiversity loss can occur as a direct consequence of the use of explosives or chemicals. Yet, this conclusion might be premature, as much of the evidence comes either from protracted, low-intensity conflicts such as the FARC insurgency in Colombia, or from conflicts that resulted in profound institutional and economic disruptions, such as the DRC’s civil war.

This paper focuses on the direct effects of warfare on the environment. In order to isolate the effects of war technology, we focus on the 1982 Falklands War between Argentina and the United Kingdom. Contrary to most contemporary conflicts, the Falklands War did not result in profound institutional changes; neither did it directly cause economic collapse (Royle, 1994) or large-scale labor redeployment — two consequences of wars that can indirectly affect wildlife (Dudley et al., 2002; Gaynor et al., 2016). Likewise, we found no report of deliberate tactics aiming at causing environmental damages, which are another oft-mentioned mechanism linking warfare and biodiversity loss (Gurses, 2012). Environmental consequences of the Falklands War can thus be directly traced back to the technology of the conflict.

The Falklands War has other interesting properties. One difficulty in analyzing the environmental consequences of warfare is that conflict is endogenous to environmental degradation. For example, there is mounting evidence that global warming increases the risk of civil war (Hsiang et al., 2011, 2013; Burke et al., 2015). The impact of biodiversity on warfare has not been systematically studied, but some anecdotal evidence suggests that wildlife resources (e.g. ivory, fur or rhino horns) may lead to similar security issues as high-value natural resources such as diamonds, while states’ attempts to deter illegal wildlife trade have turned increasingly violent (Douglas and Alie, 2014; Barron, 2015). At least one quantitative analysis finds that protected areas are used as shelters by insurgents and are associated with an increase in the intensity of violence in their vicinity (Canavire-Bacarreza et al., 2018). By contrast, the Falklands War broke out for reasons that were completely unrelated to environmental issues. Indeed, the conflict is often cited as the textbook example of a “diversionary” war (Levy and Vakili, 1992; Dassel, 1998; Oakes, 2006) and it is widely agreed that the intensity of the war was disproportionate to the economic or strategic value of the remote, sparsely populated Falkland Islands (Freedman, 1982; Mueller, 2009): the conflict was famously described by Argentinian writer Jorge Luis Borges as “a fight between two bald men over a comb.”

A second common identification issue lies in the fact that contemporary conflicts tend to cluster in space (Ward and Gleditsch, 2002; Gleditsch, 2007; Buhaug and Gleditsch, 2008) and in time (Walter, 2015). This

complicates longitudinal analyses—because studies analyzing the effects of recent outbreaks then run the risk of severely underestimating the real effect size—but also cross-country comparisons: indeed, many conflict-affected countries lack adequate control cases, that is, similar (often neighboring) countries that are also home to the species of interest but did not experience war during the investigation period. In contrast to most contemporary conflicts, the Falklands War is a unique event: the Falkland Islands did not experience large-scale military conflict prior to the 1982 war,¹ and the conflict did not relapse afterwards. To summarize, the Falklands War can be considered representative of a broader class of events—namely conventional warfare without deliberate tactic to destroy the environment—but it presents unique advantages in terms of causal identification.

With regard to the outcome, we analyze population dynamics of king penguins (*Aptenodytes Patagonicus*) in the Falkland Islands before and after the war. King penguins possess many of the desirable properties of an indicator species—that is, a species used to monitor ecosystem changes that are unobservable or too costly to measure directly (Hilty and Merenlender, 2000; Hazen et al., 2019). First, king penguins breed on land and are therefore conspicuous, contrary to most marine organisms that are found only underwater; furthermore, since they are colonial and breed at predictable times, their population size can be counted annually (contrast, for example, fish populations). Second, they occupy high trophic levels, and are therefore responsive to lower-level processes—such as ocean productivity or pollution—that occur throughout the entire food web (Burger and Gochfeld, 2004; Durant et al., 2009; Le Bohec et al., 2013; Hazen et al., 2019). Third, three characteristics make them particularly sensitive to ecosystem changes: (a) they are specialist feeders, and thus cannot switch diet in order to cope with lower prey availability (Hilty and Merenlender, 2000; Le Bohec et al., 2013); (b) contrary to many marine top predators, they are philopatric, and therefore unable to relocate to avoid disturbances (Hilty and Merenlender, 2000; Burger and Gochfeld, 2004); (c) they are slower than flying seabirds, which means that their foraging range is relatively restricted. Note also that because of their unusually long breeding cycle (the longest among all penguin species), they are bound to their breeding ground for long periods and thus especially dependent on the quality of their local environment. Finally, in spite of their sensitivity to ecosystem changes, king penguins are a long-lived species with a low reproductive output and thus a relatively stable demography (contrast, again, fish populations): in other words, their response to environmental changes is likely to display a high signal-to-noise ratio (Furness and Camphuysen, 1997).

Our method consists in two steps. First, we use a longitudinal dataset containing yearly estimates of king penguins’ population size over the period 1963-1997. Using interrupted time series (ITS) analysis, we find that the 1982 Falklands War led to a significant trend break and slowed the post-treatment growth of

¹The last significant battle fought in the vicinity of the Falkland Islands occurred in 1914, as a German fleet unsuccessfully attempted to destroy the British naval base on the Falklands (Chehabi, 1985).

the population, even when controlling for the effects of climate change: according to a conservative estimate (based on the lower limit of the 95% CI in our main estimate), the population in 1997 would have been about 1.5 as high in the absence of the war. Second, in order to alleviate potential biases due to unobserved confounders, we build a control unit based on a weighted combination of several other king penguins colonies unexposed to the war. This control unit is obtained using the Synthetic Control method ([Abadie and Gardeazabal, 2003](#); [Abadie et al., 2010](#)). Following [Linden \(2018\)](#), we use this synthetic unit in our regression model in order to evaluate the difference in the post-1982 trends between the king penguins’ colony at the Falkland Islands and its control unit. We find that, while the two trends are not significantly different before the war, they significantly diverge after the war. The magnitude of the effect obtained is roughly similar with and without the control unit. In sum, we find robust evidence that the war led to a long-term slowing of the population growth rate.

2 Past research on warfare and biodiversity

There are very few comparative studies on the effect of warfare on biodiversity: most works focus on a single country, region or conflict episode. To the best of our knowledge, the single analysis using a large sample of countries is a very recent study by [Daskin and Pringle \(2018\)](#) which estimates the impact of armed conflict on large herbivore populations in 126 protected areas spread over 19 sub-Saharan African countries between 1946 and 2010. In their analysis, war-affected zones saw a systematic decrease of most populations (with an effect stronger than any other indicator of human activity), although complete population collapse was rare.

The impact of wars on biodiversity is not necessarily negative: armed conflict often encourages rural exodus ([Pech and Lakes, 2017](#)), farmland abandonment ([Witmer and O’Loughlin, 2009](#)) or the decline of extractive industries through labor redeployment and the withdrawal of timber or mining companies from war zones ([Draulans and Van Krunkelsven, 2002](#); [Mitchell and Thies, 2012](#); [Burgess et al., 2015](#); [Butsic et al., 2015](#)). Relatedly, conflicts sometimes create “no man’s lands” that protect wildlife from human activities, like the oft-cited case of the buffer zone between North Korea and South Korea which has become a biodiversity hotspot ([Dudley et al., 2002](#); [Gaynor et al., 2016](#)). Yet, on the whole, it does not seem that these positive externalities are sufficient to compensate the adverse impact of warfare. [Hendrix and Glaser \(2011\)](#) indeed find that exposure to civil war decreases fish catches, yet, [Mitchell and Thies \(2012\)](#) find the opposite effect. With regard to the refuge-effect, the evidence remains largely anecdotal. Finally, farmland abandonment is only a local and short-term phenomenon, as refugees eventually resettle in other areas ([Baumann et al., 2015](#); [Eklund et al., 2017](#)). An assessment of the early literature ([Dudley et al., 2002](#)) as well as two relatively

recent reviews on wars and biodiversity ([Lawrence et al., 2015](#); [Gaynor et al., 2016](#)) conclude that deleterious effects are observed for all types of warfare (naval, aerial, terrestrial and nuclear), for all phases of the conflict (including production, training and war preparation), and across a wide range of locations and species.

The most obvious negative effect of warfare on wildlife populations is inadvertent killing through the use of explosives or chemicals ([Gaynor et al., 2016](#)). Landmines, for example, do not discriminate between humans and animals, and especially affect large mammals ([Berhe, 2007](#)). However, the two main drivers of wildlife populations decline in wartime appear to be overexploitation due to the economic consequences of the war on the one hand, and habitat loss on the other hand. Overexploitation (e.g., in the form of hunting and wildlife trafficking) results from two interrelated factors. First, institutional collapse during large-scale civil wars destabilizes the conservation sector and greatly reduces the effectiveness of legal enforcement of environmental protection ([Baral and Heinen, 2005](#); [Glew and Hudson, 2007](#)). Second, disruption of the economy in war-affected zones may force local populations or refugees to resort to poaching for subsistence and/or trafficking ([Draulans and Van Krunkelsven, 2002](#); [De Merode and Cowlshaw, 2006](#); [Beyers et al., 2011](#); [Lindsell et al., 2011](#); [Nackoney et al., 2014](#); [Waller and White, 2016](#)); rebels and members of regular military forces also often engage in poaching to fund the insurgency or for personal gain ([Draulans and Van Krunkelsven, 2002](#); [De Merode and Cowlshaw, 2006](#); [Loucks et al., 2009](#); [Benz and Benz-Schwarzburg, 2010](#); [Lindsell et al., 2011](#); [Velho et al., 2014](#)). Poaching often continues in peacetime, as it is often facilitated by the proliferation of small arms in the aftermath of the conflict ([Loucks et al., 2009](#); [Beyers et al., 2011](#); [Nackoney et al., 2014](#)).

Habitat loss can result from guerrilla or counter-insurgency tactics during asymmetric wars. For example, forests are often used as shelters by combatants: as a consequence, deforestation can result either from rebels' direct use of forest resources ([Fjelds  et al., 2005](#)) or from governments' attempt to cut off insurgents from their staging ground ([Van Etten et al., 2008](#); [Gurses, 2012](#)) or to destroy their sources of funding, e.g. illicit crop or timber trafficking ([Alvarez, 2003](#); [Fjelds  et al., 2005](#); [Reuveny et al., 2010](#)). Scorched earth tactics may also be employed to deter local populations from providing shelter, intelligence or monetary resources to combatants, or to deprive rebels' supporters from their livelihood ([Van Etten et al., 2008](#); [Gurses, 2012](#)). The best-known example is the Vietnam War, during which the US military practiced systematic "carpet-bombing" and sprayed herbicides and napalm on Vietnamese forest, with far-ranging consequences for the ecosystem ([Hupy, 2008](#); [Lacombe et al., 2010](#)). Habitat loss may also result from the political and economic impact of the conflict (e.g., post-conflict economic recovery or the resettlement of internally displaced persons), which sometimes last for decades ([Ordway, 2015](#); [Nita et al., 2018](#)).

Generally, protracted civil wars in sub-Saharan Africa, Asia and Latin America are overrepresented in these studies ([Gaynor et al., 2016](#)). This imbalance with regard to location and conflict type somewhat

limits the generality of the findings, and probably explains why most of these studies identify the human, economic and political consequences of the war as the main mechanisms driving biodiversity loss. As a consequence, little is known about the direct impact of the hostilities on biodiversity. Relatedly, few studies have investigated the effects of interstate wars, or, more generally, shorter conflicts that involve tactics more destructive to the environment. Some rare studies have analyzed the consequences of the 1991 Gulf War and its massive oil spills ² on seabird populations (Evans et al., 1993; Price, 1998): they found heavy mortality rates among some species (>50 percent) but also evidence of post-conflict recovery.

3 Case description

3.1 The Falklands War

The Falklands War was a ten-week war between Argentina and the United Kingdom, taking place from April to June 1982. It found its roots in Argentina’s sovereignty claim over the islands, which lie some 600 km east of the Argentine coast (and about 13,000 km from the UK). Numerous disputes occurred between the two countries since the UK established control over the islands in the early 19th century: the Argentine claimed that they were the legitimate owner of the territory and characterized British presence as forceful occupation, while the United Kingdom argued that Falkland Islanders—who were mostly descendants of British settlers—should have the right to self-determination (Weisiger, 2013). Argentina’s sudden decision to seize the Falklands by force is often attributed to domestic factors: in 1982, the military junta led by Leopoldo Galtieri faced intense domestic turmoil and hoped (correctly) that the invasion of the islands would rally the public behind the regime (Levy and Vakili, 1992; Dassel, 1998; Oakes, 2006). Another common explanation is that the outbreak of the conflict resulted from a series of escalatory steps and miscalculations from both governments following a diplomatic incident in the dependency of South Georgia³ (Freedman, 1982). The Argentine military did not expect the UK to be willing and able to retake the islands by force, and believed the international community would condone the invasion since it occurred without British casualties (Weisiger, 2013).

The war started on April 2 with the invasion of the Falkland Islands by the Argentine navy. There was no real deterrent force on the islands, so the UK had to rapidly dispatch a naval task force, which needed several weeks to reach the South Atlantic. Meanwhile, all diplomatic resolution attempts failed. The first effective encounters occurred by late April/early May, with a first British airstrike on the airfield near the capital Stanley (operation Black Buck 1, April 30th), and the sinking of the Argentine light cruiser ARA

²Oil spills in the marine environment amounted to the equivalent of 6-8 millions of barrels (Price, 1998).

³In March 1982, a team of Argentine metal workers illegally landed on South Georgia and raised the Argentine flag. It is unclear whether the action was planned by the junta.

General Belgrano (2 May) which resulted in more than 300 casualties.

During the first weeks of the war, most of the combats took place at sea between the British navy and the Argentine air force. British troops landed in the area of San Carlos Water (northwestern coast of East Falkland) on May 21, and in Bluff Cove (eastern coast of East Falkland) in early June, after which they marched toward Stanley (the capital). After several weeks of land battles, the Argentine garrison in Stanley surrendered, which put an end to the hostilities. The junta briefly considered the option of carrying on attacks on the British task force from the air, hoping to make the costs of retaining the islands unbearable for the UK on the long run (Weisiger, 2013). However, at this point, the junta had lost the support of both the public and the armed forces. Within days, Galtieri was forced to step down, and the military defeat ultimately precipitated the breakdown of the military junta and Argentina’s transition to democracy in 1983.

In spite of its shortness, the Falklands War is responsible for killing some 1,000 military personnel, and for the destruction of dozens of military warships (among others: the 12,000 tons cruiser ARA General Belgrano on the Argentine side; and the 5,500 tons landing ship RFA Sir Galahad as well as the 4,000 tons destroyers HMS Coventry and HMS Sheffield on the British side). The Argentine air forces alone lost more than 100 military aircraft (Smith, 2006).

3.2 King penguins

There are five penguin species breeding on the Falkland Islands. We focus on king penguins because this is the only species for which we could gather sufficient data before and after 1982. King penguins progressively recolonized the islands after possible extirpation at the end of the 19th century, and the first documented successful breeding occurred in 1965 at Volunteer Point, at the northeast of East Falkland (Pistorius et al., 2012).

King penguins have an unusually long breeding cycle, which lasts 14 to 15 months from moulting and courtship to chick fledging: this means that colonies are occupied continuously by chicks, and intermittently by adults (Stonehouse, 1956; Bost et al., 2013). The typical breeding cycle starts with molt by the end of the austral winter. Egg-laying occurs usually in November or December, and hatching in January/February. King penguins can lay one single egg per season, which they incubate on their feet without nest. During incubation and the early stages of the chick’s growth (until mid- to late April), the male and the female exchange duties several times, one bird parenting while the other is foraging at sea. After this period in which they are fed regularly, chicks are left at the colony and fed only intermittently (or not at all) during the winter (May-August), while decreasing food availability forces adults to undertake long foraging trips that often extend over hundreds of kilometers. Parents then resume feeding in spring until fledging.

Because the breeding period last for more than one year, king penguins’ breeding cycle is asynchronous:

successful breeders from the previous year settle later in the colony and start molting and courting with up to two months delay. These breeding attempts often fail because the chick did not store enough food to survive the long winter fast (Stonehouse, 1956; Weimerskirch et al., 1992). The premature death of their offspring allows failed breeders to settle earlier in the colony in the subsequent year: in other words, successful breeders from year 0 become “late breeders” at year 1, and again “early breeders” at year 2 if their previous attempt failed, which is very likely (Le Maho et al., 1993; Dobson et al., 2008).⁴ In spite of the low chances of success of late breeders, the majority of the birds (over 80 percent) attempt to breed annually and only a small minority take a “sabbatical” after successful breeding (Jiguet and Jouventin, 1999; Le Bohec et al., 2007).

3.3 Exposure

The war took place at a time where most adult king penguins had left the colony for foraging trips, with the possible exception of some late breeders (see Table 1). All chicks remained in the colony for the entire duration of the war. Yet, paradoxically, it is probable that breeding adults were more exposed to the war than chicks. The king penguin colony of the Falkland Islands is located at Volunteer Point, in the north-east of East Falkland. Although it is relatively close to the capital Stanley (about 20 kilometers as the crow flies), the colony was insulated from the main land, air, and sea battles. Battles were concentrated around key locations, e.g., Stanley and San Carlos Water (in the west of East Falkland). No landmines were laid by the Argentine at Volunteer Point. Meanwhile, most of the combats at air and sea took place within the Total Exclusion Zone—a circle of radius 370 km from the center of the Falkland Islands—declared by the United Kingdom in April 1982.⁵ This circle was within king penguins’ typical foraging range at this time of the year (Pütz, 2002; Baylis et al., 2015).

