Warfare’s ecological footprint: A Synthetic Control (SCM) approach with data from the Falkland Islands∗

Sophie Panel† Antoine Pietri‡

March 1st, 2019 - Preliminary version

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 20th century, namely the 1982 Falklands War. The fact that the conflict was unrelated to environmental issues, did not spill over to neighboring countries, and did not relapse afterwards, allows us to circumvent identification issues that commonly affect comparative studies on the ecological footprint of warfare. As an indicator of the marine ecosystem status, we analyze population trends of king penguins breeding at the Falkland Islands, which we compare to ten other penguin colonies over a thirty-year period (1965-1995). We use the Synthetic Control method – a method that was explicitly designed for small-N comparative studies – in order to build “counterfactual” Falkland Islands whose characteristics closely resemble those of the Falklands before the war and allows us to approximate the trends of the outcome variable in the absence of the war. Results indicate that the war has led to a substantial decrease of king penguins’ breeding population: by the mid-1990s, the number of breeding pairs on the islands was approximately half what it would have been in the absence of the war. We find no evidence of lasting effects, suggesting that the population decrease is attributable to direct war mortality rather than to permanent environmental damages.

JEL codes: C22, F51, Q57.

Keywords: interstate war, biodiversity, marine ecosystems.

∗We thank Mike Bingham, Boyin Huang, Valentina Marconi, and Angelique Palle for their help at various stages of this project. Our warmest thanks to Quentin David, Caroline Le Pennec-Caldichoury, and Luigi Moretti for their thorough reading and very valuable suggestions. This manuscript also greatly benefited from feedback by participants in the EP@L workshop (Lille, Dec. 2018) and Séminaire Politiques Publiques (Univ. Paris 1, Feb. 2019).
†Institute for Strategic Research (IRSEM, Paris) and Sciences Po Bordeaux. Email address: sophie.panel@scpox.fr
‡Institute for Strategic Research (IRSEM, Paris). Email address: apietri.pro@gmail.com
1 Introduction

The fact that warfare has deleterious effects on biodiversity seems self-evident but surprisingly, empirical evidence remains scant. Most statistical analyses to date have analyzed the environmental consequences of war-related economic deprivations such as poaching and deforestation, but few studies have systematically investigated whether biodiversity loss can occur as a “collateral damage” of the hostilities, e.g., as a direct consequence of the use of explosives or chemicals. This paper focuses on this latter aspect and investigates the effect of large-scale armed conflict on animal population trends.

For several reasons, analyzing the relationship between warfare and biodiversity is tricky. The first, most obvious hurdle lies in the heterogeneity of the outcome: conducting a global (or cross-regional) statistical analysis would involve comparing population dynamics of very different species under highly heterogeneous environmental conditions. Some synthetic indicators exist (such as the Living Planet Index) but they do not rely on exhaustive population censuses of all animal species in a given territory.

Second, 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).

Third, comparative studies face the difficulty of selecting 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: if the latter condition is not fulfilled, statistical analyses will underestimate the real effect size. This is especially problematic in the case of civil wars, which constitute the large majority of contemporary wars and tend to cluster in space (Ward and Gleditsch, 2002; Gleditsch, 2007; Buhaug and Gleditsch, 2008). Relatedly, the majority of today’s civil wars are repeat civil wars (Walter, 2015): studies analyzing the effects of recent outbreaks run the risk of capturing only the marginal effect of a single conflict episode on an already weakened animal population.
A final difficulty concerns the identification of the causal mechanisms at work: Gaynor et al. (2016) list no less than 24 potential causal pathways linking warfare to biodiversity, ranging from direct killings during the hostilities to the increase of wildlife poaching resulting from the long-term economic effects of the war. The problem is especially acute since most studies on this topic focus on a single conflict, which makes it hard to distinguish direct from indirect effects (as the same conflict may trigger several effects at the same time).

