# 'God did not save the kings': Environmental consequences of the 1982 Falklands War[*]

Sophie Panel[†]        Antoine Pietri[‡]

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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 conflict of the late 20st 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 based on data from six other Sub-Antarctic colonies and find similar results.

**JEL codes**: C22, F51, Q57.

**Keywords**: ecological warfare, Falklands War, seabirds.

# 1   Introduction

The fact that warfare has deleterious effects on biodiversity is relatively consensual but uncertainty persists regarding the mechanisms behind this relationship. 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. Neither did the Falklands War spillover to other sub-Antarctic islands (e.g. the Kerguelen or Crozet Islands), which therefore represent ideal control cases. 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).

With regard to the method, we combine an interrupted time series analysis (ITSA) and a synthetic control

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[1]The 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).

([Abadie et al.](), 2010), following the approach proposed by [Linden]() ([2018]()). More concretely, we proceed in three steps. First, we apply ITS to a longitudinal dataset that contains yearly estimates of king penguins' population size over the period 1963-1997: we find that the Falklands War led to a significant trend break and slowed the post-treatment growth of the population, even when controlling for the effects of climate change. Second, in order to alleviate potential biases due to unobserved confounders affecting the treated unit, we use the synthetic control method (SCM), which consists in creating a counterfactual that closely replicates pre-treatment trends of the outcome in the treated unit based on a weighted combination of several other, unexposed control units ([Abadie and Gardeazabal](), 2003; [Abadie et al.](), 2010). Using a combination of six other colonies in four sub-Antarctic islands (Crozet, Heard, Macquarie and Kerguelen Islands), we find that the post-treatment population of the Falklands grew at a slower rate than the synthetic control. Third, following [Linden]() ([2018]()), we combine both approaches and use an ITSA regression model in order to evaluate the difference in the post-treatment trends of the outcome in the treated unit (the "real" Falklands) and its synthetic counterpart (the counterfactual Falklands). We find that, while the two trends are not significantly different before the war, they significantly diverge after the war. In sum, we find robust evidence that the war led to a long-term slowing of the population growth rate: according to our most conservative estimate (based on the lower limit of the 95% CI in our main estimate), by the end of the investigation period, the population would have been about 1.5 as high in the absence of the war.

## 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 large-N analysis on this topic 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. A large-N study by Hendrix and Glaser (2011) indeed finds 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 though 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 Cowlishaw, 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 Cowlishaw, 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 (Fjeldså 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; Fjeldså 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 British 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 Georgia[3] (Freedman, 1982). Most importantly, the Argentine military did not expect the British 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

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

[3]In 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.

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 took several weeks to reach the South Atlantic (during which 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 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 (around 15 percent) 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).

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

[5]The 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

## 4.1 King penguin population data

The outcome of interest is the population size of king penguins (expressed in breeding pairs) in the Falklands before and after the 1982 Falklands War. We collected yearly population estimates for the period 1963-1997. 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 end of our period is motivated by the fact that drilling for oil started in 1998 around the Falklands, resulting in three oil spills that killed hundreds 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 king penguin population growth in the following years (this is reflected in the post-1997 data: see Appendix A).

Yearly population estimates come from a variety of sources (see Appendix A for more details). We first gathered data from datasets made available by the Living Planet Index (LPI, 2018), the Australian Antarctic Division (Woehler, 2006), and the Sea Around Us project (Paleczny et al., 2015), which we completed by thoroughly reviewing 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: details are provided in Appendix A. We removed outliers, defined as data points reflecting a growth rate of more than 125% from one year to another: the rationale for this rule is explained in Appendix A, and takes into account both the (theoretically) maximal annual growth rate of a king penguin colony and potential counting errors discussed in Foley et al. (2020). A second difficulty lies in the missing data, which make up about half of the yearly

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[6]More 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).

observations. ITSA 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. Population trends are 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 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). Yet, these data points have to be considered with great caution. Because of the asynchronous breeding cycle of king penguins, yearly counts are particularly 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 (up to 125% at most), 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: Kings penguins population in the Falkland Islands
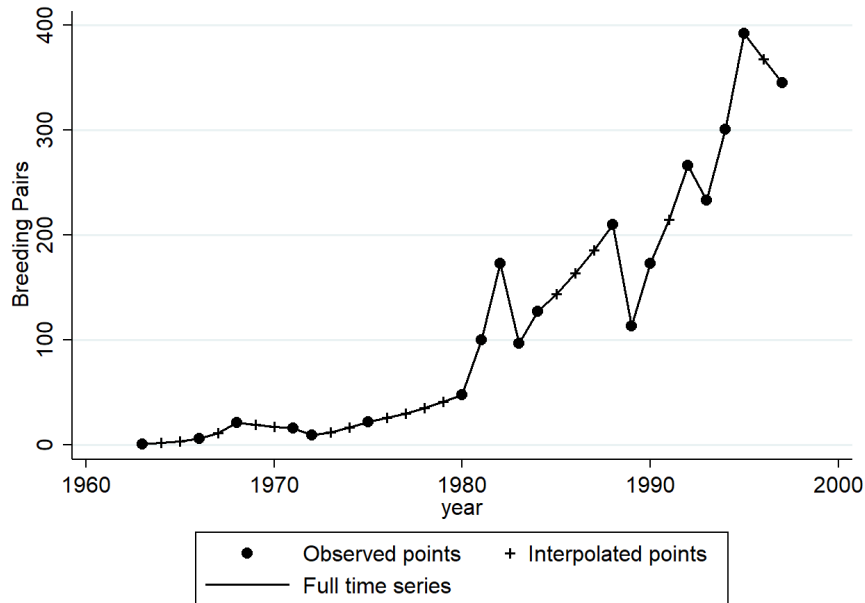
## 4.2 Method