⁴Note, however, that the timing and success of late breeding attempts depend on food availability, which varies across years and locations (Olsson and Brodin, 1997)

⁵The British first declared a Maritime Exclusion Zone on April 7, prohibiting Argentine warships from approaching the islands. On April 30, the UK government declared a Total Exclusion Zone covering the same area: any aircraft or ship (military or civilian) from any country entering the zone was liable to be fired upon without further warning. Argentina responded by declaring the entire South Atlantic an Exclusion Zone (May 11). The British TEZ served several purposes, the most important of which were to deter neutral vessels from approaching the area (in order to simplify target identification problems) and to limit the conflict to a single area and to the forces dispatched in this zone (Young, 1992). Obviously, it did not entail any commitment not to fight outside the zone: the most lethal episode of the Falklands War—the sinking of ARA General Belgrano by a British submarine on May 2—occurred some 55 km outside the TEZ.

Table 1: War months and king penguins’ breeding cycle

War months			
	April	May	June
Early breeders	Chick feeding/winter foraging trips	Winter foraging trips	Winter foraging trips
Late breeders	Brooding/chick feeding	Chick feeding/winter foraging trips	Winter foraging trips
Chicks	Ashore	Ashore	Ashore

Note: adapted from [Stonehouse \(1956\)](#)

4 Empirical analysis

In order to assess the effect of the Falklands War on king penguins’ population dynamics, we use interrupted time series analysis. The outcome of interest is the size of the population (expressed in breeding pairs) before and after 1982. The investigation period starts in 1963 and ends in 1997: 1963 corresponds to the onset of the development of the king penguins colony at Volunteer Point (with the first successful breeding occurring in 1965). The choice of 1997 as an end point is motivated by the fact that drilling for oil started in 1998 around the Falklands, resulting in three oil spills that killed hundreds of penguins ([Garcia-Borboroglu et al., 2008](#)).⁶ Another reason is the 1997-98 El Niño event, which triggered an exceptional increase of sea surface temperatures and probably affected population growth in the following years ([Le Bohec et al., 2008](#)).

4.1 King penguin population data

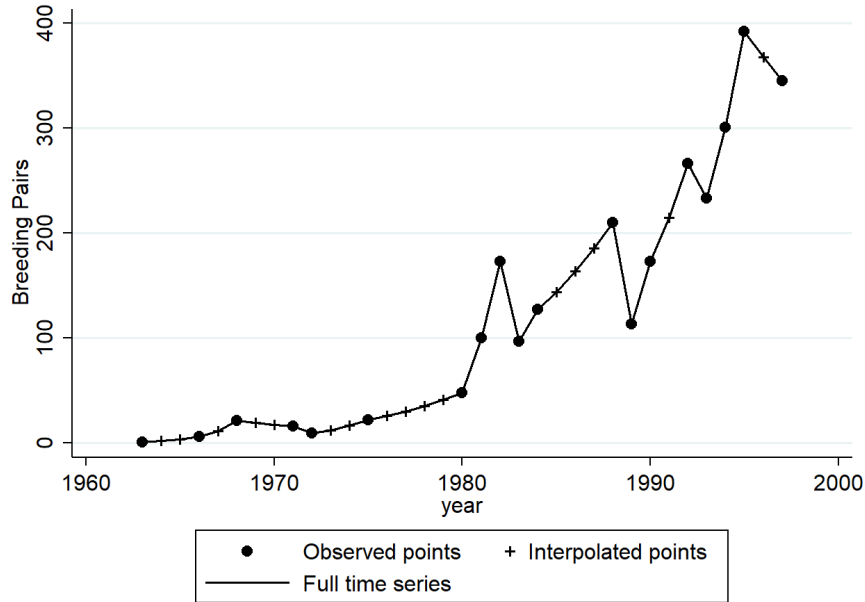
Yearly population estimates come from a variety of sources (see Appendices [A](#) and [F](#) for more details), including datasets made available by research institutes such as the Australian Antarctic Division ([Woehler, 2001](#)), scholarly articles, and grey literature. A first difficulty is that estimates are given in different units (breeding pairs, adults or chicks) depending on the source. In order to obtain homogeneous time series, we converted all observations in breeding pairs-equivalent numbers and removed outliers, defined as data points reflecting a growth rate of more than 125% from one year to another (see Appendix [A](#) for details). A second difficulty lies in the missing data, which make up about half of the yearly observations. ITS analysis abhors a vacuum so we interpolated missing data assuming a geometric growth rate. We thus obtain a time series without gaps including 19 observations before the war, and 16 observations after.

[Figure 1](#) displays our data for the Falkland Islands (including information on observed and interpolated data points) for the whole investigation period. The population is clearly increasing in the pre-treatment as well as in the post-treatment periods, but with a somewhat slower growth rate during the latter. The two

⁶More generally, the longer the investigation period, the more vulnerable the analysis to (observed or unobserved) confounding events—for example, a toxic algal bloom occurred in the Falklands in 2002 ([Trathan et al., 2015](#)).

negative shocks occurring in the immediate aftermath of the war and in 1989 are consistent with immediate and delayed effects of the war (i.e., adult and chick mortality, respectively).⁷ Yet, these data points have to be considered with great caution. Because of the asynchronous breeding cycle of king penguins, yearly counts are susceptible to measurement error because the entire breeding population is never present at the colony at the same time: [Foley et al. \(2020\)](#) estimate potential measurement errors up to 50% in the case of South Georgia. These errors partly explain why we sometimes observe growth rates higher than 50% in consecutive years, which is biologically impossible for king penguins (recall that one pair produces one single chick in a year at best). There are two implications. First, no firm conclusion can be drawn from the observation of individual data points, although the trend over a long period can be analyzed. Second, such an analysis requires sensitivity tests addressing the underestimation problem – an issue that will be dealt with later.

Figure 1: King penguins’ population in the Falkland Islands



4.2 Method

Interrupted time series (ITS) analysis seeks to identify the effect of a shock (or ‘treatment’) in longitudinal data when no comparable control unit is available. This method is commonly used in epidemiological studies and for the evaluation of public health policies, where data are generally available at a population level and the treatment is affecting the whole population ([Bernal et al., 2017](#)). Under these conditions, ITS provides the best way to assess the magnitude of the treatment effect ([Wagner et al., 2002](#)). Interestingly, our case shares some similarities. First, Volunteer Point’s king penguin population cannot be neatly divided into a

⁷Chick mortality (which we do not observe) can be reflected in the adult population data with a delay of 5 to 8 years, which corresponds to the time king penguins take to reach maturity ([Weimerskirch et al., 1992](#)).

sub-population exposed to the war and one unexposed. In other words, there is no directly observable control group. Second, we detain only yearly data at the population level: this level of granularity of the data fits well with ITS. Concretely, ITS here relies on the use of annual time series on king penguins’ abundance prior to 1982 to build a hypothetical scenario depicting what would have happened in the absence of the war (counterfactual). The effect of the war is thus measured by the difference between observed values and the counterfactual.

We use the log-abundance of breeding pairs ($\ln brp_t$) to measure the effect of Falklands war on the population growth rate of the colony. Our main independent variables are the number of years elapsed since the beginning of the study (T_t) and since the beginning of the Falkland War (war_t).

We also include a set of variables controlling for climatic conditions, \mathbf{X}_t , which varies depending on the specification used. Climate change is one of the main threats to king penguins populations (Bost et al., 2013; Trathan et al., 2015; Ropert-Coudert et al., 2019). Direct effects of climate change include the increased frequency of heat waves, severe rains and storms during El Nino events (Ropert-Coudert et al., 2019). Indirect climate effects are food web shifts that threaten penguins’ ability to feed themselves or their chicks: changes in water temperatures affect the distribution and abundance of preys through changes in either ocean productivity or the distribution of currents. As a consequence, warmer sea surface temperature have been found to increase diving depth (which reflects higher energy expenditures) and foraging trip length and duration (Péron et al., 2012; Bost et al., 2015) among king penguins. As Southern Ocean warming has intensified since the 1980s, this may be an important confounder in our study.

We rely on prior studies by Le Bohec et al. (2008), Saraux et al. (2011) and Bost et al. (2015) to identify climatic variables associated with higher mortality of adults, juveniles, and chicks (see details in Appendix B). These three studies rely on the Southern Oscillation Index (SOI), a global climatic variation index that captures a combination of several weather features (Le Bohec et al., 2008), and a more local indicator, namely sea surface temperatures (SST) averaged over the foraging area around the colony, as a proxy for food availability and abundance. However, these three studies use partly different measurement and time lags.⁸ Furthermore, given our data, we need to account for the fact that chick or juvenile mortality (which we do not observe) can be reflected in the adult population data with some delay—up to eight years since about 90% of king penguins have started breeding at age 8 (Weimerskirch et al., 1992). We thus estimate several models with up to eight lags of SOI. With regard to SST, we use the same temporal lag structure, and we additionally define several zones over which temperatures are averaged. These measures are based on empirical studies recording foraging trips of several adult king penguins around the Falklands (Pütz,

⁸For example, Le Bohec et al. (2008) find that adult survival rates are predicted by sea surface temperatures in $t - 2$. This result can be explained by (a) a time lag between SST warming and repercussions on ocean productivity; and (b) a delayed response of adults to increased energy expenditures during poor years.

2002; Baylis et al., 2015) and on a general estimate of the maximal distance king penguins can travel on a single foraging trip before risking massive starvation of their chicks (Péron et al., 2012): details are provided in Appendix B. On this basis, we identify three foraging zones (FZ). *FZ1* corresponds to a zone of 225 km radius around the Falklands-sector (50.5-52.4°S, 56.2-60°W). *FZ2* corresponds to a zone of 545 km radius, which is equivalent to (48.6-54.3°S, 52.5-63.7°W) and *FZ3* is the sector (46.7-56.2°S, 50.6-65.6°W) corresponding to a maximum trip length of 722km – that is, the largest sustainable foraging zone according to Péron et al. (2012). On this basis, considering the different lag structures and measurements, we identify 243 combinations of variables, which we test in separate specifications. We then select the 5 “best” models according to the Akaike’s Information Criterion with correction for small samples (*AICc*). Details on the selection of predictors can be found in Appendix B and descriptive statistics are provided in Table A3.

The regression model used to fit these data is:

$$\ln brp_t = \beta_0 + \beta_1 T_t + \beta_2 war_t + \beta_{env} \mathbf{X}_t + \varepsilon_t \quad (1)$$

where β_0 captures information relative to the situation in 1963, β_1 corresponds to the slope of the log-abundance before the war, and β_2 captures the effect of the war on this slope. In other words, β_2 is the measure of the effect of the Falklands War on the log abundance of king penguins. Finally, β_{env} are the coefficients associated with climatic variables.

4.3 Results

All results were obtained using Stata 14.2.⁹ Table 2 presents the five models with the lowest *AICc* (full regression table are displayed in Appendix D). Falklands War is found to have a detrimental effect on king penguins’ population growth. This effect is significant for Models 1-5, but also for all other estimated models. For the sake of parsimony, we focus on our best model according to the *AICc* criterion: M1. Estimated value of the coefficient associated with the Falklands War, $\hat{\beta}_2$, is -0.106 (95%CI = -0.183 – -0.029) which corresponds to the difference between the log-abundance slope before and after the war.

Figure 2 illustrates this negative slope change attributable to the Falklands war. The dashed lines represent counterfactual scenarios in the absence of war, which are clearly above the fitted values estimated in M1. The first scenario (black dashed line) is a projection of the log-abundance once the estimated effect of the war is removed. Cumulative effects between 1982 and 1997 are very important due to the magnitude of the coefficient $\hat{\beta}_2$. In 1997, the model predicts that the population of the Volunteer Point colony would have been of 1,715 breeding pairs (95%CI = 539 – 5,453), instead of 345. The gray dashed line in Figure 2

⁹StataCorp. 2015. Stata Statistical Software: Release 14. College Station, TX: StataCorp LP

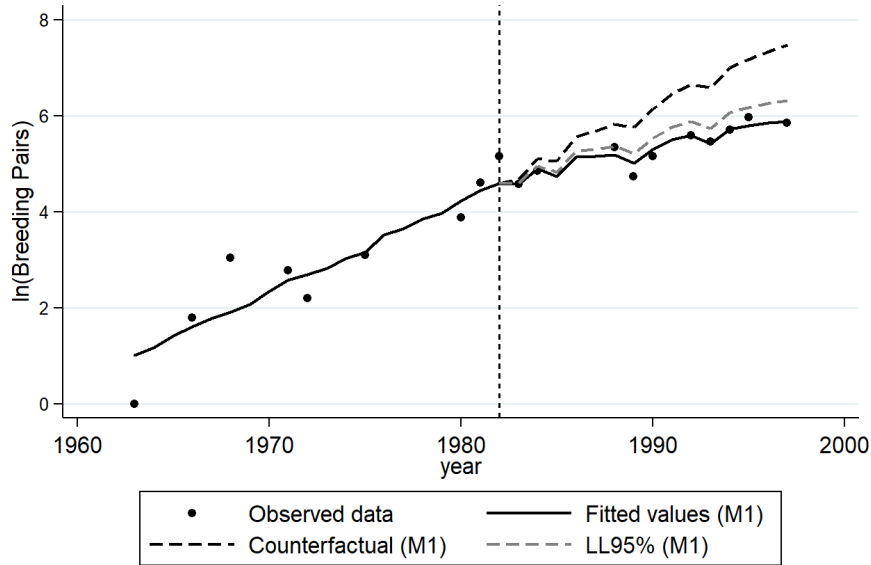
Table 2: Results of linear regressions on the log-abundance of kings penguins from Volunteer Point

Climatic variables					War variable		Goodness of fit
	Global	Lags	Local	Lags	war_t	(<i>sdt. err.</i>)	$\Delta AICc$
M1	SOI_t	-	$SST_t/FZ1$	-	-0.106***	(0.038)	0.000
M2	SOI_t	-	$SST_t/FZ3$	-	-0.103**	(0.038)	0.28
M3	SOI_t	-	$SST_t/FZ2$	-	-0.104***	(0.038)	0.45
M4	SOI_t	-	$SST_t/FZ1$	L1-4	-0.156***	(0.039)	0.87
M5	SOI_t	-	$SST_t/FZ1$	L1	-0.114***	(0.039)	1.38

The five best models are those with the lowest Akaike information criterion corrected (AICc). The best one is indicated in bold. $AICc = -2\text{Log}L + 2k + \frac{2k(k+1)}{N-k+1}$ where k is number of parameters estimated, N the number of observations and $\text{Log}L$ is the log likelihood of the estimated model. $\Delta AICc$ is the difference between the lowest the $AICc$ of the model and the lowest one. ‘Lags’ refers to the lagged values potentially added in the model. In particular, L1-L4 in model 4 means that values of SST in $t-1$, $t-2$, $t-3$ and $t-4$ are included as control variables in (1). Only coefficients associated with the war variable are reported. Newey-west standard errors are reported and the lag structure is defined by the rule-of-thumb $\lfloor N^{1/4} \rfloor = 2$. *, ** and *** mean significant at 10%, 5% and 1%, respectively. Full regression table for M1-M5 is presented in Appendix D.

depicts a conservative scenario corresponding to the lower limit (LL) of the 95% CI of the war effect.¹⁰ In that scenario, the effect size of the war corresponds to 194 breeding pairs, that is, 56.23% of the actual size of the population in this year.

Figure 2: Estimation of the log-abundance of king penguins at Volunteer Point, 1963-1997



As a side note, the results presented above confirm the detrimental effect of climate warming, in line with previous studies by [Le Bohec et al. \(2008\)](#) and [Bost et al. \(2015\)](#). We regularly find a negative and significant

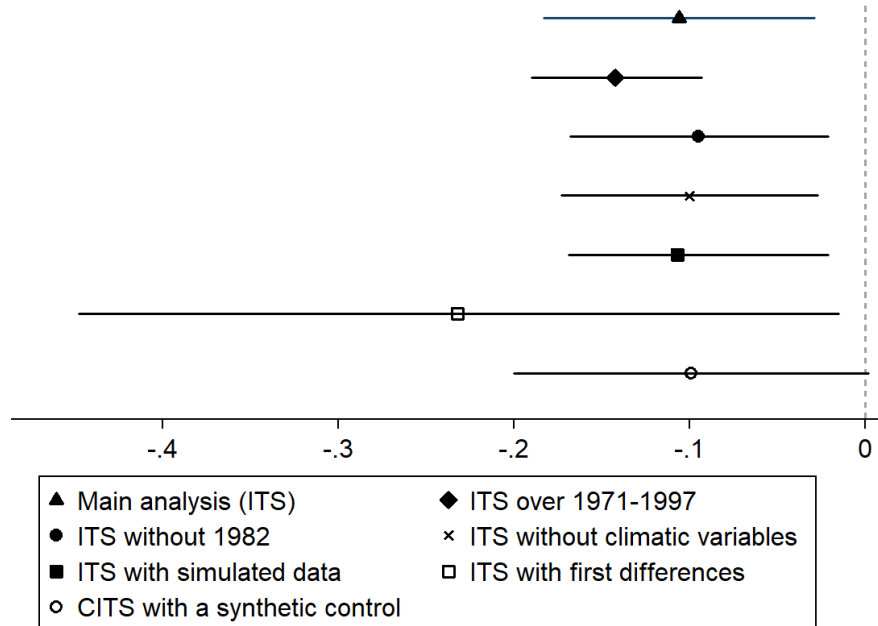
¹⁰One should note that the lower limit of the effect corresponds to the upper limit of the coefficient estimated. In the rest of the paper we will systematically use lower limit without specifying that we mean lower limit of the war detrimental effect.

impact of SST in t on the log-abundance of breeding pairs. Lagged values of SST do not consistently improve the predictive power of the models, but when they are included, their sign is also negative. SOI, on the other hand, does not exhibit the expected sign, but the variable does not attain statistical significance.