While these issues are serious, they can be partly addressed through the selection of the case, the indicator, and the method. With regard to the case, we chose to focus on the 1982 Falklands War between Argentina and Britain. The first reason underlying this choice is that 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.” Second, and contrary to most contemporary conflicts, the Falklands War did not result in profound institutional changes; neither did it directly cause economic collapse or large-scale labor redeployment – two consequences of wars that can indirectly affect wildlife (Dudley et al., 2002; Gaynor et al., 2016). Environmental degradations that took place can thus be directly traced back to the technology of the conflict. Finally, the Falklands War is a unique event: the Falkland Islands did not experience large-scale military conflict prior to the 1982 war, 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.

With regard to the indicator, we compare population dynamics of king penguins (*Aptenodytes Patagonicus*) at the Falkland Islands with several other penguin colonies on sub-Antarctic islands and Antarctica, before and after the war. Penguins are seabirds (and as such, they are completely reliant on sea resources) but they come ashore to breed in dense colonies at regular intervals: it is thus easier to obtain long-term data on their population dynamics than for other marine species. Furthermore, penguins, as upper-level predators, are highly sensitive to environmental changes throughout the food chain, which makes them interesting “proxies” of the marine ecosystem status (Le Maho et al., 1993; Boersma, 2008; Mallory et al., 2010; Trathan et al., 2015). Seabirds in general are commonly used to monitor environmental stress factors such as marine pollution or increases in sea temperature (Furness and Camphuysen, 1997; Durant et al., 2009;  

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1The last significant battle fought in the vicinity of the Falkland Islands occurred in 1914, as a German fleet unsuccessfully attempted to destroy the British naval base on the Falklands (Chehabi, 1985).
Penguins in particular are slower than flying seabirds (which are able to hunt for food beyond their immediate surroundings) and therefore heavily dependent on local foraging conditions during their breeding period. Furthermore, king penguins are highly specialized predators and are characterized by high breeding and foraging site fidelity (Barrat, 1976; Weimerskirch et al., 1992; Olsson, 1997; Saraux et al., 2011; Baylis et al., 2015): these features make them especially dependent on their local ecosystem and thus likely to show quick responses to war-related deterioration of the environment. In sum, penguins’ biology makes it possible to draw inferences from their population dynamics to the general effects of warfare on marine environments. Besides, penguins are a critical chain link of marine ecosystems and their population dynamics have tremendous importance for local biodiversity (Otero et al., 2018).

With regard to the method, we use a specific method specifically designed for comparative case studies: the Synthetic Control Method – hereafter ‘SCM’ – initially designed by Abadie and Gardeazabal (2003) and developed by Abadie et al. (2010, 2015). Broadly speaking, this method allows us to compare the evolution of a variable in an unit affected by a treatment relative to a set of unaffected units called ‘donors’. In our case, the treatment is the Falklands war, the treated unit is the Falklands and the donor pool is made of sites on which penguins breed. The aim of the SCM is to build a synthetic counterfactual as a linear combination of donors. This article covers the period 1965-1995 decomposed into a 17 years period of pre-treatment (1965-1981) and post-treatment period lasting 14 years (1982-1995). We build a Synthetic Falkland that correctly fits with the evolution of king penguin population over the pre-treatment period but largely differs after the Falkland wars. We find that the Falklands War had a severe short-term impact on the size of this population: the data strongly suggests that the war led to the death of breeding adults and unfledged chicks, resulting in two sharp decreases in the number of breeding pairs immediately after the hostilities and then some seven years afterwards (which roughly corresponds to the time king penguins take to reach maturity). Taken together, this shocks have substantially reduced the size of the Falklands’ breeding population. However, we find no evidence that environmental damages resulting from the war have led to a permanent slowing of the population growth rate: as our results indicate, the population has recovered quite quickly after both of these two shocks, although it never reached the level it would have reached in the absence of the conflict.

The article proceeds as follows. The next section reviews the literature on the effects of war on biodiversity. In section 3, we present our case study and detail possible causal mechanisms that drive our empirical analysis. In the fourth section we present our empirical strategy and the data used. Section 5 presents our main results and several additional tests. The last section concludes.
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, with an effect 13 times larger than El Nino Southern Oscillation (ENSO) events; 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) concluded that deleterious effects were 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 environ-
mental 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 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.