Interrupted time series analysis (ITSA) offers a way to evaluate the impact of an intervention ("treatment") in a time series. In our case, ITSA 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 can thus be measured by the difference between observed values and the counterfactual.[7]

Our main objective is to assess the potential effect of Falklands war on the average population growth rate of the colony: we therefore use as a dependent variable the log of abundance of breeding pairs, which we estimate as a linear function of the time ($T_t$). Our main predictor is $war_t$ representing the number of years elapsed since the war. We also include a set of variables controlling for environmental conditions. Details on the selection of predictors can be found in Appendix B and descriptive statistics are provided in Table A3. Concretely, our empirical strategy consists in estimating the following segmented regression model:

$$\ln brp_t = \beta_0 + \beta_1 T_t + \beta_2 war_t + \boldsymbol{\beta_{env}} \boldsymbol{X_t} + \varepsilon_t \tag{1}$$

where $\ln brp_t$ is the log-abundance of breeding pairs in $t$, $T_t$ and $war_t$ are respectively the number of years elapsed since the beginning of the investigation period and since the war, and $\boldsymbol{X_t}$ is a set of climatic variables which varies depending on the specification (more on this below). Regarding the interpretation of coefficients: $\beta_0$ captures information relative to the situation in 1963, $\beta_1$ corresponds to the slope of the log-abundance before the war, $\beta_2$ captures the effect of war on this slope. In other words, $\beta_2$ is the measure of the effect of the Falklands War on the log abundance of king penguins. Finally, $\boldsymbol{\beta_{env}}$ are the coefficients associated with climatic variables.

As mentioned above, our main covariates are climate-related variables. 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, (warm) 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). 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

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[7]See Bernal et al. (2017) for more details on ITSA.

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, 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): all details can be seen 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 $(50.5\text{-}52.4°S, 56.2\text{-}60°W)$. $FZ2$ corresponds to a zone of 545 km radius, which is equivalent to $(48.6\text{-}54.3°S, 52.5\text{-}63.7°W)$ and $FZ3$ is the sector $(46.7\text{-}56.2°S, 50.6\text{-}65.6°W)$ corresponding to a maximum trip 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 ($AICc$).

## 4.3   Results

Results were obtained using Stata v.14.2. Table 2 presents the five models with the lowest AICc: full regression tables are displayed in Appendix A4.

Turning to the main results, it appears that the Falklands War had a detrimental effect on king penguins' population growth. This effect is significant for M1-5, but also for all other estimated models. For the sake of parsimony, we focus on our best model (lowest AICc).[9] In M1, the estimated value of the coefficient associated with the Falklands war, $\hat{\beta}_2$, is -0.106 (95%CI = -0.183 – -0.029) which indicates the difference between the log-abundance slope between the pre-war and the post war period. 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

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[8]For 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.

[9]Full diagnostic of M1 is provided in Appendix C.

Table 2: Model selection

| | Climatic variables | | | | War variable | | Goodness of fit |
|---|---|---|---|---|---|---|---|
| | Global | \| Lags | Local | \|Lags | $war_t$ | $(sdt.\ err.)$ | $\Delta AICc$ |
| **M1** | $\boldsymbol{SOI_t}$ | - | $\boldsymbol{SST_t/FZ1}$ | - | $\boldsymbol{-0.106^{***}}$ | **(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 5 best models are reported (lowest AICc). The best one is indicated in bold. $AICc = -2LogL + 2k + \frac{2k(k+1)}{N-k+1}$ where $k$ is number of parameters estimated, $N$ the number of observations and $LogL$ 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. 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 of M1-M5 is presented in Appendix D.

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 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: Log-abundance of king penguins at Volunteer Point, 1963-1997



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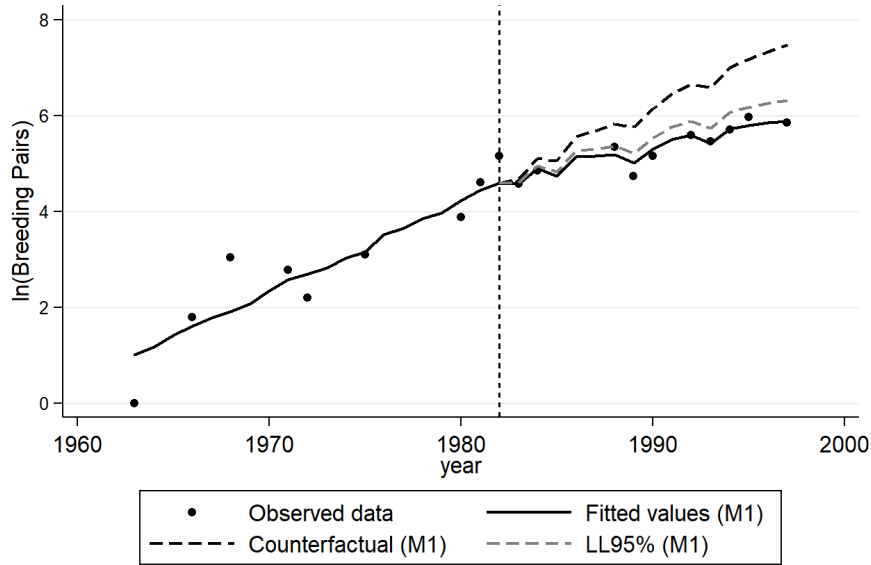
[10]One should note that the lower limit of the effect corresponds to the upper limit of the coefficient estimated. In the rest of the paper we will systematically use lower limit without specifying that we mean lower limit of the war detrimental effect.