4.4 Sensitivity analysis

We perform several sensitivity tests. A first concern is that the start of our investigation period coincides with the origins of the Volunteer Point’s colony. This raises two related issues: artificial growth due to potential immigration from South Georgia¹¹, and high variability of growth rates—since these tend to be more volatile in small populations (Weimerskirch et al., 1992). More concretely, this means in our case that the trend in the first years might be abnormally high, leading to an overestimation of the slope of the log-abundance in the pre-treatment (and thus of the effect of the war). Moreover, the diagnostic of the residuals of M1 – presented in Table A2 – suggests that the fit of the model is relatively poor up to 1970, which can cast doubt on our results. To address these concerns, we delay the start of our study to 1971 and re-estimate M1. The war coefficient is -0.142 (95%CI= $-0.190 - -0.093$) which is even stronger than the coefficient estimated for the full period.

Figure 3: Comparison of the magnitude of the war’s effect (95%CI) on the king penguins log-abundance trend in Volunteer Point using different models



Second, we drop the 1982 data point because we lack information on the timing of the observation, i.e., we

¹¹The first adults that recolonized the Falklands probably originated from South Georgia. Massive migrations between these two locations are very unlikely, since South Georgia lies about 1,550 km away from the Falklands and king penguins are generally philopatric, but considering that the Falklands’ population is a very small one at the beginning of the treatment, even sporadic migrations may have affected the overall trend.

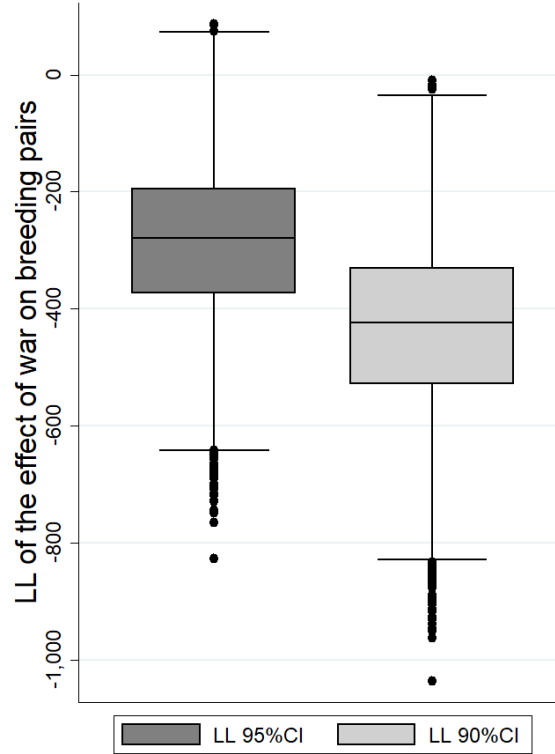
do not know whether the count took place before or after the war. We obtain: $\hat{\beta}_2 = -0.095$ (95%CI=-0.168 – -0.021). Third, we estimate a model without any covariates: results remain remarkably stable. Fourth, for reasons that are related to the peculiarities of king penguins’ breeding cycle, population counts tend to be chronically underestimated. [Foley et al. \(2018, 2020\)](#) estimate that measurement error can be as high as 50%. To address this concern, we run simulations in which each observation is multiplied by a random number lying between 1 and 1.5 to account for underestimation. We randomly draw 10,000 alternative times series and discard those presenting abnormal annual growth rate (*i.e.* superior to 125%). We obtain 6,080 credible alternative time series and re-estimate M1. The effect of the war on the slope of the log-abundance is -0.107 (95%CI = -0.169 – -0.044) which implies that our main findings are not driven by measurement error. Moreover, in all 6,080 estimations, the coefficient associated with the war remains negative and at least significant at the 10% level. A visual representation of the lower limits (LL) of the 95% and 90% CI of the effect of war on the number of breeding pairs in 1997 is provided in [Figure 4](#). Most of the 95% LL values lie between -200 and -400 meaning that the LL obtained with M1 (-194) is a quite conservative measure. Fifth and finally, an autoregressive integrated moving average model with explanatory variables – in our case, an ARIMAX(0,1,0) – was used as an alternative to the Newey-West standard errors to account for autocorrelated data. We difference all variables included in M1 and we find a war coefficient equal to -0.232 (95%CI=-0.448 – -0.015). If the size of the coefficient is larger than those obtained with log-abundance data, it is still in line with prior findings.

4.5 Controlled interrupted time series

In the previous analysis, the construction of the counterfactual was based on the pre-treatment population trend and the evolution of environmental conditions. However, [Linden \(2017\)](#) identifies some factors that can cast doubts on results obtained without considering any control group. First, the decrease in the trend observed after 1982 may have been caused by unobserved events (for example, conservation efforts leading to an increase of the number of predators, *e.g.*, killer whales). Second, our period of investigation is long (35 years) and the evolution of measuring instruments may give rise to validity issues: for example, if measurement errors are decreasing over time, it may have impacted our statistical analysis. Third, the effects of climate change on king penguins are complex ([Le Bohec et al., 2008](#)) and it is possible that we only imperfectly captured the phenomenon. Consequently, a part of the decrease in the slope of king penguins log-abundance may be actually due to climate warming and not the war. For all these reasons, our coefficient of interest $\hat{\beta}_2$ may be biased by unobserved time-varying confounders.

A solution to deal with this issue is to include a control group in our analysis, *i.e.*, to use a controlled interrupted time series (CITS) design ([Bernal et al., 2018](#)). Comparing the treated unit to other, comparable

Figure 4: Simulations of the lower limit of the detrimental effect of Falklands War on king penguins



Based on 6,080 simulated time series corrected for measurement errors ranging from 0 to 0.5. The exponential of the difference between the fitted value and the lower limit of the counterfactual in 1997 are presented. For the sake of comparison, recall that the 95% LL was -194 in our main estimation.

but unexposed units would allow us to distinguish the effect that can safely be attributed to the war from the one that is in fact due to time-varying confounders. In the absence of perfectly comparable control unit, CITS relies on the creation of a “synthetic” control using the method developed by Abadie and colleagues (Abadie and Gardeazabal, 2003; Abadie et al., 2010; Abadie, 2020). A synthetic control is a weighted average of control units unaffected by the treatment, whose aim is to closely replicate the pre-treatment trend of the treated unit in order to identify significant post-treatment divergences.¹² Following Linden (2018), we first estimate the synthetic control so that its difference with the actual population is minimized over the period 1963-1981 (*i.e.* before the Falklands War). We then use the ‘synthetic Falklands’ as a control group to perform a CITS and we compare our results with those obtained in the single-group ITS presented above.

The first step is the construction of the synthetic Falklands. Based on the availability of data, we are able to recompose time series for thirteen colonies (*donors*) over the investigation period: Baie du Marin, Crique de la chaloupe, Jardin japonais and Petite Manchotière from the Crozet Islands; Spit Bay North, Spit

¹²Technical details regarding the Synthetic Control Method are provided elsewhere (Abadie et al., 2010; Abadie, 2020).

Bay South, and Vahsel Moraine from Heard Island; Lusitania Bay from Macquarie Island; Archway Bay and Sealers' Beach from Marion Island; and Fortuna Bay, Gold Harbour and Saint Andrew's Bay from South Georgia. As for the Falklands Islands, missing data were filled using a geometric growth rate in order to obtain time series without gaps. Another issue comes from the fact that population size in 1963 varies from 1 (Falklands Islands) to 45,000 (Baie du Marin). Consequently, we re-scaled each time series by dividing each data point by the average number of breeding pairs in the colony during the period 1963-1997. By doing so, we obtain a donor pool made up of comparable time series (see Appendix E for a visual representation of re-scaled populations and for details on the Synthetic Control Method). We use the standard nested optimization procedure¹³ to obtain the donors weights offering the best fit with the observed log-abundance of king penguins from Volunteer Point. Only past values of the re-scaled populations are used as predictors to avoid overfitting issues (Kaul et al., 2015). The combination of weights minimizing the root mean squared prediction error (RMSPE) are presented in Table 3. The best way to build a comparable synthetic control is to use a combination of three colonies: Baie du Marin, Spit Bay North and Spit Bay South (the RMSPE over the pre-treatment equals 0.249).

Table 3: Weights used to build the synthetic Falklands

Colonies	Weights	Colonies	Weights
Archway Bay	0	Petite Manchotière	0
Baie du Marin	0.025	Sealer's Beach	0
Crique de la Chaloupe	0	Spit Bay North	0.22
Fortuna Bay	0	Spit Bay South	0.755
Gold Harbour	0	Saint Andrew's Bay	0
Jardin Japonais	0	Vahsel Moraine	0
Lusitania Bay	0		

The comparison of the trends of the 'real' Falklands and the synthetic Falklands (displayed in Appendix E) suggests that the war has significantly reduced the population growth rate. More specifically, since this deceleration is visible from the end of the 1980s onward, it is more consistent with a decreased breeding success than with an increased adult mortality rate. However, given the poor quality of the data, this pattern must be interpreted with caution.

The second step consists in using this synthetic control in a CITS analysis, by estimating the following regression:

$$\ln brp_{it} = \beta_0 + \beta_1 T_{it} + \beta_2 real_{it} + \beta_3 post82_{it} + \beta_4 real_T_{it} + \beta_5 war_{it} + \varepsilon_{it} \quad (2)$$

where T_{it} is the time elapsed since the start of the investigation period, $real_{it}$ is a variable that takes the

¹³See Abadie et al. (2014).

value of 1 for the “real” Falklands and 0 for the “synthetic” Falklands, $post82_{it}$ is the time elapsed since the war, $real_T_{it}$ is the interaction between T_{it} and $real_{it}$, and war_{it} is the interaction between $real_{it}$ and $post82_{it}$. war_{it} is the variable measuring the effect of the war on the slope of king penguins log-abundance. Regarding the interpretation of coefficients, β_0 captures the initial level of the log-abundance in the synthetic control (SC), β_1 indicates the slope of the synthetic control before 1982, β_2 represents the difference between the Falklands and Synthetic Falklands regarding the initial level of log-abundance, β_3 is the difference in the slope before and after 1982 for the SC, β_4 is the difference in slope between the Falklands and Synthetic Falklands before 1982, and β_5 indicates the the difference between the Falklands and Synthetic Falklands after 1982 relative to the the difference before 1982 (close to the *difference-in-differences* of trends).

Table 4: Results of the controlled interrupted time series with a synthetic control

	(1)	(2)	(3)	(4)
T_{it}	0.198*** (0.016)	0.198*** (0.016)	0.2*** (0.016)	0.189*** (0.007)
$real_{it}$	-0.109 (0.435)	-0.109 (0.435)	-0.076 (0.438)	-0.015 (0.409)
$post82_{it}$	-0.0005 (0.036)	-0.0005 (0.036)	-0.004 (0.035)	0.001 (0.013)
$real_T_{it}$	-0.009 (0.03)	-0.009 (0.03)	-0.011 (0.03)	0.0002 (0.026)
war_{it}	-0.099* (0.05)	-0.099* (0.05)	-0.095* (0.05)	-0.101** (0.038)
Constant	1.101*** (0.176)	1.101*** (0.176)	1.067*** (0.182)	1.001*** (0.097)
Level	Colonies	Colonies	Colonies	Islands
# of donors (in the SC)	13 (3)	10 (3)	7 (2)	6 (1)
Observations	70	70	70	70

Newey-West standard errors in parentheses (maximum lag = 2), with *** p<0.01, ** p<0.05, * p<0.1. (1) is the model with a synthetic control built on the basis of the full donor pool. In (2) we exclude colonies located in South Georgia from the donor pool. In (3) we keep the 7 colonies with the highest number of real data points during the pre-treatment period. In model (4) we consider a synthetic control built at the island level. All details regarding the construction of the synthetic controls are provided in Appendix E

Column (1) in Table 4 reports values of estimated coefficients. To ensure the comparability of the Falklands and Synthetic Falklands, an important prerequisite is the absence of notable differences between them prior to the war. Considering the non-significant values of coefficients associated with $real_{it}$ and $real_T_{it}$, the real and the synthetic Falklands are comparable before 1982, both in terms of initial values of log-abundance

and growth rate. The coefficient capturing the impact of war – $\hat{\beta}_5$ – equals -0.099 (95%CI=-0.2 – 0.002)¹⁴ which confirms the results we obtained, both regarding the existence and the magnitude of a negative effect of the war on king penguins’ log-abundance. Figure 3 provides a visual comparison with results from the single-group ITS.

Several issues are addressed in Columns (2)-(4) of Table 4. First, one possibility is that colonies located in South Georgia were partially affected by the Falklands War because some fighting took place there. We thus exclude Fortuna Bay, Gold Harbour and Saint Andrew’s Bay from our donor pool. Unsurprisingly, these exclusions do not change anything because these three colonies are not part of the synthetic control (see Table 3). Coefficients are thus identical in Columns (1) and (2). Second, data gathered at the colony level are very scarce, notably prior to the Falklands war (see Appendix F): as a consequence, the fitting procedure may be affected by long interpolations over the period 1963-1981. One risk is that a colony enters the synthetic control’s composition by error, which can cast doubts on the quality of the counterfactual. In order to address this issue, we only retain donors with the highest number of real data points in the pre-treatment period and thus exclude Archway Bay, Baie du Marin, Gold Harbour, Jardin Japonais, Petite Manchotière and Sealer’s Beach from the donor pool (see Appendix E for more details). Coefficients are reported in Column (3) and are very similar.

Column 4 addresses a final source of concern. Although king penguins tend to be faithful to the colony in which they were born, we cannot rule out the possibility of local migrations within each island, possibly following local shocks,¹⁵ and especially when colonies are relatively close to each other: such migrations may artificially inflate the growth rate of some colonies, thus putting into question the quality of the counterfactual. In order to mitigate that risk, we use data at a higher aggregation level, that is, at the island or archipelago level: considering the distance between locations in which king penguins breed,¹⁶ between-island migrations must be even rarer than within-island migrations. We obtain a donor pool composed of six islands/archipelagos: Crozet Islands, Heard Island, Kerguelen Islands, Macquarie Island, Marion Island, and South Georgia. Using the same procedure, we obtain a synthetic control only made up of Heard Island. In other words, the best control group is the Heard Island re-scaled population. Thus, in essence, what we obtain is an interrupted time series analysis with a control group as described by Linden (2017), with the SCM approach ensuring that the treated and the control case are comparable. Column (4) exhibits results very close to those obtained at the colony level with the full donor pool: the war coefficient is -0.101

¹⁴The corresponding p-value is 0.054.

¹⁵An example of such a shock can be found at Crozet archipelago: by the end of the 1960s, the extension of a French research station reduced the surface of the Baie du Marin colony by about 30% (Delord et al., 2004). Neighboring colony Crique de la Chaloupe, which did not exist until 1965, was possibly brought into existence by this event, and we cannot exclude the possibility that the growth rate of this colony was sustained by regular immigration over the whole investigation period.

¹⁶For example, the distance between South Georgia and Macquarie Island is about 7,800 km. The closest locations, Heard Island and the Kerguelen Islands, lie more than 500 km away from each other.

(95%CI=-0.163 – -0.038).

5 Conclusion and discussion

All our results point toward the same conclusion: the Falklands War did have a detrimental effect on king penguins’ population growth rate. The magnitude of the effect is quite large: the size of the colony should have been at least 56% larger than its observed size in 1997. These results are robust to changes in the model specification, in the estimation method and in the period, and are not affected by measurement error, in spite of some variability in the size of the confidence interval.