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

3.1 A brief narrative of the Falklands War

The Falklands War was a ten-week war between Argentina and the United Kingdom. It found its roots in Argentina’s sovereignty claim over the Falkland Islands starting in the early 19th century. Indeed, Britain seized these islands in 1833 (and expelled the few Argentine people who inhabited them) and they became a Crown colony in 1841. Since then, numerous disputes occurred between the two countries over the control of Falkland Islands (e.g. the Declaration on the Granting of Independence to Colonial Countries and Peoples, in which the Falkland Islands were included in 1962; in 1976, during the summit of the Non-Aligned Movement, Argentine declared herself the “legitimate owner of the territory”). Argentina’s decision to seize the islands by force is often attributed to domestic factors: in 1982, Argentina was a military dictatorship ruled by Leopoldo Galtieri, which faced intense domestic turmoil and internal factionalism (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 (Freedman, 1982; Lebow, 1983)

The war started on April 2 with the invasion of the Falkland Islands by the Argentine Navy. After a round of failed negotiations, the British Government led by Margaret Thatcher decided to go to war to restore its sovereignty on the islands. By late April, Argentina deployed 10,000 troops on the islands. In response to the invasion, Margaret Thatcher declared a war zone of 320 km around the Falklands. Hostilities started in April and stopped on June 20 with Argentina’s capitulation. They were followed by the breakdown of the military junta and Argentina’s transition to democracy in 1983.

In spite of its shortness, the war is directly responsible for killing some 1,000 military personnel and for the destruction of dozens of military warships (e.g. 2 destroyers, 2 frigates, 1 submarine, 4 cargo vessels) and hundreds of military aircrafts (mostly helicopters and fighters). The British captured some 11,400 Argentine prisoners during the war. In other words, Falklands war was a large-scale conflict and both countries had operated at their maximum ranges using a great numbers of submarines and antiship missiles (both air-to-sea and land-to-sea types). This technologically heavy conflict can be considered a crucial case to assess warfare’s ecological footprint.

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3 One of the most commonly invoked arguments was the fact that Falkland Islands lie 480 km east of the Argentine coast.
4 In March 1982, a team of Argentine metal workers illegally landed on the island and raised the Argentine flag. It is unclear whether the action was planned by the junta.
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). The drawback of this population is that it is a comparatively small one: approximately 1,000 pairs currently breed at the Falkland Islands, compared to almost half a million pairs in South Georgia and more than 600,000 at Crozet Archipelagos (Bost et al., 2013). This is a potential concern to the extent that small penguin populations exhibit more variability than large ones (Weimerskirch et al., 1992). On the flip side, king penguins have a peculiar breeding cycle (see below) and their colonies are populated throughout the year: this makes them the only penguin species that was present on the Falkland Islands as the war took place.

King penguins have a circumpolar distribution and breed in large colonies on sub-Antarctic islands between latitude 45° south and 55° south: locations include South Georgia, and Prince Edward, Macquarie, Crozet, Kerguelen and Heard Islands (Olsson, 1996; Bost et al., 2013). They are the second largest penguin species after the Emperor penguin of the same genus *Aptenodytes*. Contrary to smaller penguin species, king penguins have an unusually long breeding cycle, which lasts 14 to 15 months from courtship to chick fledging: this means that colonies are occupied continuously by chicks, and intermittently by adults (Pütz et al., 1998; Bost et al., 2013; Baylis et al., 2015). The typical breeding cycle starts with molt by the end of the austral winter, during which king penguins fast for about a month before going on foraging trips (about two weeks) to stock food reserves in anticipation of the breeding period (Olsson, 1996; Bost et al., 2013). Egg-laying occurs usually in November or December, and hatching in January/February (Bost et al., 2013). King penguins can lay one single egg per season, which they incubate on their feet without nest (Jiguet and Jouventin, 1999). During incubation, the male and the female exchange duties several times, one bird incubating while the other is foraging at sea. King penguin chicks take a long time to fledge and are unable to forage for themselves by the end of the austral summer: thus, they are left to fast at the colony during winter – for a time period that can last up to five months for some chicks – after which parents resume feeding in spring until fledging (Olsson, 1997; Bost et al., 2013).