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 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: all results are summarized in Figure 3. 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 problems: artificial growth due to immigration from South Georgia, and high variability of growth rates–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 that the coefficient estimated for the full period.

Figure 3: Magnitude of the war's effects (95%CI)
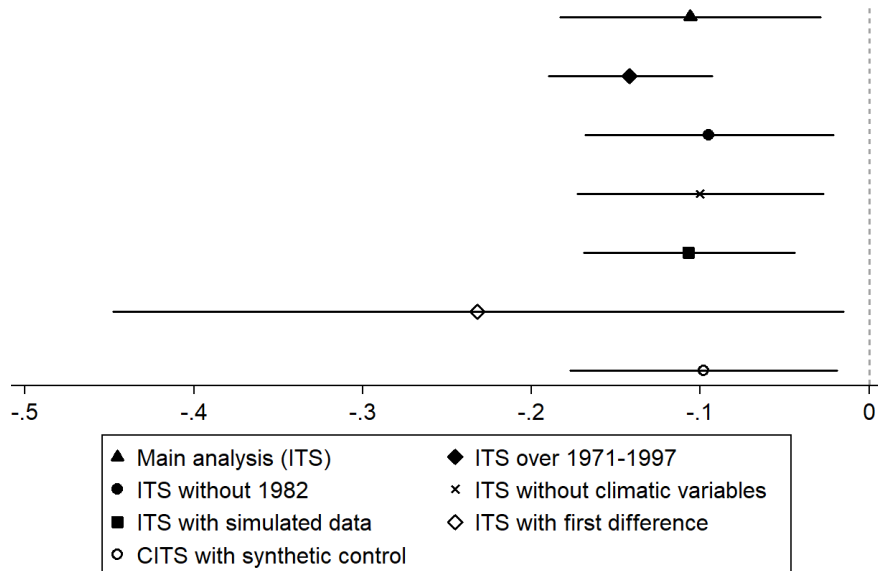


Second, we drop the 1982 data point because we lack information on the timing of the observation (before or after the war). We obtain: $\hat{\beta}_2 = -0.095$ (95%CI=-0.168 – -0.021). Third, we estimate a model without any covariates: results remain remarkably stable. Fourth, for reasons that are related 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 end up with 6,080 credible alternative time series and we 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 clearly implies that our main findings are not driven by measurement error. Moreover, in all 6,080 estimations, the coefficient associated with 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 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 still increases our confidence on the identification of a detrimental effect of Falklands war on king penguins.

## 4.5  Controlled interrupted time series

In our single-group ITSA, 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 changes and not related to the effect of 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 series in our statistical analysis, *i.e.* using a Controlled Interrupted Time Series (CITS) design (Bernal et al., 2018). Comparing the king penguins' colony of Volunteer Point to other comparable colonies 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. Consequently, using data

Figure 4: Simulations of the minimal effects of Falklands war on king penguins
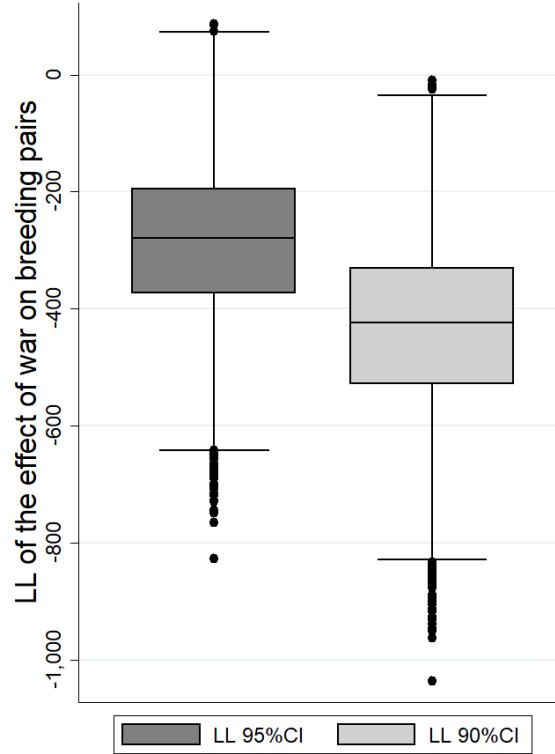


Based on 6,080 simulated time series corrected for measurement errors ranging from 0 to 0.5. The exponential of the difference between the fitted value and the lower limit of the counterfactual in 1997 are presented. For the sake of comparison, recall that the 95% LL was - 194 in our main estimation.

on colonies located in other sub-antarctic islands would clearly mitigate risks of bias in the identification of the treatment effects.

Because there are no perfectly comparable colonies, we recreate a "synthetic" one 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 cases 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.[11] Following Linden (2018), we proceed in two steps. First, we estimate the synthetic control so that its difference with the real colony is minimized until 1981 (*i.e.* before the Falklands war). Second, we use the 'Synthetic Falklands' as a (synthetic) control to perform a CITS and we compare our results with those obtained in the single-group ITSA.

The first step is the construction of the Synthetic Falklands. Based on the availability of data over the period 1963-1997, we identify six colonies (*donors*) that can compose Synthetic Falklands: Kerguelen Island,

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

Heard Island, Macquarie Island, and three colonies from Crozet Islands: Petite Manchotière, Crique de la Chaloupe and Ile aux Cochons. 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 the size of the colony in 1963 varies from 1 breeding pairs (Falklands Islands) to more than 200,000 at Ile aux Cochons. 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 donors 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). Weights of donors are obtained with the Stata package SYNTH[12] with the 'nested' option. Only past values of the re-scaled population 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. According to the RMSPE criterion, the best way to build a comparable synthetic control is to use a combination of two colonies: Heard Island and Macquarie Island.