What happened to the Falklands’ king penguins? Among possible impacts of the war, oil spills are the most likely. The Falklands War resulted in a dozen of shipwrecks, all of which occurred within king penguins’ foraging range. Penguins are especially vulnerable to oil spills because they cannot feed ashore, do not fly (and are therefore probably unable to detect petroleum), and swim at or near the surface ([Garcia-Borboroglu et al., 2008](#)). Furthermore, as central-place foragers, they are susceptible to swimming repeatedly through contaminated zones ([Trathan et al., 2015](#)). Today, oil pollution through shipwrecks and oil leakages is considered one of the main anthropogenic causes of mortality among penguins ([Trathan et al., 2015](#)). Among adverse consequences, oil causes feathers to lose their waterproofing ability (which, in turn, impacts the bird’s thermoregulatory capacities during its stay in the water) and can even reduce buoyancy: birds that depend on marine resources for food then face a trade-off between starvation and drowning or hypothermia ([Leighton, 1993](#); [Garcia-Borboroglu et al., 2008](#)). Moreover, poisoning can follow from ingestion of contaminated water or food sources (e.g., marine invertebrates or fishes), with potentially irreversible damages to vital organs such as the stomach, the liver and kidneys, or to the immune system ([Leighton, 1993](#); [Garcia-Borboroglu et al., 2008](#)). Although oil ingestion has often sublethal effects on adults, exposure of eggs during incubation can reduce eggshell thickness and slow embryonic growth, resulting in high rates of embryo mortality (even after minimal exposure) and developmental defects ([Leighton, 1993](#)). Reduced reproductive output can also result from a decrease of sexual hormones ([Fowler et al., 1995](#)), and from the fact that oiled breeders have lessened foraging abilities and are unable to meet their chick’s energy demands ([Wolfaardt et al., 2008](#)).

Oil spills probably affected adults only. As discussed earlier, chicks ashore in the colony were relatively insulated from land and sea combats. No habitat loss occurred: for example, we found no report of landmines laid at Volunteer Beach. However, one factor that perhaps affected chicks is exposure to noise. The RAF organized several long-distance aerial bombing of Stanley’s airport (Operations Black Buck 1 to Black Buck 7) during which bombers arriving from Ascension Island (a military airfield in the Atlantic) overflew the

northeastern part of East Falkland at short distance from the colony.¹⁷ Noise—which has recently been recognized as a global pollutant—can impact birds in multiple ways (Ortega, 2012). Among immediate effects, approach of aircraft can induce panic responses and lead birds to flee and trample their eggs or chicks, or leave them undefended against predators (Cooper et al., 1994; Hughes et al., 2008).¹⁸ Among long-term effects, noise can trigger stress responses resulting in weight loss, change in hormone levels, and damages to the immune system, and cause hearing impediments¹⁹ that compromise adults’ ability to hear predators, to communicate with their conspecifics, or to identify their chicks in a crowded colony (Ortega, 2012). The latter aspect is especially important as King penguins recognize their mates and chicks exclusively through vocalization (Aubin and Jouventin, 1998; Jouventin et al., 1999).

An important limitation of the present study is that we are unable to distinguish between these mechanisms. This is due both to data limitations and to the indeterminacy of some of the potential effects. For example, adult mortality during the war was necessarily followed by chick mortality, since chicks are unable to forage for themselves until fledgling. Likewise, a long-term decrease of the population growth rate (if any) could be attributed to damages to the marine ecosystem but this effect could be confounded by lasting impacts of war exposure on affected individuals: for example, oil spills can have sublethal effects that do not kill adults immediately but reduce their life expectancy in the long run, and Wolfaardt et al. (2008) find that oiled penguins have a reduced breeding success several years after exposure. This drawback is partially compensated by the fact that we have time series that are long enough for delayed responses (e.g., reproductive success) to be incorporated in the data. Yet, while scholars (Mallory et al., 2010) recommend using multiple indicators (e.g., contaminants concentration in feathers, body mass, changes in diet, increases in foraging distance or duration, breeding success) to reflect changes at various temporal scales, our main indicator is too crude to distinguish between, for example, decreased breeding success vs. long-term decrease of adults’ life expectancy, or to detect possible sublethal effects.

The results of this study have two implications for the analysis of the ecological impact of warfare. First, we show that conflicts can have deleterious consequences on the local ecosystem, even in the absence of deliberate strategies to destroy the environment, and even when the conflict did not have severe economic and institutional repercussions in the affected country. Second, the ecological consequences of armed conflict are lasting and go beyond mere accidental killings: the present analysis shows that the Falklands War affected king penguins’ population growth rate long after the hostilities stopped. These conclusions must be nuanced, given the specificity of our case: the Falklands War was a technologically heavy conflict and king penguins are

¹⁷The precise flight trajectory and the exact distance from the colony are unknown.

¹⁸Cooper et al. (1994) report an incident of mass mortality at the Lusitania Bay colony on Macquarie Island, which was possibly the result of a flypast that caused the birds to panic and die by asphyxiation. They suggest that king penguins may perceive aircraft as aerial predators.

¹⁹According to Ortega (2012), physical damages to ears are rarely permanent among birds, but the recovery time varies from one species to another.

a highly sensitive species. Future research will have to assess whether the present results can be generalized to other types of conflicts and ecosystems.

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Appendices

A Data appendix (Falkland Islands)

This appendix presents our data for the Falkland Islands. Data were gathered from a variety of sources, which are displayed in the table below. In addition to large public datasets, we used estimates found in scholarly articles and reports.

These counts do not necessarily rely on similar methods. Broadly speaking, systematic census data rely either on counts of breeding pairs, often made at the beginning of the reproductive season,²⁰ or on counts of chicks at the end of the austral winter. The latter method is used more often in the Falklands, where breeding success is relatively high (Pütz, 2002; Otley et al., 2007) and the number of living chicks can be viewed as a proxy of the number of breeding pairs with a margin of error due to winter mortality (Bingham, 1998). For each data point, we indicated the counting method when available, but note that many of them are not census data but rough estimates and were not provided with precise information.

One important step was to convert estimates expressed in chicks or adults in breeding pairs-equivalent numbers. Following Jiguet and Jouventin (1999) and Le Bohec et al. (2007), the number of adults was converted into numbers of breeding pairs assuming a 15% rate of non-breeders (these articles estimate this rate at 13% and 17%, respectively). The sole exception is data provided by Paleczny et al. (2015): since they converted estimates given in breeding pairs in adult population estimates assuming one-third of non-breeding birds, we converted these numbers back to their original unit. The number of chicks in a colony can be considered a reasonable approximation of the number of breeding pairs²¹ once chick mortality is accounted for: estimates expressed in number of chicks were converted into numbers of breeding pairs assuming a 20% mortality rate (Bingham, 1998).

These conversion procedures assume that (1) the ratio of non-breeders in the total adult population is stable over time, and (2) chicks' mortality rate is also stable over time. The first assumption is reasonable: it would be false only if king penguins were more likely to refrain from breeding under poor environmental conditions, but to the best of our knowledge, there are no reports of such strategies among king penguins. King penguins are more likely to take a sabbatical following a successful attempt in the previous year (Le Bohec et al., 2007), which suggests that the share of nonbreeders may increase after exceptionally good years; however, the annual proportion of non-breeders displays a small inter-annual variability and does not seem to be influenced by environmental conditions (Jiguet and Jouventin, 1999; Le Bohec et al., 2007).

²⁰These methods are typically used for large colonies such as the ones at Crozet archipelago (Delord et al., 2004).

²¹This makes sense because a) a breeding pair is defined by the fact that it produces a chick, and b) a breeding pair never produces more than one chick per year.

Assumption (2) is more problematic because it depends on actual breeding success, which may vary depending on environmental conditions and food availability before and during the winter fast. However, the amount of error is limited due to constraints on the reproductive output of king penguins (i.e., the chicks will never exceed a certain share of the total population).

Regardless of the method, population estimates are susceptible to systematic underestimation. As noted by [Foley et al. \(2020\)](#), “[a]synchronously breeding species present unique challenges to population monitoring, because there is no single point in the season where all potential breeders (or offspring) are available to be counted” (pp. 251-252). For example, counts of adults made in January miss all the late breeders that start breeding at a later point. Counts based on chicks are also problematic because of either mortality (for late counts) or delayed births (for early counts). According to [Foley et al. \(2020\)](#), counting error can be as high as 50%. As a consequence, interpreting individual data points in the time series is risky at best, and only the analysis of the general trend can provide reliable information.

Counting errors can explain abnormal growth rates observed in some periods of the time series, even with harmonized counting methods. For example, Volunteer Point’s population more than doubled between 2005 and 2006, which is biologically impossible: recall that a breeding pair can only produce a chick (that is, roughly speaking, half a breeding pair) per breeding season, *i.e.*, the highest possible growth rate from one year to another is 50%.²² However, once counting errors are taken into account, it is possible to observe yearly variations such as $(brp_t/brp_{t-1}) \leq 1.5(1 + m) \leq 2.25$, brp being the number of breeding pairs and m the 50% measurement error. Considering this rule, there are no outliers in the Falklands’ time series, except for one impossibly high estimate in 1971 which we discarded.

²²A 50% growth rate from Y_{t-1} to Y_t is unrealistic but theoretically possible. It assumes a) a null or negative growth rate from Y_{t-6} to Y_{t-1} (recall that it takes on average 5-6 years before chicks are recruited into the breeding population) and b) a 100% breeding success in Y_{t-6} .

Data source

Location	Year	Estimate 1	Source	Census	Estimate 2	Source	Census	Final count ^a
Falkland Islands	1963	3 adults	Pistorius et al. (2012)	No				1 (C)
Falkland Islands	1964							2 (I)
Falkland Islands	1965							3 (I)
Falkland Islands	1966	5 chicks	Pistorius et al. (2012)	No				6 (C)
Falkland Islands	1967							11 (I)
Falkland Islands	1968	17 chicks	Pistorius et al. (2012)					21 (C)
Falkland Islands	1969							19 (I)
Falkland Islands	1970							18 (I)
Falkland Islands	1971	40 adults ^{b, c}	Pistorius et al. (2012)	Unknown	15 pairs	Woods & Woods 1997	Unknown	16 (C, A)
Falkland Islands	1972	22 adults ^b	Pistorius et al. (2012)	Unknown				9 (C)
Falkland Islands	1973							12 (I)
Falkland Islands	1974							16 (I)
Falkland Islands	1975	22 pairs	Williams et al. (1979)	Unknown				22 (O)
Falkland Islands	1976							26 (I)
Falkland Islands	1977							30 (I)
Falkland Islands	1978							35 (I)
Falkland Islands	1979							41 (I)
Falkland Islands	1980	38 chicks	Bingham (1998)	Yes ^d				48 (C)
Falkland Islands	1981	100 pairs	Woods (1982)	Unknown				100 (O)
Falkland Islands	1982	495 adults	Paleczny et al. (2015)	Unknown				173 (C)

Notes:

^a The final count (rounded) is expressed in breeding pairs (O: original estimate; C: converted estimate; A: averaged; I: interpolated data).

^b [Pistorius et al. \(2012\)](#) also report counts of chicks in the years 1971 and 1972, but since it might lead to an underestimate of the number of breeding pairs, we use only adults counts.

^c Another estimate is available from [Paleczny et al. \(2015\)](#) for the year 1971, but they report an impossibly high number of adults (660), so we discarded this data point.

^d The census method is unknown.

Data source (continued)

Location	Year	Estimate 1	Source	Census	Estimate 2	Source	Census	Final count
Falkland Islands	1983	78 chicks ^e	Pistorius et al. (2012)	No				97 (C)
Falkland Islands	1984	83 chicks ^e	Pistorius et al. (2012)	No	150 pairs	Woehler and Croxall (1997)	Unknown	127 (C, A)
Falkland Islands	1985							144 (I)
Falkland Islands	1986							163 (I)
Falkland Islands	1987							185 (I)
Falkland Islands	1988	194 pairs	Strange (1992)	Unknown	225 pairs	Strange (1992)	Unknown	210 (A)
Falkland Islands	1989	250 adults	Pistorius et al. (2012)	Unknown	120 pairs	Woods and Woods (1997)	Unknown	113 (C, A)
Falkland Islands	1990	495 adults	Paleczny et al. (2015)	Unknown				173 (C)
Falkland Islands	1991							215 (I)
Falkland Islands	1992	213 chicks	Pistorius et al. (2012)	Yes ^f				266 (C)
Falkland Islands	1993	200 pairs ^g	Woods and Woods (1997)	Unknown	230 chicks	Pistorius et al. (2012)	Yes	233 (C, A)
Falkland Islands	1994	382 pairs	Woehler and Croxall (1997)	yes	626 adults	Paleczny et al. (2015)	Unknown	301 (C, A)
Falkland Islands	1995	314 chicks	Pistorius et al. (2012)	Yes ^f				392 (C)
Falkland Islands	1996							368 (I)
Falkland Islands	1997	276 chicks	Pistorius et al. (2012)	Yes ^f				345 (C)
Falkland Islands	1998	276 chicks	Pistorius et al. (2012)	Yes ^f				345 (C)
Falkland Islands	1999	279 chicks	Pistorius et al. (2012)	Yes ^f				349 (C)
Falkland Islands	2000	258 chicks	Clausen and Huin (2003)	Yes ^f	275 chicks	Pistorius et al. (2012)	Yes	334 (C, A)
Falkland Islands	2001	414 chicks	Pistorius et al. (2012)	Yes ^f				518 (C)
Falkland Islands	2002	341 chicks	Pistorius et al. (2012)	Yes ^f				426 (C)
Falkland Islands	2003	275 chicks	Pistorius et al. (2012)	Yes ^f				344 (C)

Notes:

^e In 1983 and 1984, (Pistorius et al., 2012) report two counts (July and September, and June and July, respectively). Given the season in which these counts were made, we took the earliest ones (i.e., July 1983 and June 1984), which correspond to the largest numbers in both cases. In both years, counts of chicks and adults are provided: however, we selected only counts of chicks, since the counts were made in periods in which a large number of adults were on winter foraging trips while all chicks were ashore. In both cases, the data suggests that the number of adults is underestimated given the number of chicks (78 chicks and 135 adults in July 1983, and 83 chicks and 115 adults in June 1984).

^f Pre-fledged chicks (i.e., those that survived the austral winter) are counted in November at Volunteer Point. Overwinter mortality is comparatively low on the Falklands, where about 90% of chicks survive winter (Outley et al., 2007). This estimate is consistent with the 20% overall mortality rate we apply to convert the number of fledglings into numbers of breeding pairs.

^g We took the largest estimate from the range given by the authors.

B Selection of covariates in the main estimations

Environmental conditions

We use two indicators of climate change. First, the Southern Oscillation Index (SOI) is a global climatic variation index that captures a combination of several weather features (Le Bohec et al., 2008). Second, we use a more local indicator, namely sea surface temperatures (SST) averaged over the area around the Falkland Islands, as a proxy for food availability and abundance. Both SST and SOI have been identified as important predictors of king penguins survival and breeding success (Le Bohec et al., 2008; Saraux et al., 2011; Bost et al., 2015).

Monthly sea surface temperatures are averaged in king penguins’ foraging zone, which we define as an area around the Falklands²³ representing the distance potentially covered by a “typical” adult king penguin during one of its foraging trips.²⁴ We use several estimates of king penguins’ foraging range, two of which rely on empirical studies (Pütz, 2002; Baylis et al., 2015) that recorded foraging trips of several adult king penguins around the Falklands and reported, for each individual trip, the date and the maximal distance to the colony as the crow flies. We use the median of maximal distances reported by both studies ($N = 47$) in selected months. First, we compute the median maximal distance in April and May to define $FZ1$ (225 km, $N = 31$). April and May are the end of the chick-rearing period, that is, a period in which food abundance starts to decrease while chicks’ energy demands are growing (*i.e.*, they must be fed abundantly and frequently in order to be able to sustain several months of fasting during winter). The area thus defined corresponds to a narrow foraging range, in which SST increase may jeopardize king penguins’ reproductive success.²⁵

Second, we define $FZ2$ (545 km, $N=16$) as the median maximal distance to the colony for foraging trips undertaken in June, July and August. The austral winter months correspond to a period of poor food availability, in which chicks fast and adults undertake longer foraging trips to feed.

Although they are empirically grounded, these two measures have a limitation: they rely on a relatively small number of cases in two particular years, which makes them dependent on the climatic conditions during those years.²⁶ For example, if SST were relatively close to average in these years, foraging trips were correspondingly short and foraging areas are therefore too narrowly defined. In order to address this issue,

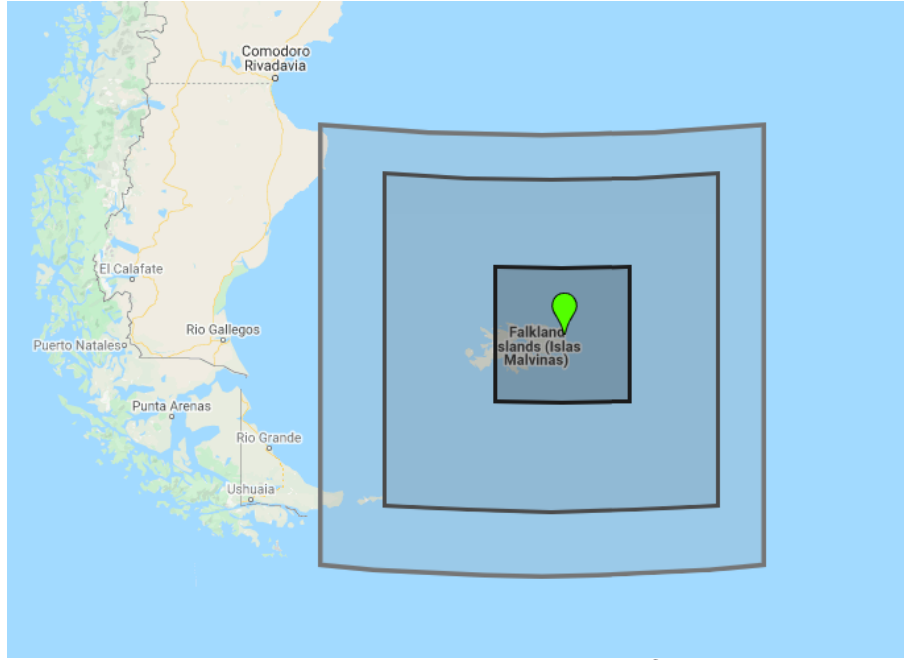
²³King penguins from other colonies typically travel South, but Falklands’ king penguins have been found to travel North and East as well (Pütz, 2002; Baylis et al., 2015). Thus, the foraging areas we define encompasses every direction around the Falklands.