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 virtually always fail, because the chick did not store enough food to survive the long winter fast (Weimerskirch et al., 1992; Dobson et al., 2008). 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 Y become “late breeders” at year Y+1, and again “early breeders” at year Y+2 if their previous attempt failed, which is very likely (Le Maho et al., 1993; Dobson et al., 2008). This means that the average king penguin will successfully raise one chick every second year at best (Weimerskirch et al., 1992; Le Maho et al., 1993). 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).

King penguins are specialized predators that feed mainly on myctophids (lantern fishes), whose availability is lower during the austral winter (Le Maho et al., 1993; Olsson, 1996). Food availability (i.e., its abundance and its distance from the colony) is crucial during the breeding period, because first the parents and then the chick will have to sustain long fasting periods. Food availability may impact long-term population dynamics in several ways. A lengthening of foraging trips due to decreased food availability may result in the desertion of the adult ashore and ultimately breeding failure. Alternatively, if the foraging bird returns on time, its food reserves may be insufficient to insure the chick’s survival (Le Maho et al., 1993; Olsson, 1997).

### 3.3 Possible causal mechanisms

We investigate several potential effects of the war on the population dynamics of king penguins at the Falkland Islands. To begin, there is a possibility that some birds were killed during the hostilities (e.g. by air strikes or oil spills from sunk vessels) although we do not deem it very likely. Given that the war took place between April and June, it is almost certain that unfledged chicks were fasting ashore during the hostilities: thus, the possibility exists that some of them were killed during the combats. If this was the case, this should be reflected in a decrease of the breeding populations’ growth rate with a six to eight-year delay (that is, apparent from 1988 onward) given that king penguins’ mean age at first breeding is around 6 and about 90 percent of king penguins have started breeding by age 8 (Weimerskirch et al., 1992). Note, however, that the main king penguin colony at Volunteer Point was not directly exposed to combats (Bingham, pers. comm.): if anything, chick mortality during the war should have been rather low.

With regard to adult birds, probably few of them were ashore during the hostilities since the war took place after the breeding season and during winter foraging trips. This, however, does not mean that adults

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6 Like many long-lived species, penguins have evolved to prioritize their own survival and to stop their fast and abandon their offspring before their body fuels are completely depleted (Olsson and Van der Jeugd, 2002).

7 There is variation in the age of first breeding in king penguins, with some birds starting to breed at age 3 and some others at age 8 (Weimerskirch et al., 1992). The decrease could thus be visible as early as in 1985 and last until the beginning of the 1990s.
were unaffected by the war. Baylis et al. (2015) recorded winter foraging trips of king penguins breeding at
the Falkland Islands between April and October, and found that the maximal distance to the colony for each
individual trip ranged from 127 km to 971 km, with a mean of 295 km and a median of 213 km: these latter
values are well below the limits of the Total Exclusion Zone, which covered a circle of 200 nautical miles (i.e.,
about 370 km) around the Falkland Islands. This mechanism, if at work, should translate into a short-term
decline in the number of breeding pairs in the immediate aftermath of the war.

From a theoretical perspective, the most interesting potential mechanism affecting penguin population
growth is a long-term depletion of sea resources caused by the marine and coastal pollution (e.g., oil spills)
resulting from air and naval battles. Decreased food availability may have resulted either in increased adult
mortality in the aftermath of the war or in more frequent breeding failures in the subsequent seasons (Olsson
and Van der Jeugd, 2002). Thus, this effect of the war should be reflected in a slowing of the breeding
population growth visible either upon the treatment or from 1988 onward.