Table 3: Donors weights

| Locations | Weights |
|---|---|
| Crique de la Chaloupe (Crozet Is.) | 0 |
| Heard Is. | 0.638 |
| Ile aux Cochons (Crozet Is.) | 0 |
| Kerguelen Is. | 0 |
| Macquarie Is. | 0.362 |
| Petite manchotière (Crozet Is.) | 0 |
| **Sum** | **1.000** |

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, since the synthetic control is based on two donors, this pattern must be interpreted with caution.

The second step consists in using this synthetic control in a CITS analysis. We try to replicate prior specifications as closely as possible. As we do not have comparable information on the foraging zones of king penguins breeding in Heard and Macquarie Islands, we use SST averaged over the largest foraging zone (FZ3) around each island, corresponding to a maximal distance of 722 km: the advantage of this upper limit is that it is an universally applicable estimate (i.e., it is independent from local conditions). Therefore, we

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[12]See Abadie et al. (2014).

replicate our second-best model, M2. We estimate the following segmented regression:

$$\ln brp_{it} = \beta_0 + \beta_1 T_{it} + \beta_2 real_{it} + \beta_3 post82_{it} + \beta_4 real\_T_{it} + \beta_5 war_{it} + \beta_6 SOI_{it} + \beta_7 SST_t/FZ3_{it} + \varepsilon_{it} \quad (2)$$

where $T_{it}$ is the time elapsed since the start of the study, $real_{it}$ is a variable which takes the 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 $T_{it}$ and $post82_{it}$. In this setting, $war_{it}$ is the variable measuring the effect of the war. Regarding the interpretation of coefficients, $\beta_0$ captures initial level of the log-abundance in the synthetic control (SC), $\beta_1$ indicates the slope of the synthetic control before 1982, $\beta_2$ represents the difference between Falklands and SC regarding the initial level of log-abundance, $\beta_3$ is the difference in the slope before and after 1982 for the SC, $\beta_4$ is the difference in slope between Falklands and SC before 1982, and $\beta_5$ indicates the the difference between Falklands and SC after 1982 relative to the the difference before 1982 (close to the *difference-in-differences* of trends). Finally, $\beta_6$ and $\beta_7$ are coefficient associated with climatic variables.

Table 4 reports the value of estimated coefficients for M2 as well as two other specifications as a robustness check: one only including SOI, and the other without covariates. An important prerequisite is the absence of differences between the real and the synthetic Falklands 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 -0.098 (95%CI=-0.177 – -0.019) which confirms the results we obtained, regarding the existence of a negative effect of war on king penguins' log-abundance, but regarding its magnitude (see Figure 3 for a visual comparison with results using single-group ITS).

# 5    Conclusion and discussion

All 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

Table 4: CITS with synthetic control

| | CITS with synthetic control | | |
| --- | --- | --- | --- |
| | (1) | (2) | (3) |
| $T_{it}$ | 0.163*** | 0.163*** | 0.164*** |
| | (0.00595) | (0.00670) | (0.00777) |
| $real_{it}$ | 0.745 | -0.355 | -0.355 |
| | (0.755) | (0.406) | (0.402) |
| $post82_{it}$ | -0.00412 | -0.00402 | -0.00329 |
| | (0.0155) | (0.0161) | (0.0171) |
| $real\_T_{it}$ | 0.0294 | 0.0254 | 0.0254 |
| | (0.0268) | (0.0265) | (0.0265) |
| $war_{it}$ | -0.0984** | -0.0963** | -0.0963** |
| | (0.0395) | (0.0394) | (0.0397) |
| $SOI_{it}$ | -0.00590 | -0.00615 | |
| | (0.00410) | (0.00426) | |
| $SST_t/FZ3$ | -0.305 | | |
| | (0.194) | | |
| Constant | 2.381*** | 1.359*** | 1.347*** |
| | (0.629) | (0.0460) | (0.0585) |
| | | | |
| Observations | 70 | 70 | 70 |
| R2 | 0.963 | 0.962 | 0.962 |
| AICc | 53.14 | 52.32 | 51.11 |

Newey-West wtandard errors in parentheses (maximum lag=2), with *** p<0.01, ** p<0.05, * p<0.1. (1) is the model where both local and global climate variables are considered. Value of $SST_t/FZ3$ is a synthetic measure corresponding to the average weighted average of all donors, where weights are those reported in Table 3. (2) only includes SOI as climatic variable and (3) is the model without climatic variable.

(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[13]. 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).[14] 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 impediments[15] 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.

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[13]The precise flight trajectory and the exact distance from the colony are unknown.

[14]Cooper 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.

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

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# Appendices

## A  Data appendix

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 photograph-based counts of breeding pairs made in January Delord et al. (2004)[16] or on counts of chicks made on the ground in November. 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 in November 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 pairs[17] 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).

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[16]These methods are typically used for large colonies such as the ones at Crozet archipelago.

[17]This 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" (*ibid.* pp. 251-252). For example, counts 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%. We address this issue in a sensitivity analysis based on a simulation in which each observation is multiplied by a random number ranging from 1 to 1.5 in order to replicate the main estimates on 10,000 randomly drawn alternative times series. This correction has some limitations because it assumes that counting error is random (not systematic) and is similar for counts based on adults and chicks. 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%.[18] 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 our time series, except for one impossibly high estimate in 1971 which we discarded.