²⁴The three aforementioned studies by Le Bohec et al. (2008), Saraux et al. (2011) and Bost et al. (2015) use data from the Crozet islands, which means that, although we attempt to replicate their main specifications including lag structures as closely as possible, we define foraging areas in a different way.

²⁵If food abundance in this narrow foraging area decreases due to poor climatic conditions, adults feed their chicks either less abundantly (if they remain in this area) or less frequently (if they lengthen their foraging trips to find food). In both cases, chicks face a higher mortality risk.

²⁶There seems to be variability in the length of foraging trips depending on the year: Pütz (2002) reports larger values than Baylis et al. (2015).

Figure A1: Three foraging zones for king penguins from Volunteer Point



Map obtained using Google Maps. copyright notice: Map data ©2019 Google. The smallest/darkest area is FZ1, the intermediate one is FZ2, and largest/lightest one corresponds to FZ3. Each larger zone contains smaller ones. The green marker indicates the king penguin colony at Volunteer Point.

we use a third, more general estimate of the foraging range. According to Péron et al. (2012), the maximal distance covered during a foraging trip cannot exceed 722 km (which corresponds to 20 days of absence from the colony): longer trips dramatically increase the risk of chick starvation. This estimate ($FZ3$) therefore defines the maximal foraging range of king penguins.²⁷

We converted these distances in geographic coordinates. We then drew a square area around Volunteer Point's coordinates (51.47°S, 57.84°W) so that the maximal distance to the colony is 225 km ($FZ1$), 545 km ($FZ2$) and 722 km ($FZ3$). The size of each area is defined as follows: $Area(FZ_i) = \frac{2 \times \text{Max.Dist.}_i}{\sqrt{2}} \text{ km}^2$ where Volunteer Point is the center of the square. After converting kilometers into degrees using standard conversion rules, we obtained areas displayed in Table A1 and Figure A1.

Table A1: Foraging zones' main characteristics

	Maximal distance	Size of the rea	Geographic coordinates
FZ1	225 km	101,250km ²	(50-52.9°S , 55.6-60.2°W)
FZ2	545 km	594,050km ²	(48-54.9°S , 52.7-63.9°W)
FZ3	722 km	1,004,568km ²	(46.9-56.1°S , 51.1-66.1°W)

²⁷This does not mean that a king penguin is never allowed to forage farther than 700 km—indeed, both Baylis et al. (2015) and Pütz (2002) report longer trips in their studies on the Falklands, which is why we chose to use the median of maximal distance—but that chicks' survival would be jeopardized if foraging trips systematically exceeded this distance.

We use monthly values of SST (averaged over $FZ1$, $FZ2$ and $FZ3$) using data from the NCEP Reanalysis Dataset (Kalnay and Coauthors, 1996). This dataset has a spatial resolution of 2.5×2.5 degree latitude and longitude, therefore grid cells exceeding the FZ’s delineation are weighted accordingly. In the end, we obtain three time series labelled: $SST_t/FZ1$, $SST_t/FZ2$ and $SST_t/FZ3$.

Potential covariates not included

Human activities

The main estimates do not include covariates capturing the impact of human activities. This choice is partly justified by the fact that human activities are not currently considered one of the most significant threats to king penguins (Trathan et al., 2015). In what follows, we discuss potential anthropogenic causes of king penguins mortality in the Falklands (beyond climate change) and detail the reasons why we do not include them.

One obvious candidate is the size of the Falklands’ human population. However, data limitations prevents us to include this variable in the predictors (population counts are available only every five years, *i.e.*, 80% of the time series would be missing). Furthermore, there are several reasons to believe that human population size *per se* did not influence king penguins’ demography. First, the population of the Falklands is a small one, and did not change dramatically during the investigation period (from ca. 2,100 inhabitants in 1965 to ca. 2,500 in 1995). Second, the Falkland Islands have an extremely low population density. Most of the inhabitants live in Stanley (the capital), which is about 70 kilometers away from Volunteer Point (king penguins’ main breeding site) by land, and there are no sizeable settlements in the immediate vicinity of Volunteer Point. It seems that the main source of direct human disturbance are punctual visits by tourists (Otley, 2005). Third, the effect of human presence is not straightforward: there is evidence that king penguins repeatedly exposed to human presence learn to consider these disturbances as innocuous and exhibit attenuated stress responses (Viblanc et al., 2012). Bingham (2002) compares the breeding success of several Falkland Islands penguin species in colonies exposed and unexposed to tourism during two breeding seasons; the impact of human presence is virtually null. A further study by Otley (2005) broadly confirms these findings with regard to Gentoo penguin colonies.

Although it seems unlikely that king penguins’ demography was affected by human population *per se*, it might have been influenced by its correlates. The Falkland Islands’ economy, which used to be primarily based on agriculture (especially wool export), grew and diversified during the postwar period (Royle, 1994, 2006): developing sectors include oil, fishing industries, and tourism. For reasons explained above, tourism is not a concern; neither is oil since drilling began after our post-treatment period (Bingham, 2002; Garcia-Borboroglu et al., 2008); yet, fisheries might be. Industrial fishing in the waters surrounding the Falklands

began in the 1970s but really took off by the end of the 1980s, as the Falkland Islands government started to sell licenses allowing foreign vessels to fish within the Falklands Islands Conservation Zone (a zone of 240 km radius around the islands). The selling of these licenses has become the main source of government income shortly after the war (Royle, 1994; Bingham, 2002). The development of commercial fishing is considered a leading cause of the dramatic decline of the Magellanic and Southern Rockhopper penguin populations of the Falklands during the 1980s and 1990s (Bingham, 2002).

Industrial fishing can impact penguin population trends in at least three ways. First, industrial fishing comes with undesirable side effects such as maritime pollution and the spread of debris. Second and most importantly, fishing vessels may harvest species that are an essential part of penguins' diet, leading to chick mortality due to lengthened foraging trips, higher rates of nest abandonment or forced diet change. This mechanism probably accounts for the aforementioned mortality among Magellanic and Rockhopper penguins, as these two species feed on squids and fishes that constitute the main targets of the Falklands' fishing industries (Bingham, 2002). Finally, foraging adults or juveniles may get entangled in fishing nets and drown. We can safely rule out the two latter mechanisms. With regard to competition for sea resources, king penguins feed on myctophids (Le Maho et al., 1993; Olsson, 1996), which are not caught commercially (Bingham, 2002). With regard to accidental bycatch of king penguins, this seems to be an extremely rare phenomenon in the vicinity of the Falklands: as an order of magnitude, one fatality was reported for the 1995-2001 period, and another in 2009 (Crawford et al., 2017). These numbers are relatively recent and may partly reflect technical measures taken to reduce the risk of bycatch; however, it is very unlikely that the Falklands' king penguins experienced mass mortality due to bycatch prior to the 1990s. Regarding the first mechanism, it is difficult to estimate to which extent maritime traffic impacted Falklands' king penguins, so we cannot completely exclude this possibility. We performed additional estimations including fish catches in the waters around the Falklands²⁸ and found no significant effect of fisheries activities (results regarding the effect of the war remain unchanged; results are available upon request). However, these results must be interpreted with caution, since more declared catches can also be a sign of fish abundance (or more generally higher ocean productivity), which can also benefit king penguins.

Natural mortality causes

Disease outbreaks are a potential cause of mass mortality among king penguins (Trathan et al., 2015), but we did not find reports of any epidemics affecting the Falklands' population during the investigation period. Predators are also one of the leading factors affecting breeding failures (Descamps and Gauthier-Clerc, 2005) but in the absence of annual data on the population of common predators, we assume that the mortality rate due to predation remained constant throughout the investigation period. The only factor that would cast

²⁸We used the yearly log of tones of declared fishes catches in the water around the Falklands ($\ln(\text{catch})$). Data are collected by the *Sea Around Us* project Zeller and Pauly (2015).

doubt on this assumption is a systematic growth of predators' population in the post-treatment period, due to, *e.g.*, conservation efforts. However, such a trend would be captured by the inclusion of a control group (see Section 4.5).

C Diagnostic of M1

We present here a full diagnostic of our most predictive model, M1, which includes current SOI and current SST values in the narrowest foraging zone FZ1 (diagnostics of M2-5, which are very similar, are omitted here but available upon request).

Stationarity

We check the stationarity of the dependent variable and climatic variables using the Augmented Dickey-Fuller (ADF) and Phillips-Perron tests (PP). Results are reported in Table A2. Both tests are always passed at a maximum 10% level of significance, ensuring that the risk of spuriousness is minimal. To be conservative, we also estimate an ARIMAX(0,1,0) as a sensitivity test because $\Delta \ln(brp_t)$ strongly passes ADF and PP tests (p-value=0.001). Results of the model in first differences are presented in Section 4.4.

Table A2: Unit root tests for time series used in the paper.

	Z-statistics		MacKinnon p-values	
	ADF	PP	ADF	PP
$\ln brp_t$	-3.370	-3.235	0.0555	0.0777
SOI_t	-4.633	-5.102	0.0001	0.0000
$SST_t/FZ1$	-3.305	-7.708	0.0655	0.0000
$\Delta \ln(brp_t)$	-4.786	-4.741	0.0001	0.0001

The *trend* option is used for $\ln brp_t$ and $SST_t/FZ1$ because the time series grows over time. The lag structure was determined using Akaike's information criterion.

Descriptive statistics

Table A3: Descriptive statistics of data used in the interrupted time series analysis

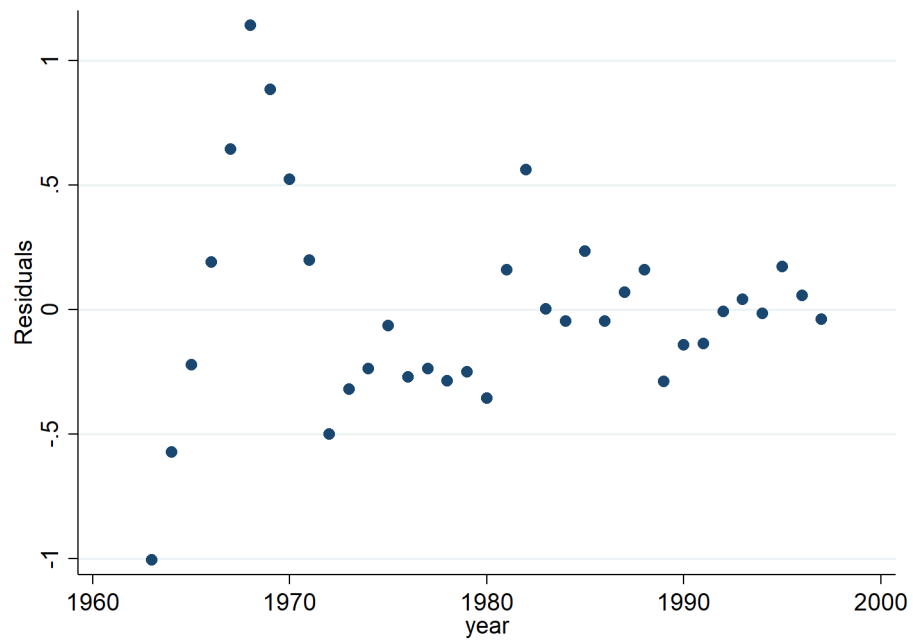
	N	mean	sd	min	max
$\ln brp_t$	35	3.872	1.595	0	5.971
T_t	35	17	10.25	0	34
war_t	35	3.429	4.937	0	15
SOI_t	46	-0.786	7.118	-13.07	13.60
$SST_t/FZ1$	46	6.891	0.189	6.541	7.435
$SST_t/FZ2$	46	7.100	0.169	6.767	7.586
$SST_t/FZ3$	46	7.128	0.169	6.820	7.639

The larger number of observations for climatic variables is due to the use of lags in the estimations.

Residuals analysis

A visual inspection of Figure A2 suggests the existence of a pattern in the residuals distribution. We first perform the Breush-Pagan/Cook-Weisberg test, which indicates the presence of heteroskedasticity (p-value = 0.000). Moreover, the examination of the table of autocorrelations and partial autocorrelations also indicates issues regarding residual auto-correlation. Consequently, we systematically report the Newey-West standard errors with a maximal lag of 2. This number was defined using the following rule-of-thumb maximum lag = $\lfloor N^{1/4} \rfloor$, where $N = 35$ is the number of observations. Moreover, according to Shapiro-Wilk and Shapiro-Francia normality tests, residuals are not normally distributed (p-values are respectively 0.081 and 0.035). Possible reasons are: i) measurement errors in the dependent variable, or ii) omission of crucial independent variables. In our sensitivity analysis, we deal with measurement error by generating 6,080 alternative time series, and obtain results that are very close to those of M1. The omission of predictor variables is also partially addressed in our sensitivity analyses: we estimate 243 different combinations of lags of SST and SOI, and define foraging areas in three alternative ways. Again, results are very similar to those of the main estimates. We also rerun M1 while controlling for the log of tones of declared fish catches in the waters of Falkland Islands ($\ln catch$), as an indicator of human activities. Data are collected by the *Sea Around Us* project following the method developed by Zeller and Pauly (2015). In this estimation, the coefficient associated with war_t equals -0.086 (95%CI = -0.143 – -0.028), which is a little larger than in our main estimate but remains of the same order of 95%CI. Finally, we address the non-normality issue by shortening the period of investigation in order to eliminate extreme values of residuals that are concentrated at the very beginning of our study (*i.e.* 1963-1970). We still find negative and significant effects of the war (-0.142 with 95%CI = -0.190 – -0.093). In light of these results, we can reasonably assume that the non-normality issue does not cast doubts on the main findings. We also check the variance inflation factors to ensure the absence of multicollinearity. Finally, we conduct visual inspection of residuals plots against predictors, and do not identify any systematic pattern.

Figure A2: Residuals of model 1 (M1) over years



D Regression tables (main analysis and sensitivity tests)

This appendix presents all regression tables for M1-M5 for the main analysis and for sensitivity analysis.

Table A4: M1-M5, main analysis

	M1	M2	M3	M4	M5
T_t	0.195*** (0.0274)	0.194*** (0.0272)	0.194*** (0.0273)	0.242*** (0.0304)	0.202*** (0.0287)
war_t	-0.106*** (0.0377)	-0.103** (0.0376)	-0.104*** (0.0378)	-0.156*** (0.0389)	-0.114*** (0.0391)
SOI_t	-0.000359 (0.00691)	-0.00267 (0.00678)	-0.00209 (0.00688)	-0.0101 (0.00977)	0.00130 (0.00707)
$SST_t/FZ1$	-0.438* (0.238)			-0.734** (0.277)	-0.541** (0.222)
$SST_{t-1}/FZ1$				-1.164* (0.568)	-0.458* (0.249)
$SST_{t-2}/FZ1$				-0.856 (0.543)	
$SST_{t-3}/FZ1$				-0.967 (0.590)	
$SST_{t-4}/FZ1$				-1.112** (0.520)	
$SST_t/FZ2$			-0.404 (0.255)		
$SST_t/FZ3$		-0.451* (0.252)			
Constant	3.940** (1.498)	4.137** (1.663)	3.803** (1.665)	33.62** (14.64)	7.725*** (2.190)
Observations	35	35	35	35	35
R-squared	0.935	0.934	0.934	0.954	0.938
AICc	47.44	47.72	47.89	48.31	48.82

Newey-West standard errors in parentheses (maximum lag=2), with *** p<0.01, ** p<0.05, * p<0.1. M1-M5 are the most predictive models described in Table 2.

Table A5: M1-M5 over the period 1971-1997 (sensitivity test)

	M1	M2	M3	M4	M5
T_t	0.229*** (0.0134)	0.227*** (0.0147)	0.227*** (0.0145)	0.232*** (0.0160)	0.236*** (0.0143)
war_t	-0.142*** (0.0232)	-0.138*** (0.0252)	-0.139*** (0.0246)	-0.145*** (0.0268)	-0.149*** (0.0246)
SOI_t	0.00362 (0.00637)	0.000555 (0.00647)	0.00133 (0.00663)	0.00349 (0.0106)	0.00534 (0.00710)
$SST_t/FZ1$	-0.560** (0.222)			-0.511** (0.200)	-0.645*** (0.193)
$SST_{t-1}/FZ1$				-0.313* (0.164)	-0.348** (0.123)
$SST_{t-2}/FZ1$				0.119 (0.209)	
$SST_{t-3}/FZ1$				0.0992 (0.190)	
$SST_{t-4}/FZ1$				-0.153 (0.270)	
$SST_t/FZ2$			-0.542** (0.240)		
$SST_t/FZ3$		-0.571** (0.239)			
Constant	4.185** (1.636)	4.406** (1.835)	4.186** (1.831)	5.512 (3.801)	7.085*** (1.676)
Observations	27	27	27	27	27
R-squared	0.970	0.968	0.967	0.975	0.973
AICc	1.964	3.810	4.181	12.85	1.798

Newey-West standard errors in parentheses (maximum lag=2), with *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$. M1-M5 are the most predictive models described in Table 2.