Potential causal pathways are illustrated in Figure 1. Each graph compares population trends of king pen-
guins before and after the war (solid line) with a counterfactual case in which the war did not occur (dashed
line). The graph in the upper left corner displays hypothetical paths of both populations in the most simple
scenario, namely adult mortality during the war followed by a catch-up. The graph in the upper right corner
illustrates the second possible scenario: chick mortality during the war or breeding failures in the follow-
ing breeding season translate into a delayed shock (visible in 1988) and a subsequent recovery. Under the
third scenario (bottom left corner), the war has affected the marine ecosystem with long-term consequence
for penguins’ population parameters (e.g., adult survival rates and breeding success) which is reflected in
differential growth rates. Finally, the last figure illustrates a combination of these three different scenarios,
namely two shocks in 1982 (adult mortality) and 1988 (breeding failures) coupled with a long-term reduction
of the population growth rate.

3.4 Causal inference and validity issues

Our design allows us to circumvent several common threats to internal validity. To start with, we can
safely assume that reverse causation is not a concern in our case. More generally, since the Falklands War
was in all likelihood exogenous to local environmental conditions, we can also exclude the possibility that
some unobserved environmental factor simultaneously triggered a change in penguin population trends and
the outbreak of the war.
A related advantage of this case is that we can evaluate the direct effects of war technology on biodiversity. The literature reviewed above has highlighted three causal pathways from war to biodiversity loss - deliberate tactics to destroy the environment (as in the oft-mentioned Vietnamese case), indirect effects from the economic and political consequences of the war, and collateral damage from the use of firearms - the last pathway being relatively understudied. In the case of the Falklands War, the first two mechanisms can be ruled out. With regard to the indirect effects of warfare, there is no evidence that conservation became ineffective after the conflict, or even that the local economy declined as a consequence of the war (Royle, 1994) Likewise, we found no report of deliberate tactics aiming at causing environmental damages. In other words, this case selection fulfills the criteria of a “pathway case” (Gerring, 2007): first, the general relationship between warfare and biodiversity is well-established but causal pathways are unclear; second, the case features potential factor $X_1$ (war technology) but not alternative factors $X_2$ and $X_3$ (war tactics and long-lasting economic repercussions); finally, there is causal equifinality of $X_1$, $X_2$ and $X_3$, which means that the effect of $X_1$ is sufficient to trigger outcome $Y$ independently of $X_2$ and $X_3$.

External validity is more of a concern. The Falklands War can be considered representative of a broader class
of events, namely conventional warfare without deliberate tactic to destroy the environment. However, this case arguably takes on extreme values on the independent variable (i.e., the technology deployed). Likewise, if seabirds are unusually sensitive to environmental degradation, results may generalize poorly to other species. In this regard, the Falklands can be viewed as a most-likely case that displays a high probability of featuring the hypothesized relationship given the values of the explanatory and outcome variables (Levy, 2008). Therefore, observing this relationship in our case would provide tentative evidence in support of the general hypothesis that the use of war technology has detrimental effects on biodiversity (probing the generalizability of the findings would then be left to future research). By contrast, an absence of relationship would provide compelling evidence against the general hypothesis: if the Falklands War had no lasting effect on the local environment, it is unlikely that smaller-scale conflicts can have such an effect. Importantly, however, such a result would not constitute a rebuttal of the well-established fact that war is harmful for biodiversity, but only rule out one particular explanation of this relationship.

4 Empirical strategy

King penguins have a circumpolar distribution and consistently breed on several sub-Antarctic islands: 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). We excluded South Georgia from the donor pool because combats took place on the island during the Falklands War. We also had to discard the Prince Edward Islands due to the lack of data for the pre-treatment period. Thus, beside the Falkland Islands, we were able to gather data for six king penguin colonies: 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). 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. A possible drawback is that idiosyncratic events affecting individual locations may create random shocks that we cannot account for, although we attempt to control for local conditions.

In order to expand the donor pool, we gathered data on five additional colonies, which are home to different penguin species: Pointe Géologie (Geo), Cape Royds (Roy), Litchfield Island (Litch) and Ongulkalven (Ong). The first of these colonies is populated by Emperor penguins (Aptenodytes forsteri), and the three remaining ones by Adelie penguins (Pygoscelis adeliae). All of these are located in Antarctica.\(^8\) Thus, our

\(^8\) As we will discuss in further detail below, the findings are not biased by differences between species: we obtain similar results when we restrict the sample to king penguin colonies.
4.1 Overview of the synthetic control method

In order to estimate a causal effect of the Falklands War on 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 king penguins). The general idea is to build a ‘synthetic’ counterfactual unit (‘Synthetic Falklands’) using a weighted combination of observed control units which are not exposed to the treatment. If the synthetic counterfactual and the observed outcome diverge sufficiently, then one can assume that the treatment has an effect.