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[18]A 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 sources

**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 (Otley 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.

## Data source (continued)

| Location | Year | Estimate 1 | Source | Census | Estimate 2 | Source | Census | Final count |
|---|---|---|---|---|---|---|---|---|
| Falkland Islands | 2004 | 215 chicks | Pistorius et al. (2012) | Yes[f] | | | | 269 (C) |
| Falkland Islands | 2005 | 260 chicks | Pistorius et al. (2012) | Yes[f] | | | | 325 (C) |
| Falkland Islands | 2006 | 533 chicks | Pistorius et al. (2012) | Yes[f] | | | | 666 (C) |
| Falkland Islands | 2007 | 501 chicks | Pistorius et al. (2012) | Yes[f] | | | | 626 (C) |
| Falkland Islands | 2008 | 594 chicks | Pistorius et al. (2012) | Yes[f] | | | | 743 (C) |
| Falkland Islands | 2009 | 419 chicks | Pistorius et al. (2012) | Yes[f] | | | | 524 (C) |
| Falkland Islands | 2010 | 547 chicks | Crofts and Stanworth (2019) | Yes[f] | | | | 684 (C) |
| Falkland Islands | 2011 | 472 chicks | Crofts and Stanworth (2019) | Yes[f] | | | | 590 (C) |
| Falkland Islands | 2012 | 717 chicks | Crofts and Stanworth (2019) | Yes[f] | | | | 896 (C) |
| Falkland Islands | 2013 | 610 chicks | Crofts and Stanworth (2019) | Yes[f] | | | | 763 (C) |
| Falkland Islands | 2014 | 754 chicks | Crofts and Stanworth (2019) | Yes[f] | | | | 943 (C) |
| Falkland Islands | 2015 | | | | | | | |
| Falkland Islands | 2016 | 397 chicks | Crofts and Stanworth (2019) | Yes[f] | | | | 496 (C) |
| Falkland Islands | 2017 | 632 chicks | Crofts and Stanworth (2019) | Yes[f] | | | | 790 (C) |
| Falkland Islands | 2018 | 507 chicks | Crofts and Stanworth (2019) | Yes[f] | | | | 634 (C) |
| Falkland Islands | 2019 | 815 chicks | Crofts and Stanworth (2019) | Yes[f] | | | | 1019 (C) |

Notes:

# 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 Falklands[19] whose radius corresponds to the maximal distance covered by a typical adult king penguin during one of its foraging trips.[20] 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 may jeopardize king penguins' reproductive success.[21]

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.[22] 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,

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[19]King 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.

[20]The 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.

[21]If 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.

[22]There 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: Foraging zones
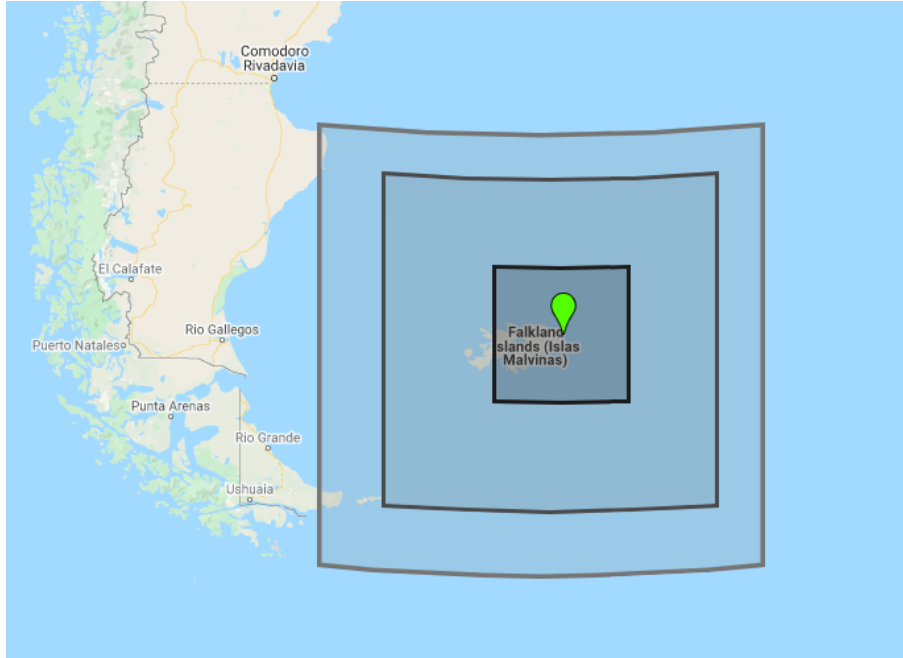


Map obtained using Google Maps. copyright notice: Map data ©2019 Google. The small-est/darkest area is FZ1, the intermediate one is FZ2, and largest/lightest one corresponds to FZ3. Each larger zone contains smaller ones. The green marker indicates the king penguin colony at Volunteer Point.

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. [23]

We converted these distances in geographic coordinates. We started from the Volunteer Point coordinates (51.47°S, 57,84°W) and we drew a square area around so that the maximal distance to Volunteer Point is 225 km (FZ1), 545 km (FZ2) and 722 km (FZ3). The size of each area is defined as follows: $Area(FZ_i) = \frac{2 \times \text{Max.Dist.}_i}{\sqrt{2}} \text{km}^2$ whereas Volunteer Point is the center of the square. After converting km 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 | 101,250km² | (50-52.9°S , 55.6-60.2°W) |
| FZ2 | 545 km | 594,050km² | (48-54.9°S , 52.7-63.9°W) |
| FZ3 | 722 km | 1,004,568km² | (46.9-56.1°S , 51.1-66.1°W) |

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[23]This 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 $2.5 \times 2.5$ 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 nests 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 Falklands[24] 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 any evidence of an epidemics affecting the Falklands' population. 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 doubt on this assumption is

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

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 CITS.