Table A6: M1-M5 without 1982 (sensitivity test)

	M1	M2	M3	M4	M5
T_t	0.191*** (0.0275)	0.191*** (0.0273)	0.190*** (0.0274)	0.237*** (0.0304)	0.198*** (0.0288)
war_t	-0.0947** (0.0359)	-0.0921** (0.0357)	-0.0929** (0.0359)	-0.143*** (0.0372)	-0.102** (0.0371)
SOI_t	0.00371 (0.00546)	0.00120 (0.00544)	0.00187 (0.00547)	-0.00464 (0.00910)	0.00526 (0.00581)
$SST_t/FZ1$	-0.471* (0.242)			-0.808*** (0.276)	-0.568** (0.229)
$SST_{t-1}/FZ1$				-1.098* (0.549)	-0.430* (0.241)
$SST_{t-2}/FZ1$				-0.799 (0.548)	
$SST_{t-3}/FZ1$				-0.997 (0.603)	
$SST_{t-4}/FZ1$				-0.979* (0.531)	
$SST_t/FZ2$			-0.468* (0.254)		
$SST_t/FZ3$		-0.517** (0.247)			
Constant	4.187*** (1.516)	4.623*** (1.613)	4.271** (1.645)	32.61** (14.70)	7.741*** (2.228)
Observations	35	35	35	35	35
R-squared	0.938	0.938	0.938	0.956	0.941
AICc	45	45.16	45.37	46.45	46.47

Newey-West standard errors in parentheses (maximum lag=2), with *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$. M1-M5 are the most predictive models described in Table 2.

Table A7: M1-M5 without climatic variables (sensitivity test)

	(1)
T_t	0.190*** (0.0253)
war_t	-0.0996*** (0.0359)
Constant	0.992** (0.398)
Observations	35
R-squared	0.932
AICc	43.59

Newey-West standard errors in parentheses (maximum lag=2), with *** p<0.01, ** p<0.05, * p<0.1.

Table A8: First difference M1-M5 (sensitivity test)

	M1	M2	M3	M4	M5
Δwar_t	-0.232** (0.106)	-0.232** (0.106)	-0.232** (0.107)	-0.230** (0.104)	-0.231** (0.104)
ΔSOI_t	0.00497 (0.00564)	0.00377 (0.00566)	0.00404 (0.00571)	0.00347 (0.00585)	0.00580 (0.00554)
$\Delta SST_t/FZ1$	-0.381** (0.180)			-0.596** (0.271)	-0.539** (0.204)
$\Delta SST_{t-1}/FZ1$				-0.658* (0.353)	-0.313 (0.203)
$\Delta SST_{t-2}/FZ1$				-0.344 (0.353)	
$\Delta SST_{t-3}/FZ1$				-0.420 (0.360)	
$\Delta SST_{t-4}/FZ1$				-0.521* (0.288)	
$\Delta SST_t/FZ2$			-0.400* (0.210)		
$\Delta SST_t/FZ3$		-0.458** (0.223)			
Constant	0.280*** (0.0705)	0.281*** (0.0708)	0.280*** (0.0714)	0.293*** (0.0692)	0.282*** (0.0689)
Observations	34	34	34	34	34
R-squared	0.242	0.237	0.223	0.378	0.299
AICc	24	24.25	24.83	30.66	24.30

Standard errors in parentheses, with *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$. M1-M5 are the most predictive models described in Table 2.

E Controlled interrupted time series with a synthetic control

We follow [Linden \(2018\)](#) by using a synthetic control in a CITS analysis. This appendix briefly introduces the synthetic control method (SCM) and provides more details on the CITS analysis included in the paper.

Brief description of the synthetic control method

In order to find a suitable control, we use the *SCM* developed by [Abadie et al. \(2010\)](#). The SCM examines how a treatment (here, the Falklands War) affects an outcome variable (the log-abundance of kink penguins). The general idea is to build a synthetic counterfactual unit (‘Synthetic Falklands’) using a weighted combination of observed control units which were not exposed to the treatment.

We follow here [Klößner et al. \(2018, p. 4\)](#)’s presentation. The SCM requires two types of information: the outcome variable – labelled Y – and the k predictors, X . We denote by the subscript 1 the treated unit, and by 0 the control units. We build a $(k \times J)$ -matrix X_0^{pre} and a k -dimensional vector X_1^{pre} including the averages of the k predictors in the pre-treatment period (1963-1981) respectively for control units and the treated unit. We also build a $(L \times J)$ -matrix Y_0^{pre} and a L -dimensional vector Y_1^{pre} reporting the value of the outcome variable for the J control units and the treated one, where L is the duration of the pre-treatment period. In our main application, we have a pool of 13 control units ($J = 13$), the pre-treatment lasts 18 years ($L = 18$) which also corresponds to the number of predictors ($k = 18$).

For given predictor weights $V = (v_1, \dots, v_k)$, the donor weights $W^*(V)$ is the solution of the following program:

$$\min_W \sum_{m=1}^k v_m (X_{1m}^{pre} - X_{0m}^{pre} W)^2 \quad s.t. W \geq 0, \mathbb{1}'W = 1 \quad (A1)$$

where $\mathbb{1}$ is a J -dimensional vector of ones. In the SCM, optimal predictor weights, V^* , are the solution of:

$$\min_V ||Y_1^{pre} - Y_0^{pre} W^*(V)||^2 \quad s.t. V \geq 0, \mathbb{1}'V = 1 \quad (A2)$$

The SCM offers a counterfactual \hat{Y}_1^{post} of the values of the outcome variable of the treated unit during the post-treatment Y_1^{post} with the optimal donor weights $W^*(V^*)$ and the post-treatment values of control units Y_0^{post} :

$$\hat{Y}_1^{post} = Y_0^{post} W^*(V^*) \quad (A3)$$

Finally, the effect of the treatment, $\hat{\alpha}$ is therefore given by:

$$\hat{\alpha} = Y_1^{post} - \hat{Y}_1^{post} \quad (A4)$$

Abadie et al. (2010) show that if the pre-treatment period is sufficiently long,²⁹ then the difference between the treated unit and its synthetic is an unbiased estimation of the effect of the treatment on the outcome variable.

Donor pool

King penguins have a circumpolar distribution and consistently breed on several sub-Antarctic islands between latitude 45° south and 55° south. During our investigation period,³⁰ colonies could be found in the Falklands and South Georgia in the South Atlantic Ocean, Prince Edward, Crozet, Kerguelen and Heard islands in the Indian Ocean, and Macquarie Island in the Pacific Ocean (Bost et al., 2013).

Volunteer Point is the only king penguin colony of the Falkland Islands, meaning that it can be meaningfully compared to either total (island-wide) populations or to other colonies: indeed, most of the aforementioned islands host several king penguin colonies.³¹ Both strategies have their own strengths and drawbacks. At the colony level, data is more abundant (albeit relatively scarce) and probably more reliable. The main inconvenient of this level of analysis is that we cannot exclude the possibility that migrations take place between colonies within each island or archipelago, thus artificially inflating the growth rate of some colonies among our donors (something we cannot check because we do not have exhaustive data on every colony of each island). This argues in favor of an analysis at the island level. However, even barring data quality issues, such an analysis would not allow us to capture population dynamics that are specific to each colony and do not necessarily reflect at the whole population level. Indeed, the literature shows that there can be significant differences in the growth rate of colonies that are situated on the same island—e.g., Possession Island (Weimerskirch et al., 1992; Delord et al., 2004) or Heard Island (Budd, 2000).³²

We thus pursue both strategies: we begin with an analysis at the colony level, which we then complement with an analysis at the island/archipelago level to probe the robustness of the findings. We start at the colony level and gather population data for thirteen donors, which are listed in Table A9.

This sample is by no means exhaustive with regard to the number of existing colonies, or even islands

²⁹Costalli et al. (2017, p. 83) use the following criteria “[f]or wars lasting less than ten years, we require a pre-treatment period of at least ten years”. In the present work, we have a 19-year pre-treatment period for a three-month war.

³⁰Tierra del Fuego’s small colony exists since the 2010s, and first breeding events have recently been reported at the South Shetland Islands (Juárez et al., 2017). These locations are not included since they did not exist at all during the investigation period.

³¹A partial exception is Macquarie Island, in which only one colony—Lusitania Bay—existed at the beginning of the pre-treatment period.

³²To take a hypothetical example, consider a colony in which local conditions deteriorate for unknown reasons. If this colony is situated on an island in which several other colonies exist, individuals have the possibility of migrating, which means that the colony’s population would decrease while the whole island population would remain stable. If, on the other hand, there is only one colony on the island—as in the case of the Falklands—the whole island population would decrease, considering that king penguins are often unable to breed successfully without a large colony. Thus, from a certain point of view, the possibility of migration argues in favor of an analysis at the colony level.

Table A9: Donors at the colony level

Colony	Island	Archipelago	Location	Country
Archway Bay	Marion Island	Prince Edward Islands	Indian Ocean	South Africa
Baie du Marin	Possession Island	Crozet Islands	Indian Ocean	France
Crique de la Chaloupe	Possession Island	Crozet Islands	Indian Ocean	France
Fortuna Bay	South Georgia	-	Atlantic Ocean	United Kingdom
Gold Harbour	South Georgia	-	Atlantic Ocean	United Kingdom
Jardin Japonais	Possession Island	Crozet Islands	Indian Ocean	France
Lusitania Bay	Macquarie Island	-	Pacific Ocean	Australia
Petite Manchotière	Possession Island	Crozet Islands	Indian Ocean	France
Sealers' Beach	Marion Island	Prince Edward Islands	Indian Ocean	South Africa
Spit Bay North	Heard Island	Heard & McDonald	Indian Ocean	Australia
Spit Bay South	Heard Island	Heard & McDonald	Indian Ocean	Australia
St. Andrew's Bay	South Georgia	-	Atlantic Ocean	United Kingdom
Vahsel Moraine	Heard Island	Heard & McDonald	Indian Ocean	Australia

(for example, we could not find suitable data for the Kerguelen Islands at the colony level). Note also that some locations are overrepresented among donors (e.g., the Crozet Islands). Furthermore, parts of some archipelagos are absent from the donor pool: for example, we only have colonies for Marion Island and Possession Island, and none for the remaining parts of the Prince Edward and the Crozet Islands (that is, Prince Edward Island, and Île de l'Est and Île aux Cochons, respectively).

The main advantage of this sample is that it is relatively representative in terms of locations, since it includes colonies from the Indian Ocean and the Pacific Ocean besides the South Atlantic. To put it otherwise, it limits the risk that the construction of the synthetic is fully driven by some idiosyncratic local conditions.

Time series for the donor pool are based on counts of breeding pairs (the most frequent case), adults, chicks, or (more rarely) total population. Appendix F reports all data points with the corresponding source and—whenever possible—the census method. A few remarks are in order. First, there seems to be no systematic variation in the counting method over time, meaning that we can confidently exclude the possibility of artificial trend breaks due to changes in counting methods. Most of the variation is local (as some colonies were repeatedly censused by the same researcher team) and/or dependent on the colony size. Second, while all counting methods are susceptible to systematic underreporting (Foley et al., 2020), we do not know in which case the bias is the most severe. Third, the most important comparability issue arises from possible differences in chick mortality between locations: indeed, breeding success seems to be higher in the Falklands than in most other locations (Pütz, 2002; Otley et al., 2007). Although this means that population trends of the treated case and the donor pool are not perfectly comparable, this also means that the synthetic control is likely to be an underestimate of the trajectory of the counterfactual Falklands if population data for the donors relies on chick counts.

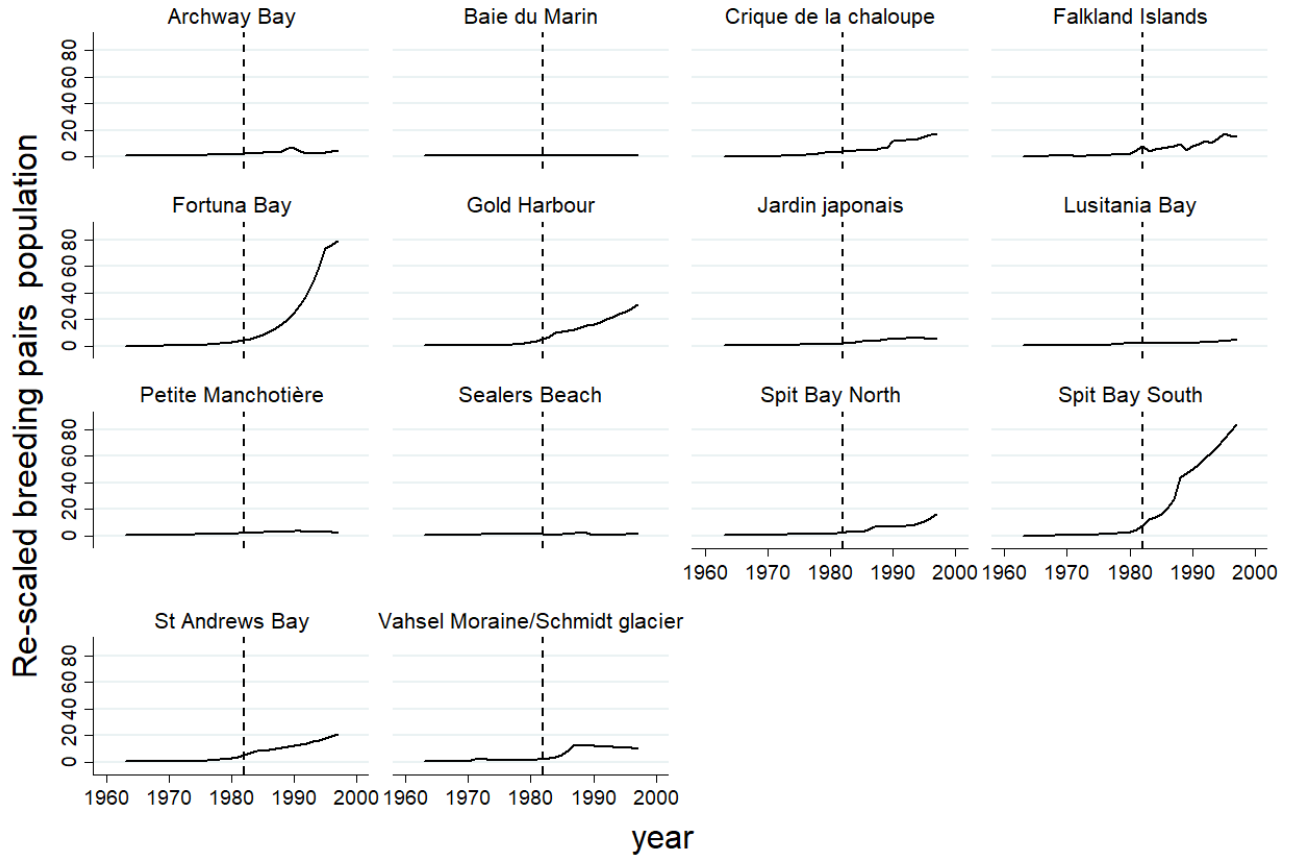
We cannot use raw data as an outcome variable because population sizes are very heterogeneous among

locations: for example, at the beginning of our period of investigation, there were 45,000 breeding pairs at Baie du Marin. We thus re-scaled populations with the following transformation:

$$\text{Re-scaled } brp_t = \frac{brp_t}{\sum_{k=1963}^{1981} brp_k}$$

where brp_t is the number of breeding pairs in period t . Re-scaled populations are displayed in Table A3.

Figure A3: Re-scaled breeding pairs population by colony



The synthtetic control

We use the Stata package SYNTH with the “nested” option and all pre-war outcomes included as predictors. Weights of Synthetic Falklands are reported in Table 3 and a visual comparison between Falklands and its synthetic is provided in Figure A4. The fit is reasonably good for the period 1963-1982 ($RMSPE = 0.2487$) and the two curves clearly split up after the war. In order to be sure that this result is not entirely driven

by one particular donor, we run a leave-one-out test. This sensitivity test consists in excluding one of the thirteen donors and running the exact same procedure with the twelve donors left. Figure A5 shows that the synthetic control obtained in the main procedure (black dashed line) is very similar to to leave-one-out estimates (gray solid lines).