Let $I$ be a set of eleven locations such as $I = \{Falk, Kerg, Macq, Hea, Ile, Cri, Manch, Geo, Roy, Litch, Ong\}$ and $Y_i = (Y_{i,65}, \ldots, Y_{i,95})$ is a vector of observed penguin populations in location $i$ during the period 1965-1995. The observed outcome can be decomposed in a treatment-free part, $Y_{i,t}^N$ and the effect of the treatment $\alpha_{i,t}$:

$$Y_{i,t} = Y_{i,t}^N + \alpha_{i,t},$$  \hspace{1cm} (1)

The treatment is the exposure to the Falklands war in 1982 and, considering the geographic coordinates of donors, one can assume that Falklands are the only locations affected by the war such that

$$\alpha_{i,t} \begin{cases} 
\neq 0 & \text{if } t \geq 82 \text{ and } i = Falk \\
= 0 & \text{otherwise.} 
\end{cases}$$  \hspace{1cm} (2)

Consequently, $\alpha_{Falk,t}$ is the effect of the treatment on our outcome variable. By definition, the treatment effect is not directly observable and requires building of a counterfactual. If the pre-treatment period is sufficiently long, Abadie et al. (2010) show that the treatment effect can be estimated such that, $\forall t \geq 1982$

$$\hat{\alpha}_{Falk,t} = Y_{Falk,t} - \sum_{i \neq Falk} w_i Y_{i,t},$$  \hspace{1cm} (3)

where $\hat{\alpha}_{Falk,t}$ is the estimated effect of treatment for Falklands and $w_i$ is the vector of individual weights representing the contribution of each donors to the synthetic control $\left(\sum_{i \neq Falk} w_i = 1\right)$. Weights that form Synthetic Falklands are computed by minimizing the pre-treatment difference (i.e. over the period 1965-1981) in the outcome variable between Falklands and its Synthetic counterpart. Abadie et al. (2010) show
that if the pre-treatment period is sufficiently long,\textsuperscript{9} then the SCM offers an unbiased estimation of the counterfactual trend of the outcome variable in the treated unit. The difference between the Falklands and Synthetic Falklands is therefore an unbiased estimation of the effect of the Falklands War on the population of king penguins.\textsuperscript{10}

4.2 Data

4.2.1 Outcome variable

Yearly population estimates on king penguins come from a variety of sources. 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). Second, we completed these estimates by thoroughly reviewing scholarly articles and grey literature.

One difficulty is that population counts or estimates are given in different units from one source to another. Most estimates are in breeding pairs (e.g., those from the Living Planet Index) or adult birds, but some of our data points for the Falkland Islands are either in small chicks or in fledglings (i.e., large chicks that survived the winter fast). We transformed adult population or chicks estimates in numbers of breeding pairs (our unit of analysis) using the following rules. The number of adults was converted into numbers of breeding pairs assuming a 15% rate of non-breeders (Heezik et al., 1994; Jiguet and Jouventin, 1999; Le Bohec et al., 2007) with the exception of data provided by Paleczny et al. (2015).\textsuperscript{11} Converting numbers of chicks into numbers of breeding pairs is trickier because of the peculiarities of king penguins’ breeding cycle. Generally, the number of small chicks counted early in the year approximates the number of breeding pairs present at the beginning of the current breeding season (Lewis Smith and Tallowin, 1979). The fledglings/breeding pairs ratio is more variable depending on environmental conditions and food availability before and during the winter fast. We thus decided to discard data on large chicks, because fledgling numbers are determined not only by the number of breeding pairs but also by breeding success: since one of our hypotheses is that the Falklands War has impacted king penguins’ breeding success, using this data would bias our estimates. We kept data on small chicks,\textsuperscript{12} which we converted into numbers of breeding pairs assuming a 20% early

\textsuperscript{9}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 17 years.