# C  Diagnostic of M1

We present here a full diagnostic of our most predictive model, M1, using current SOI and current SST in the closest foraging zone (FZ1). Diagnostics of M2-5 are very similar but omitted here.

**Stationarity**

We check the stationary of 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 there is no risk of spurious regressions. 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

|  | 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. Lag structure was determined using the Akaike's information criterion.

**Descriptive statistics**

Table A3: Descriptive statistics

|  | 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 |

Larger number of observations for climatic variables is due to the use of lags in our estimations.

**Residuals analysis**

A visual inspection of Figure A2 clearly suggests the existence of a specific pattern in the residuals distribution. First, we perform the Breush-Pagan / Cook-Weisberg test indicating the presence of heteroskedasticity (p-value= 0.000). Moreover the examination of the table of autocorrelations and partial autocorrelations also suggest 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). It mainly stems from two potential reasons: i) measurement errors in the dependent variable or ii) omission of crucial independent variables. In our sensitivity analysis, we deal with measurement errors by generating 6080 alternative time series. We obtain similar results than those exhibited with M1 meaning that measurement errors seems not to be an issue regarding the identification of the effect of war on kings penguins. The omission of independent variables is also partially addressed in our sensitivity analysis. Indeed, we estimate 243 different combinations of lags of SST and SOI, and use different sizes of foraging zones. In each situation we find a negative and statistically significant effect of war on the Volunteer Point king penguins' colony. Another point should be taken into consideration: the absence of a variable capturing human activities. To overcome this potential issue, we include the log of tones of declared fish catches in the waters of Falkland Islands (ln $catch$) in M1. Data are collected by the *Sea Around Us* project following the method developed by Zeller and Pauly (2015). Coefficient associated with $war_t$ equals -0.086 (95%CI=-0.143 – -0.028), which is a little superior to our main estimate but remains of the same order of 95%CI. Another way to deal with non-normality issues of residuals is to shorten 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). This truncated estimation has been performed as a sensitivity test and we still find negative and significant effects of war (-0.142 with 95%CI=-0.190 – -0.093). Regarding these elements, we can reasonably assume that the non-normality of residuals distribution is not crucial here. We also check the variance inflation factors to ensure the absence of multicolinearity issues. Finally, we conduct visual inspection of residuals plots against predictors without finding any systematic pattern. To conclude, one can be confident regarding the diagnostic of M1.