Figure A4: Re-scaled population: Falklands Vs Synthetic Falklands

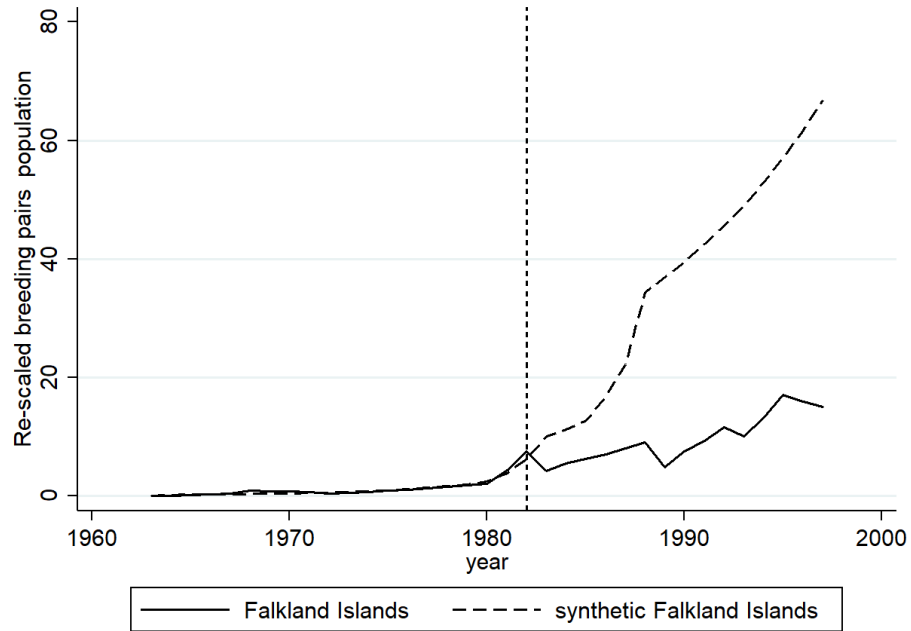
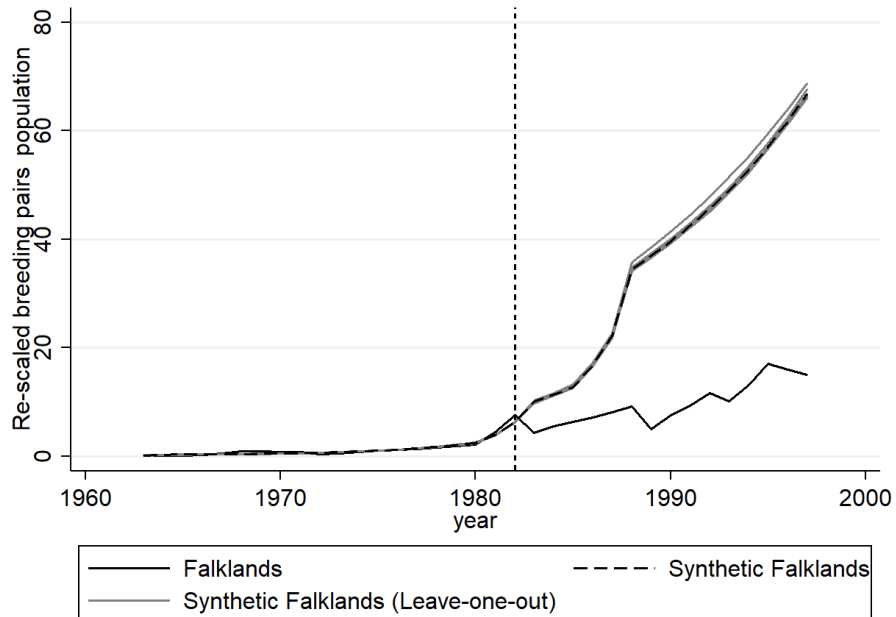


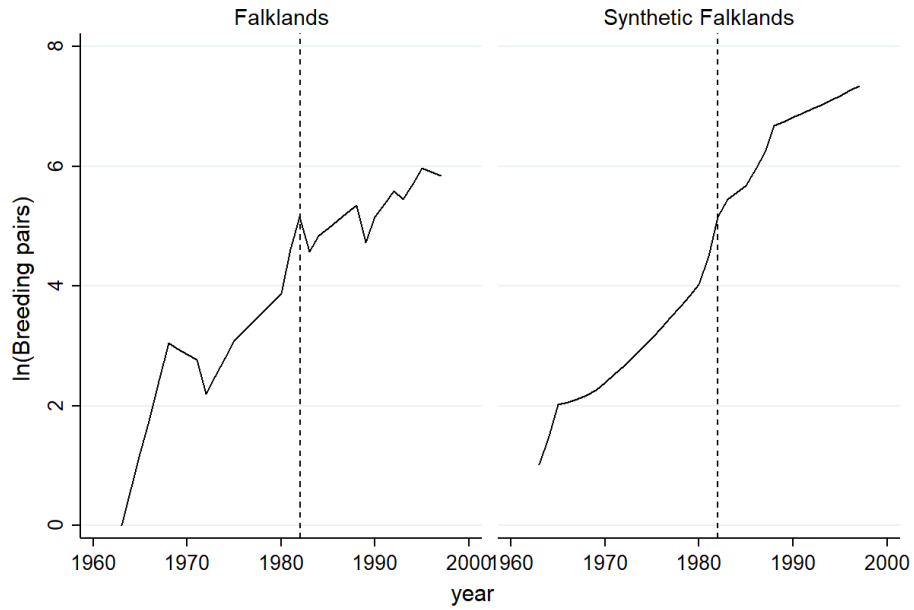
Figure A5: Leave-one-out distribution of the synthetic Falklands



CITS analysis with the synthetic control

Once we obtain the synthetic control, we need to make it comparable with our ‘real’ time series of breeding pairs log-abundance. Concretely, we multiply the synthetic control time series by the average number of breeding pair in Volunteer Point during the pre-treatment period (22.972). We thus obtain a time series corresponding to the counterfactual number of breeding pairs between 1962 and 1997. Finally, we compute the log-abundance of the synthetic control. Figure A6 displays the two curves. These two curves represent the time series which are compared in the CITS analysis (see Table 4). A quick visual inspection confirms the existence of a similar trend during the pre-treatment and a sharp difference after 1982.

Figure A6: Log-abundance : Falklands Vs Synthetic Falklands



Other sensitivity analyses

As mentioned in Section 4.5, several potential issues may cast some doubt on the validity of the CITS. First, one may suspect that colonies located in South Georgia are partially affected by the Falklands War. We thus exclude Fortuna Bay, Gold Harbour and Saint Andrew’s Bay from the donor pool. This does not have any impact on our results because these three colonies were not part of the synthetic control. Second, the number of observations in the time series is sometimes very small, especially during the pre-treatment period (see Appendix F). In particular, one risk is that a colony may artificially enter the synthetic control due to the long interpolations. In order to address this concern, we compute a new synthetic control with a donor pool made of the 50% colonies having the most available observations. Concretely, we exclude Archway Bay, Baie du Marin, Gold Harbour, Jardin Japonais, Petite Manchotière and Sealer’s Beach (6 out of 13 donors).

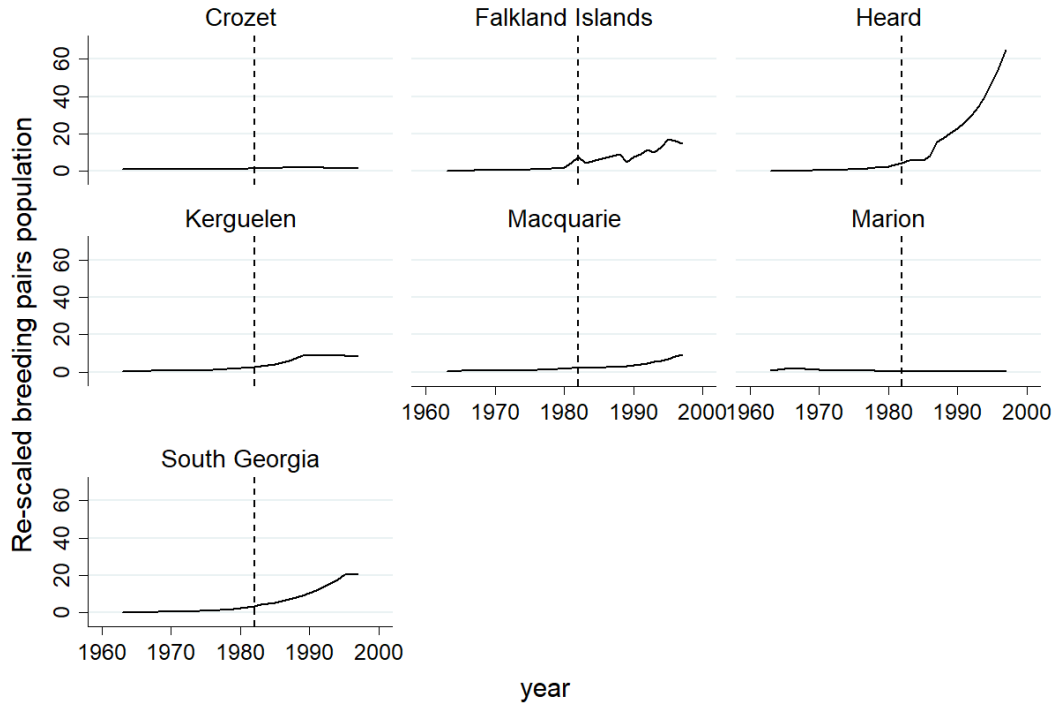
New weights are reported in Table A10. Because the two main contributing donors Spit Bay North and Spit Bay South belong to the set of the relatively better documented colonies, results are very close to those exhibited in Table 3. We then perform the same procedure as above to obtain values of the model (3) in Table 4.

Table A10: Weights used to build the ‘synthetic’ Falklands with the 7 ‘best’ donors

Colonies	Weights	Colonies	Weights
Archway Bay	-	Petite Manchotière	0
Baie du Marin	-	Sealers Beach	-
Crique de la Chaloupe	0	Spit Bay North	0.255
Fortuna Bay	-	Spit Bay South	0.745
Gold Harbour	-	Saint Andrews Bay	0
Jardin Japonais	-	Vashel Moraine	0
Lusitania Bay	0		

The last test addresses the aforementioned issue of migrations within the donor pool. In order to mitigate that risk, we gathered aggregate data at the island level and obtain a smaller pool of six donors: Crozet Islands, Heard Island, Kerguelen Island, Macquarie Island, Marion Island, and South Georgia.³³ Re-scaled populations at the island level are displayed in Table A7.

Figure A7: Re-scaled breeding pairs population by island



³³Note that the migration issue is only partly addressed, since we could not find data for the Prince Edward (close to Marion Island) and the McDonald Islands (close to Heard Island).

As can be seen in Table A11, the synthetic control is composed of one single donor, namely Heard Island. Consequently, the synthetic control used in the CITS presented in Table 4 – model (3) – is a direct comparison between the log-abundance of king penguins in the Falkland Islands and a re-scaled measure of the log-abundance of those located in Heard Island (the re-scaling procedure is the same as above). Results are very similar to those obtained at the colony level with the full donor pool.

Table A11: Weights used to build the ‘synthetic’ Falklands at the island level

Colonies	Weights
Crozet	0
Heard	1
Kerguelen	0
Macquarie	0
Marion	0
South Georgia	0

F Data and sources (donor pool)

Data source

Location	Year	Estimate	Source	Census	# breeding pairs ^a
Archway Bay	1952	200 adults	Rand (1955)	Yes ^b	85 (C)
Archway Bay	1981	479 chicks	Crawford et al. (2003)	Yes ^c	599 (C)
Archway Bay	1982	618 chicks	Crawford et al. (2003)	Yes ^c	773 (C)
Archway Bay	1983	578 chicks	Crawford et al. (2003)	Yes ^c	723 (C)
Archway Bay	1986	1,132 chicks	Crawford et al. (2003)	Yes ^c	1415 (C)
Archway Bay	1987	812 chicks	Crawford et al. (2003)	Yes ^c	1,015 (C)
Archway Bay	1988	473 chicks	Crawford et al. (2003)	Yes ^c	Discarded ^d
Archway Bay	1989	1,725 chicks	Crawford et al. (2003)	Yes ^c	2,156 (C)
Archway Bay	1990	1,775 chicks	Crawford et al. (2003)	Yes ^c	2,219 (C)
Archway Bay	1992	584 chicks	Crawford et al. (2003)	Yes ^c	730 (C)
Archway Bay	1993	600 chicks	Crawford et al. (2003)	Yes ^c	750 (C)
Archway Bay	1994	652 chicks	Crawford et al. (2003)	Yes ^c	815 (C)
Archway Bay	1996	364 chicks	Crawford et al. (2003)	Yes ^c	Discarded ^d
Archway Bay	1997	1,285 chicks	Crawford et al. (2003)	Yes ^c	1,606 (C)
Baie du Marin ^e	1962	45,000 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	45,000 (O)
Baie du Marin	1965	43,400 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	43,400 (O)
Baie du Marin	1981	32,000 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	32,000 (O)

Notes:

- ^a The final count (rounded) is expressed in breeding pairs (O: original estimate; C: converted estimate; A: averaged).
- ^b The estimate relies on field counts, direct counts from photographs, and population density estimates assuming a ratio of eight adults per square yard.
- ^c Chicks were counted in September/October on the field, using a tally counter. According to Crawford et al. (2009), overwinter survival of chicks at Marion Island is relatively high, but the conversion of chicks numbers into numbers of breeding pairs might result in a slight underestimate.
- ^d Abnormally low count.
- ^e This colony is named “Grande Manchotière” in Weimerskirch et al. (1992), and “Baie du Marin” in Delord et al. (2004).
- ^f Counts were obtained from lateral photographs taken from promontories in January. Only birds incubating an egg were counted.

Data source (continued)

Location	Year	Estimate	Source	Census	# breeding pairs ^a
Baie du Marin	1986	40,500 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	40,500 (O)
Baie du Marin	1987	28,200 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	28,200 (O)
Baie du Marin	1997	27,749 breeding pairs	Delord et al. (2004)	Yes ^f	27,749 (O)
Crique de la Chaloupe	1965	0	Weimerskirch et al. (1992)	Yes ^f	0 (O)
Crique de la Chaloupe	1967	8 breeding pairs ^g	Weimerskirch et al. (1992)	Yes ^f	8 (O)
Crique de la Chaloupe	1979	169 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	169 (O)
Crique de la Chaloupe	1986	310 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	310 (O)
Crique de la Chaloupe	1987	280 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	280 (O)
Crique de la Chaloupe	1988	357 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	357 (O)
Crique de la Chaloupe	1989	360 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	360 (O)
Crique de la Chaloupe	1990	676 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	676 (O)
Crique de la Chaloupe	1993	740 breeding pairs	Delord et al. (2004)	Yes ^f	740 (O)
Crique de la Chaloupe	1996	946 breeding pairs	Delord et al. (2004)	Yes ^f	946 (O)
Crique de la Chaloupe	1998	982 breeding pairs	Delord et al. (2004)	Yes ^f	982 (O)
Crozet Islands (total)	1962	424,000 breeding pairs	Guinet et al. (1996)	Partly ^h	424,000 (O)
Crozet Islands (total)	1975	455,000 breeding pairs	Williams et al. (1979)	Unknown	455,000 (O)

Notes

^g The colony apparently did not exist until the 1966/67 breeding season and was possibly brought into existence following a shock occurring at a neighboring colony: by the end of the 1960s, the extension of a French research station reduced the surface of the Baie du Marin colony (a.k.a. Grande Manhotière) by about 30%. As a result, breeding king penguins of this colony (which is located in the bottom of a valley) started colonizing new nesting sites on the valley's slopes (Weimerskirch et al., 1992). However, we cannot rule out the possibility that some individuals migrated to neighboring colonies (incl. Crique de la Chaloupe), and that the growth rate of these colonies was sustained by regular immigration over the whole investigation period.

^h An island-wide census was conducted at Possession Island in 1962. Counts of breeding pairs for each colony were obtained from lateral photographs taken from promontories in January (Weimerskirch et al., 1992). However, colonies at Île de l'Est and Île aux Cochons were not systematically counted. Estimates of population sizes for both islands were added to the Possession Island population to obtain an estimated total population size for Crozet archipelago.