\textsuperscript{10} It should also be noted that, contrary to difference in differences models, the SCM “allows the effects of confounding unobserved characteristics to vary with time” Abadie et al. (2010, p. 495).

\textsuperscript{11} Paleczny et al. (2015) converted estimates given in breeding pairs in adult population estimates assuming one-third of non-breeding birds; we therefore converted these numbers back to their original unit.

\textsuperscript{12} We classified as “small chicks” those counted between March and the beginning of the austral winter. Counts made in January or February are discarded (unless information about the development stage of the chicks is explicitly provided) as these numbers may include small chicks just born or large chicks born in the previous breeding season (Lewis Smith and Tallowin, 1979)
mortality rate (Bingham, 1998). In case several counts were available for a given location-year, we computed the geometric mean to obtain one single estimate. We excluded outliers, e.g. cases in which the growth rate between two consecutive years was above 100%. Finally, in order to deal with missing data, we interpolated the population assuming an exponential\textsuperscript{13} growth.

Figure 2 displays our data for the Falkland Islands (including information on original and interpolated data points) for the whole investigation period (see the Appendix for similar information on donors). These data provide some interesting information, but with important caveats. First, population trends are clearly increasing in the pre-treatment as well as in the post-treatment periods (with a mean growth rate of about 20\% per year), but this does not exclude the possibility that the war has slowed the population growth rate: the empirical analysis will help clarify this question. Second, the two shocks occurring respectively in 1982/3 and in 1989 are consistent with immediate and delayed effects of the war:\textsuperscript{14} the population decrease starting in 1982 may reflect war mortality among adult birds, while the 1989 decrease of the breeding population may result from chick mortality seven years earlier. These patterns, however, must be interpreted with great caution. Our information on the 1982/3 decreasing trend is limited by the fact that the data is interpolated for the year 1982: thus, we cannot ascertain whether the shock occurred before or after the treatment. Furthermore, since data points preceding 1981 are also interpolated, the decreasing trend may have started earlier. We face a similar issue with regard to the delayed shock, since data are interpolated for the year 1987. Moreover, and although this pattern is consistent with chick mortality, we would have expected a smoother trend since king penguins do not always start breeding at the same age (Weimerskirch et al., 1992).

In order to obtain our outcome variable, we perform a final transformation of the raw data. Indeed, king penguins populations do not start from the same level in all considered locations. At the beginning of our period of investigation (1965), there were four breeding pairs at the Falkland Islands compared with 216,588 at Ile aux Cochons (Crozet Islands). Consequently, we can neither use as outcome variable the number of breeding pairs – which is too heterogeneous – nor the growth rate which is more volatile in small populations. We thus decided to choose, for each location $i$ in year $t$, the index of breeding pairs base 100 in 1980:\textsuperscript{15}

\[ I_{soi,t} = \frac{Pop_{i,t}}{Pop_{i,80}} \times 100 \]

\textsuperscript{13} We assume that penguin population growth follows a geometric pattern such that: $Pop_t = (1 + r)Pop_{t-1}$, where, $r$ is the annual rate of growth between the two considered years. Consequently, if we have data on $t_1$ and $t_n$, we can rebuild the missing data by considering that the annual rate can be estimated by:

\[ r_{geo} = \left( \frac{Pop_{t_n}}{Pop_{t_1}} \right)^{1/n} - 1 \]

\textsuperscript{14} We do not have any explanation for the last, smaller shock occurring in 1993.

\textsuperscript{15} The choice of 1980 rather than 1981 is data driven: we have more reliable data for 1980.
4.2.2 Predictors

The choice of our predictors is determined by current scholarship on factors affecting penguin population trends. The main threats to king penguins are, first, those related to climate change, such as climate variability (Le Bohec et al., 2008; Boersma, 2008; Barbraud et al., 2012; Trathan et al., 2015) and changes in sea surface temperatures (Le Bohec et al., 2008; Saraux et al., 2011; Barbraud et al., 2012; Bost et al., 2013; Trathan et al., 2015); and, second, those related to human footprint, including human settlements (Trathan et al., 2015), marine pollution (Boersma, 2008; Trathan et al., 2015), or commercial fishing (Boersma, 2008; Barbraud et al., 2012; Bost et al., 2013; Trathan et al., 2015). These variables mostly impact population trends through food availability and subsequent changes in breeding parameters.