Figure A2: Residuals of M1 over years



43
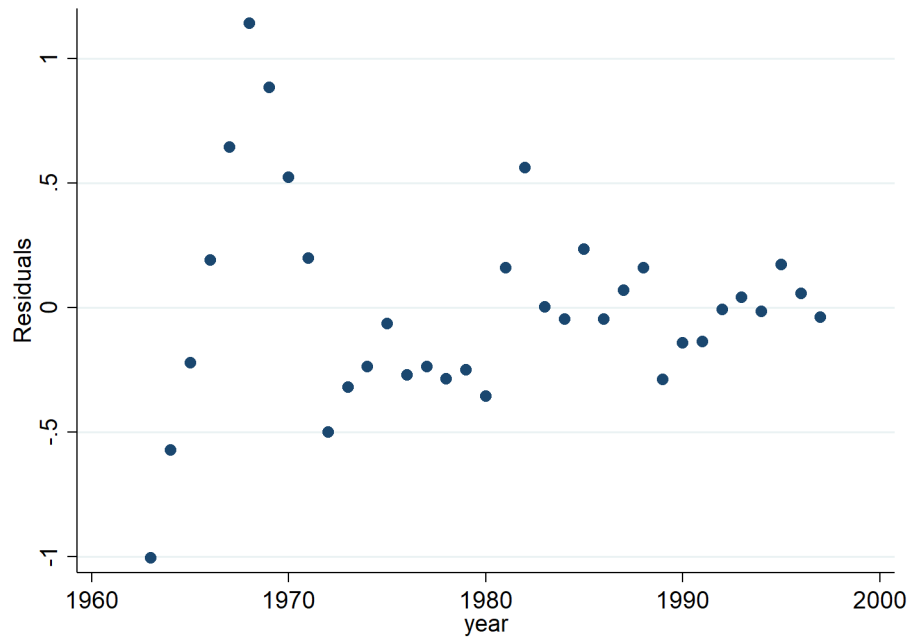
# D   Regression tables

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*** | 0.194*** | 0.194*** | 0.242*** | 0.202*** |
|  | (0.0274) | (0.0272) | (0.0273) | (0.0304) | (0.0287) |
| $war_t$ | -0.106*** | -0.103** | -0.104*** | -0.156*** | -0.114*** |
|  | (0.0377) | (0.0376) | (0.0378) | (0.0389) | (0.0391) |
| $SOI_t$ | -0.000359 | -0.00267 | -0.00209 | -0.0101 | 0.00130 |
|  | (0.00691) | (0.00678) | (0.00688) | (0.00977) | (0.00707) |
| $SST_t/FZ1$ | -0.438* |  |  | -0.734** | -0.541** |
|  | (0.238) |  |  | (0.277) | (0.222) |
| $SST_{t-1}/FZ1$ |  |  |  | -1.164* | -0.458* |
|  |  |  |  | (0.568) | (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** | 4.137** | 3.803** | 33.62** | 7.725*** |
|  | (1.498) | (1.663) | (1.665) | (14.64) | (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*** | 0.227*** | 0.227*** | 0.232*** | 0.236*** |
| | (0.0134) | (0.0147) | (0.0145) | (0.0160) | (0.0143) |
| $war_t$ | -0.142*** | -0.138*** | -0.139*** | -0.145*** | -0.149*** |
| | (0.0232) | (0.0252) | (0.0246) | (0.0268) | (0.0246) |
| $SOI_t$ | 0.00362 | 0.000555 | 0.00133 | 0.00349 | 0.00534 |
| | (0.00637) | (0.00647) | (0.00663) | (0.0106) | (0.00710) |
| $SST_t/FZ1$ | -0.560** | | | -0.511** | -0.645*** |
| | (0.222) | | | (0.200) | (0.193) |
| $SST_{t-1}/FZ1$ | | | | -0.313* | -0.348** |
| | | | | (0.164) | (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** | 4.406** | 4.186** | 5.512 | 7.085*** |
| | (1.636) | (1.835) | (1.831) | (3.801) | (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*** | 0.191*** | 0.190*** | 0.237*** | 0.198*** |
| | (0.0275) | (0.0273) | (0.0274) | (0.0304) | (0.0288) |
| $war_t$ | -0.0947** | -0.0921** | -0.0929** | -0.143*** | -0.102** |
| | (0.0359) | (0.0357) | (0.0359) | (0.0372) | (0.0371) |
| $SOI_t$ | 0.00371 | 0.00120 | 0.00187 | -0.00464 | 0.00526 |
| | (0.00546) | (0.00544) | (0.00547) | (0.00910) | (0.00581) |
| $SST_t/FZ1$ | -0.471* | | | -0.808*** | -0.568** |
| | (0.242) | | | (0.276) | (0.229) |
| $SST_{t-1}/FZ1$ | | | | -1.098* | -0.430* |
| | | | | (0.549) | (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*** | 4.623*** | 4.271** | 32.61** | 7.741*** |
| | (1.516) | (1.613) | (1.645) | (14.70) | (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. 2.

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

|  | M1 | M2 | M3 | M4 | M5 |
|---|---|---|---|---|---|
| $\Delta war_t$ | -0.232** | -0.232** | -0.232** | -0.230** | -0.231** |
|  | (0.106) | (0.106) | (0.107) | (0.104) | (0.104) |
| $\Delta SOI_t$ | 0.00497 | 0.00377 | 0.00404 | 0.00347 | 0.00580 |
|  | (0.00564) | (0.00566) | (0.00571) | (0.00585) | (0.00554) |
| $\Delta SST_t/FZ1$ | -0.381** |  |  | -0.596** | -0.539** |
|  | (0.180) |  |  | (0.271) | (0.204) |
| $\Delta SST_{t-1}/FZ1$ |  |  |  | -0.658* | -0.313 |
|  |  |  |  | (0.353) | (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*** | 0.281*** | 0.280*** | 0.293*** | 0.282*** |
|  | (0.0705) | (0.0708) | (0.0714) | (0.0692) | (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    Synthetic Control Method

## Brief description

In order to estimate the effect of the Falklands War on king penguins' population, 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 population of 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. If the synthetic counterfactual and the observed outcome diverge sufficiently, then we can assume that the treatment had an effect. In this overview, we follow 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 (1965-1981) respectively for control units and the treated one. 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 application, we have a pool of 6 control units ($J = 6$), the pre-treatment lasts 19 years ($L = 19$) which also corresponds to the number of predictors ($k = 19$).

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

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

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

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

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 $Y_0^{post}$:

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

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

$$\hat{\alpha} = Y_1^{post} - \hat{Y}_1^{post} \tag{A4}$$

Abadie et al. (2010) show that if the pre-treatment period is sufficiently long,[25] then 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: 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). Beside the Falkland Islands, we were able to gather data for six other locations: Kerguelen Islands (*Kerg*), Macquarie Island (*Macq*), Heard Island (*Hea*), and three colonies at Crozet Archipelago, Ile aux Cochons (*Ile*), Crique de la Chaloupe (*Cri*) and Petite Manchotiere (*Manch*). This sample does not include all islands populated by king penguins between 1965 and 1995[26]: we excluded South Georgia from the donor pool because combats took place on the island during the Falklands War, and had to discard the Prince Edward Islands due to the lack of data for the pre-treatment period. However, the sample is relatively representative in terms of locations, since it includes colonies from the Indian Ocean and the Pacific Ocean besides the South Atlantic. Note also that the sample is not exhaustive with regard to colonies: we have full population counts for Heard, Macquarie and Kerguelen Islands, but only counts of individual colonies from Crozet.

An interesting feature of these locations is that they lie very far away from the Falkland Islands (the closest islands, namely Crozet Archipelago, lie about 7,300 kilometers away) which means that we can avoid the contamination of the control cases by the treatment. The second advantage of the remoteness of these islands is that we can completely rule out the possibility of migration from the treated case to the donor pool. A possible drawback of this case selection is that idiosyncratic events affecting individual locations may create random shocks that we cannot account for, although we attempt to control for local conditions. Furthermore, all islands were not necessarily subject to common trends, which limits comparability.

Time series for the donor pool are based on counts of either adults or breeding pairs. We do not use data based on chick counts in order to ensure comparability of the times-series: indeed, breeding success seems to be higher in the Falklands than in most other locations (Pütz, 2002; Otley et al., 2007). While this indicates that population trends of the treated case and the donor pool are not strictly comparable, this also means that the synthetic control is likely to be an underestimate of the trajectory of the counterfactual Falklands.

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

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[25]Costalli 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, the pre-treatment period lasts 19 years.