Data source (continued)

Location	Year	Estimate	Source	Census	# breeding pairs ^a
Crozet Islands (total)	1985	800,000 breeding pairs	Guinet et al. (1996)	Partly ⁱ	800,000 (O)
Crozet Islands (total)	1988	1,000,000 breeding pairs	Woehler and Croxall (1997)	Unknown	1,000,000 (O)
Crozet Islands (total)	1995	2,233,659 adults	Paleczny et al. (2015)	Unknown	781,780 (C)
Crozet Islands (total)	2009	673,700 breeding pairs	Bost et al. (2013)	Unknown	673,000 (O)
Fortuna Bay	1957	7 adults	Foley et al. (2018)	Unknown	3 (C)
Fortuna Bay	1965	37 birds ^j	Foley et al. (2018)	Unknown	12 (C)
Fortuna Bay	1974	120 adults	Foley et al. (2018)	Unknown	51 (C)
Fortuna Bay	1975	55 chicks	Foley et al. (2018)	Unknown	69 (C)
Fortuna Bay	1984	376 chicks	Foley et al. (2018)	Unknown	470 (C)
Fortuna Bay	1995	5,000 breeding pairs	Foley et al. (2018)	Unknown	5,000 (O)
Fortuna Bay	2001	6,228 breeding pairs	Foley et al. (2018)	Unknown	6,228 (O)
Gold Harbour	1960	244 chicks	Foley et al. (2018)	Unknown	305 (C)
Gold Harbour	1971	350 adults	Foley et al. (2018)	Unknown	149 (C)
Gold Harbour	1972	500 birds ^j	Foley et al. (2018)	Unknown	159 (C)
Gold Harbour	1984	4,022 chicks	Foley et al. (2018)	Unknown	5,028 (C)
Gold Harbour	2002	25,000 breeding pairs	Foley et al. (2018)	Unknown	25,000 (O)
Heard Island (total)	1963	30 breeding pairs	Woehler (2001)	Yes ^k	30 (O)
Heard Island (total)	1965	68 breeding pairs	Woehler (2001)	Yes ^k	68 (O)
Heard Island (total)	1969	155 breeding pairs	Woehler (2001)	Yes ^k	155 (O)
Heard Island (total)	1980	899 breeding pairs	Woehler (2001)	Yes ^k	899 (O)
Heard Island (total)	1983	2,145 breeding pairs	Woehler (2001)	Yes ^k	2,145 (O)

Notes:

ⁱ An island-wide census was conducted at Possession Island in 1985, during which counts of breeding pairs were obtained from lateral photographs taken from vantage points in early January (Weimerskirch et al., 1992). The population size at Île aux Cochons was estimated using satellite remote sensing Guinet et al. (1995). The numbers for Île de l'Est are rough estimates. ^j Estimates are rarely expressed in total number of birds. Considering that about 15% of adults are non-breeders, that one breeding pair equals two adults, and that the number of chicks is about 80% of the number of adults (see Appendix A), we estimate that the number of breeding pairs corresponds to roughly 31.7% of the total number of birds. We use this conversion rule for all counts of individuals.

^k Birds of each colony were counted on the ground by researchers. The count was replicated once or twice. Ground counts were complemented by oblique ground and aerial photography.

Data source (continued)

Location	Year	Estimate	Source	Census	# breeding pairs ^a
Heard Island (total)	1985	2,093 breeding pairs	Woehler (2001)	Yes ^k	2,093 (O)
Heard Island (total)	1986	2,931 breeding pairs	Woehler (2001)	Yes ^k	2,931 (O)
Heard Island (total)	1987	5,651 breeding pairs	Woehler (2001)	Yes ^k	5,651 (O)
Heard Island (total)	1992	10,745 breeding pairs	Woehler (2001)	Yes ^k	10,745 (O)
Heard Island (total)	2000	37,500 breeding pairs	Woehler (2001)	Yes ^k	37,500 (O)
Jardin japonais	1967	5,000 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	5,000 (O)
Jardin japonais	1981	14,700 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	14,700 (O)
Jardin japonais	1986	33,000 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	33,000 (O)
Jardin japonais	1987	28,000 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	28,000 (O)
Jardin japonais	1988	35,100 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	35,100 (O)
Jardin japonais	1989	41,600 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	41,600 (O)
Jardin japonais	1994	51,504 breeding pairs	Delord et al. (2004)	Yes ^f	51,504 (O)
Jardin japonais	1997	42,033 breeding pairs	Delord et al. (2004)	Yes ^f	42,033 (O)
Kerguelen Islands (total)	1963	20,400 breeding pairs	Bauer (1967)	Yes ^l	20,400 (O)
Kerguelen Islands (total)	1973	30,000 breeding pairs	Williams et al. (1979)	Yes ^m	30,000 (O)
Kerguelen Islands (total)	1985	173,000 breeding pairs	Weimerskirch et al. (1989)	Yes ⁿ	173,000 (O)
Kerguelen Islands (total)	1989	837,870 adults	Paleczny et al. (2015)	Unknown	356,095 (C)
Kerguelen Islands (total)	1998	342,113 breeding pairs	Chamaillé-Jammes et al. (2000)	No	342,113 (O)
Lusitania Bay	1960	9,030 chicks	Rounsevell and Copson (1982)	Yes ^o	11,288 (C)
Lusitania Bay	1976	22,860 chicks	Rounsevell and Copson (1982)	Yes ^p	28,575 (C)

Notes:

^l Censuses were made using aerial photography. These are not individual counts but population estimates based on population density in each area and the total surface of the colony.

^m Census method unknown.

ⁿ Counts were made in January using aerial photographs (taken from a helicopter) or oblique photographs (taken from a vantage point). Population size was estimated on the basis of breeding density calculated on photographs and colony area.

^o Chicks were counted in August (i.e., by the end of the austral winter) on the basis of photographs. Counts were made by piercing the image of each chick and counting the number of holes made on the reverse of the photograph.

^p See footnote (o) for the counting method. Rounsevell and Copson (1982) warn that the number of chicks might be slightly underestimated (by about 5%) because bad weather forced chicks to gather in particularly dense creches during this year.

Data source (continued)

Location	Year	Estimate	Source	Census	# breeding pairs ^a
Lusitania Bay	1978	41,454 chicks	Rounsevell and Copson (1982)	Yes ^o	51818 (C)
Lusitania Bay	1980	46,595 chicks	Rounsevell and Copson (1982)	Yes ^o	58,244 (C)
Lusitania Bay	1984	56,000 breeding pairs	Woehler (1993)	Unknown	56,000 (O)
Lusitania Bay	1990	72,000 breeding pairs	Woehler and Croxall (1997)	Unknown	72,000 (O)
Lusitania Bay	1996	130,000 breeding pairs	Woehler and Croxall (1997)	Unknown	130,000 (O)
Lusitania Bay	2000	160,000 breeding pairs	Heupink et al. (2012)	Unknown	160,000 (A)
Macquarie Island (total)	1960	9,030 chicks	Rounsevell and Copson (1982)	Yes ^q	11,288 (C)
Macquarie Island (total)	1973	<i>3,000 adults</i>	Wienecke and Robertson (2002)	<i>Unknown</i>	<i>Discarded</i> ^d
Macquarie Island (total)	1976	22,860 chicks	Rounsevell and Copson (1982)	Yes ^r	28,585 (C) ^s
Macquarie Island (total)	1980	46,675 breeding pairs ^t	Woehler and Croxall (1997)	Unknown	46,675 (O)
Macquarie Island (total)	1982	46,600 chicks	Wienecke and Robertson (2002)	Unknown	58,250 (C)
Macquarie Island (total)	1984	54,000 breeding pairs	Woehler and Croxall (1997)	Unknown	54,000 (O)
Macquarie Island (total)	1999	323,162 breeding pairs	Goldsworthy et al. (2001)	No	323,162 (O)
Marion Island (total)	1952	52,410 adults	Rand (1955)	Yes ^b	(C)
Marion Island (total)	1966	2,000,000 birds ^j	Conroy and White (1973)	Unknown	634,000 (C)
Marion Island (total)	1974	215,230 breeding pairs	Williams et al. (1979)	Yes ^u	215,230 (O)

Notes:

^q Chicks were counted in August (i.e., by the end of the austral winter) on the basis of photographs taken in the colony of Lusitania Bay (which was the only king penguin colony at Macquarie Island at the time). Counts were made by piercing the image of each chick and counting the number of holes made on the reverse of the photograph.

^r For census methods, see note (q) above. [Rounsevell and Copson \(1982\)](#) warn that the number of chicks might be slightly underestimated (by about 5%) because bad weather forced chicks to gather in particularly dense creches.

^s We added 10 breeding pairs to the count made in Lusitania Bay ([Rounsevell and Copson, 1982](#)) in order to take into account the recent development of a second king penguin colony in Sandy Bay. Lusitania Bay was the only colony on Macquarie Island until 1975. In 1976, there were about 10 breeding pairs in Sandy Bay according to [Wienecke and Robertson \(2002\)](#).

^t This is a sum of the number of breeding pairs in Sandy Bay and Lusitania Bay (80 and 46,595, respectively). There was no other king penguin colony at Macquarie Island at this point.

^u Censuses were conducted during January 1974–April 1975 and April 1976–May 1977. All breeding birds were counted on the ground. Counts in large colonies were then checked using aerial photographs taken from a helicopter.

Location	Year	Estimate	Source	Census	# breeding pairs ^a
Marion Island (total)	1975	215,230 breeding pairs	Williams et al. (1979)	Yes ^u	215,230 (O)
Marion Island (total)	1976	215,230 breeding pairs	Williams et al. (1979)	Yes ^u	215,230 (O)
Marion Island (total)	1977	215,230 breeding pairs	Williams et al. (1979)	Yes ^u	215,230 (O)
Marion Island (total)	1987	41,717 chicks	Crawford et al. (2003)	Yes ^v	52,146 (C)
Marion Island (total)	1990	30,015 chicks	Crawford et al. (2003)	Yes ^v	37,519 (C)
Marion Island (total)	1992	29,455 chicks	Crawford et al. (2003)	Yes ^v	36,819 (C)
Marion Island (total)	1994	40,593 chicks	Crawford et al. (2003)	Yes ^v	50,741 (C)
Marion Island (total)	1996	16,220 chicks	Crawford et al. (2003)	Yes ^v	Discarded ^d
Marion Island (total)	1997	100,059 chicks	Crawford et al. (2003)	Yes ^v	125,074 (C)
Petite manchotière	1965	4,000 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	4,000 (O)
Petite manchotière	1967	3,500 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	3,500 (O)
Petite manchotière	1981	11,700 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	11,700 (O)
Petite manchotière	1986	21,700 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	21,700 (O)
Petite manchotière	1987	17,500 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	17,500 (O)
Petite manchotière	1988	18,337 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	18,337 (O)
Petite manchotière	1989	21,200 breeding pairs	Weimerskirch et al. (1992)	Yes ^f	21,200 (O)
Petite manchotière	1993	16,525 breeding pairs	Delord et al. (2004)	Yes ^f	16,525 (O)
Petite manchotière	1997	14,133 breeding pairs	Delord et al. (2004)	Yes ^f	14,133 (O)
Sealers' Beach	1952	2,000 adults	Rand (1955)	Yes ^b	850 (C)
Sealers' Beach	1981	1,855 chicks	Crawford et al. (2003)	Yes ^c	2,319 (C)
Sealers' Beach	1982	1,493 chicks	Crawford et al. (2003)	Yes ^c	1,866 (C)
Sealers' Beach	1983	957 chicks	Crawford et al. (2003)	Yes ^c	1,196 (C)
Sealers' Beach	1986	1,818 chicks	Crawford et al. (2003)	Yes ^c	2,273 (C)
Sealers' Beach	1987	2,450 chicks	Crawford et al. (2003)	Yes ^c	3,063 (C)

Notes:

^v Counts of chicks that survived the winter fast were made in September or October, either on the ground or using photographs taken from vantage points. In small colonies, counts were made using a tally counter. In large colonies, chicks numbers were estimated by counting chicks in a subsection of the colony and gauging the proportion of the overall area of the colony. According to Crawford et al. (2009), overwinter survival of chicks at Marion Island averages ca. 0.8 (period 1999-2008), which is consistent with the conversion rule we use.

Data source (continued)

Location	Year	Estimate	Source	Census	# breeding pairs ^a
Sealers' Beach	1989	2,546 chicks	Crawford et al. (2003)	Yes ^c	3,183 (C)
Sealers' Beach	1990	1,300 chicks	Crawford et al. (2003)	Yes ^c	1,625 (C)
Sealers' Beach	1992	1,220 chicks	Crawford et al. (2003)	Yes ^c	1,525 (C)
Sealers' Beach	1993	1,569 chicks	Crawford et al. (2003)	Yes ^c	1,961 (C)
Sealers' Beach	1994	1,175 chicks	Crawford et al. (2003)	Yes ^c	1,469 (C)
Sealers' Beach	1996	803 chicks	Crawford et al. (2003)	Yes ^c	Discarded ^d
Sealers' Beach	1997	2,321 chicks	Crawford et al. (2003)	Yes ^c	2,901 (C)
South Georgia (total)	1946	12,000 birds ^j	Lewis Smith and Tallowin (1980)	Yes ^w	3,804 (C)
South Georgia (total)	1965 ^x	11,887 adults	Williams et al. (1979)	Unknown	5,052 (C)
South Georgia (total)	1978	34,000 breeding pairs	Woehler and Croxall (1997)	Unknown	34,000 (O)
South Georgia (total)	1983	83,000 breeding pairs	Trathan et al. (1996)	Yes ^w	83,000 (O)
South Georgia (total)	1986	122,000 breeding pairs	Woehler and Croxall (1997)	Unknown	122,000 (O)
South Georgia (total)	1995	400,000 breeding pairs	Woehler and Croxall (1997)	Unknown	400,000 (O)
South Georgia (total)	2002	450,000 breeding pairs	Bost et al. (2013)	Unknown	450,000 (O)
Spit Bay North	1963	17 breeding pairs	Gales and Pemberton (1988)	Partly ^y	17 (O)
Spit Bay North	1965	62 breeding pairs	Gales and Pemberton (1988)	Partly ^y	62 (O)
Spit Bay North	1969	52 breeding pairs	Gales and Pemberton (1988)	Partly ^y	52 (C)
Spit Bay North	1980	83 breeding pairs	Gales and Pemberton (1988)	Partly ^y	83 (O)
Spit Bay North	1983	220 breeding pairs	Gales and Pemberton (1988)	Partly ^y	220 (O)
Spit Bay North	1985	192 breeding pairs	Gales and Pemberton (1988)	Partly ^y	192 (O)
Spit Bay North	1987	427 breeding pairs	Gales and Pemberton (1988)	Partly ^y	427 (O)
Spit Bay North	1988	434 breeding pairs	Gales and Pemberton (1988)	Partly ^y	434 (O)
Spit Bay North	1993	464 breeding pairs	Green (1993)	Unknown	464 (O)
Spit Bay North	2000	1,800 breeding pairs	Wienecke and Robertson (2006)	Unknown	1,800 (O)
Spit Bay South	1963	6 breeding pairs	Gales and Pemberton (1988)	Partly ^y	6 (O)
Spit Bay South	1965	16 breeding pairs	Gales and Pemberton (1988)	Partly ^y	6 (O)

Notes:

^w Census method unknown.

^x The date is an approximation (counts were made during the 1960s but at different points of time).

^y This is a projection of the number of breeding pairs based on counts of chicks and eggs. Chicks and eggs were counted on the ground during field expeditions to Heard Island (Budd, 1970, 1973; Gales and Pemberton, 1988).

Data source (continued)

Location	Year	Estimate	Source	Census	# breeding pairs ^a
Spit Bay South	1969	40 breeding pairs	Gales and Pemberton (1988)	Partly ^y	40 (O)
Spit Bay South	1980	404 breeding pairs	Gales and Pemberton (1988)	Partly ^y	404 (O)
Spit Bay South	1983	1,765 breeding pairs	Gales and Pemberton (1988)	Partly ^y	1,765 (O)
Spit Bay South	1985	2,286 breeding pairs	Gales and Pemberton (1988)	Partly ^y	2,286 (O)
Spit Bay South	1987	3,929 breeding pairs	Gales and Pemberton (1988)	Partly ^y	3,929 (O)
Spit Bay South	1988	6,256 breeding pairs	Gales and Pemberton (1988)	Partly ^y	6,256 (O)
Spit Bay South ^z	1993	9,000 breeding pairs	Green (1993); Moore et al. (1999)	Unknown	9,000 (A)
St Andrew's Bay	1965	2,500 breeding pairs	Foley et al. (2018)	Unknown	2,500 (O)
St Andrew's Bay	1971	1,500 birds	Foley et al. (2018)	Unknown	Discarded ^d
St Andrew's Bay	1972	6,000 birds ^j	Foley et al. (2018)	Unknown	1,902 (C)
St Andrew's Bay	1974	2,000 chicks	Foley et al. (2018)	Unknown	2,500 (C)
St Andrew's Bay	1984	33,967 chicks	Foley et al. (2018)	Unknown	42,459 (C)
St Andrew's Bay	1985	32,000 chicks	Foley et al. (2018)	Unknown	40,000 (C)
St Andrew's Bay	2002	150,000 breeding pairs	Foley et al. (2018)	Unknown	150,000 (O)
Vahsel Moraine	1963	1 breeding pair	Gales and Pemberton (1988)	Partly ^y	1 (O)
Vahsel Moraine	1969	18 breeding pairs	Gales and Pemberton (1988)	Partly ^y	18 (O)
Vahsel Moraine	1970	27 breeding pairs	Gales and Pemberton (1988)	Partly ^y	27 (O)
Vahsel Moraine	1971	62 breeding pairs	Gales and Pemberton (1988)	Partly ^y	62 (O)
Vahsel Moraine	1980	48 breeding pairs	Gales and Pemberton (1988)	Partly ^y	48 (O)
Vahsel Moraine	1985	177 breeding pairs	Gales and Pemberton (1988)	Partly ^y	177 (O)
Vahsel Moraine	1987	457 breeding pairs	Gales and Pemberton (1988)	Partly ^y	457 (O)
Vahsel Moraine	1993	400 breeding pairs	Moore et al. (1999)	Unknown	400 (O)

Notes:

^z Spit Bay South is called Doppler Hill in Moore et al. (1999). Since the Doppler Hill colony is never mentioned in earlier reports (Gales and Pemberton, 1988; Budd, 2000) in spite of its size and is situated at the same place as Spit Bay South, we assume that this is the same colony. Likewise, "Spit Bay North" becomes "Spit Bay" in Moore et al. (1999).

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