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

locations: for example, at the beginning of our period of investigation, there were ca. 200,000 breeding pairs at Ile aux Cochons (Crozet Islands). We thus rescaled populations with the following transformation:

$$brp_t = \frac{brp_t}{\displaystyle\sum_{k=1963}^{1981} brp_k}$$

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

Figure A3: Re-scaled Populations
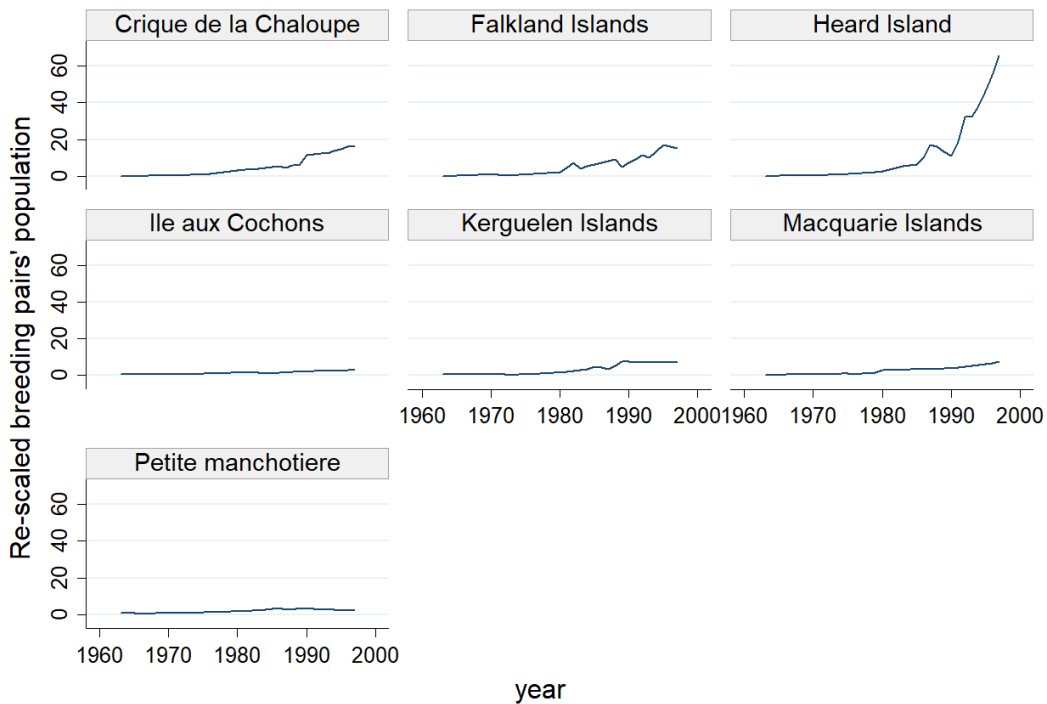


## Results

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 ($RMSPE = 22.75$), but the two curves split up after the war ($RMSPE = 230.09$). Synthetic Falklands is clearly above the real Falklands from the mid-1980s onward which suggests a delayed effect of the war. This may indicate that the effects of the war were mostly concentrated on chicks.

We use traditional permutation tests consisting in running synthetic control estimations on the six other donors to see if the effect captured here is significantly stronger than for other colonies (in-space placebo)

(Abadie et al., 2010). We find that the ratio of RMSPE before / RMSPE after 1982 is higher 4 out of 6 times, which corresponds to a "p-value" of 0.333. This number is too high to draw solid conclusions on the direct comparison of the two curves, however Figure A5 clearly shows that the magnitude of the effect find with the SCM lies between values obtained in with the ITSA and its 95%CI lower limit. Consequently, it appears to be sufficiently informative to be used in a CITS design (see Section 4.5).
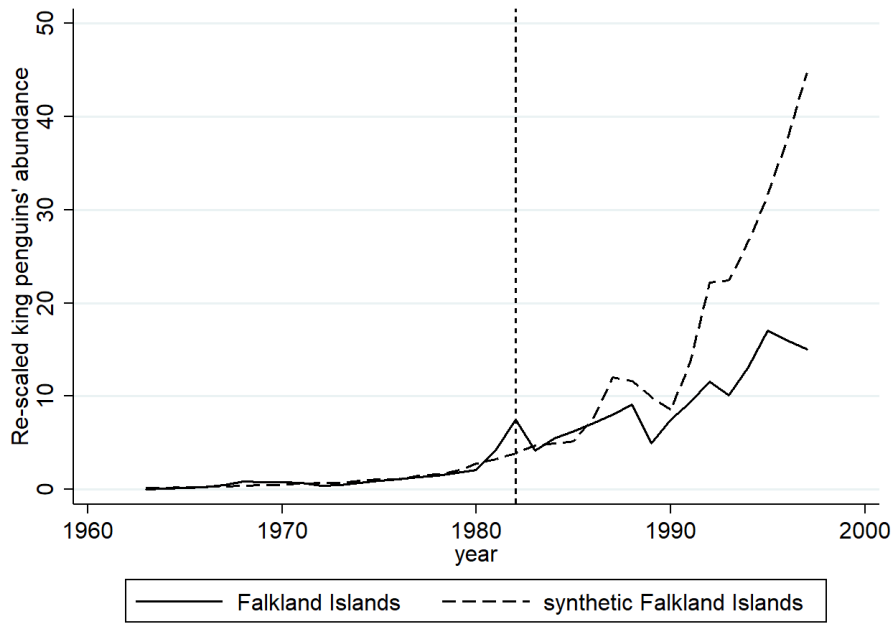
Figure A4: Falkland Islands Vs Synthetic Falkland Islands

Figure A5: Effect of wars on breeding pairs (SCM Vs ITSA)