We mostly use indicators of human activities, temperature measures and geographic characteristics of each location, and each species’ population parameters. First, in order to capture the impact of human activities, we built a variable identifying the existence of permanent human presence on the location. Indeed, if Falkland Islands is the only populated location (around 2 000 inhabitants over the period 1965-1995), all locations have a scientific base since the 1950s. Based on information of the different bases (and on their expansion) retrieved from their official websites we are able to build the following variable:

\[
HumanPop = \begin{cases} 
2 & \text{if the permanent population is superior to 50}, \\
1 & \text{if the permanent population is between 1 and 49}, \\
0 & \text{otherwise}.
\end{cases}
\]
Human activities is also captured by a log of tones of declared fishes catches in the water of each location \((\text{lnCatch})\). Data are collected by the Sea Around Us project following the method developed by Zeller and Pauly (2015). Finally, we also proxy human footprint with the log of each location’s distance to the nearest capital city in kilometers \((\text{lnDist})\).

With regard to temperatures, we use data on sea surface temperature (SST) from the Extended Reconstructed Sea Surface Temperature (ERSST), which is a global, spatially gridded \((2^\circ \times 2^\circ)\) dataset with monthly resolution (Huang et al., 2017). Using the geographic coordinates of each location, we first compute mean SST by year \((\text{Mean\_SST})\). We then compute the difference between this yearly average and the mean SST over the 1965-1995 period for each location \((\text{Dev\_SST})\): this variable captures anomalies in sea temperatures on an annual basis.

Finally, we include in the predictors list information on each species’ population parameters, in order to account for the fact that we work with heterogeneous populations with different intrinsic growth rates. These parameters are, first, adult annual survival rates \((\text{Survival})\) which we set at 0.95 for emperor penguins, 0.93 for king penguins, and 0.89 for Adelie penguins (Croxall and Davis, 1999); second, the mean age at first breeding \((\text{Firstbreed})\), respectively 5, 6 and 4.85 years); and third, the maximal number of eggs \((\text{Maxeggs})\) laid by each species in one breeding season: 1 for emperors and kings, and 2 for Adélies. We do not include information on breeding success, since the rates vary heavily across sites and periods (Weimerskirch et al., 1992; Otley et al., 2007) and are likely to be influenced by the treatment.

5 Results

5.1 Main results

As mentioned above, we build the Synthetic Falklands as a combination of locations that most closely resemble the Falkland Islands during the pre-treatment period. Among the ten locations composing the donor pool, four are used to form the Synthetic Falklands (see Table 1). Not unexpectedly, the synthetic control is exclusively made of sites that are home to king penguins: one of the Crozet colonies \((\text{Crique de la Chaloupe})\), Heard Island, and Macquarie Island take on the most important weights, which may result from the fact that their population size is relatively close to the one of the Falklands in absolute terms. The fourth donor \((\text{Kerguelen Islands})\) has a larger population but its weight in the synthetic control is almost negligible.
Table 1: Donors weights

<table>
<thead>
<tr>
<th>Location</th>
<th>Weight</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cape Royd</td>
<td>0</td>
</tr>
<tr>
<td>Crique de la Chaloupe (Crozet Is.)</td>
<td>0.418</td>
</tr>
<tr>
<td>Heard Is.</td>
<td>0.324</td>
</tr>
<tr>
<td>Ile aux Cochons (Crozet Is.)</td>
<td>0.045</td>
</tr>
<tr>
<td>Litchfield Is.</td>
<td>0</td>
</tr>
<tr>
<td>Macquarie Is.</td>
<td>0.213</td>
</tr>
<tr>
<td>Ongulkalven</td>
<td>0</td>
</tr>
<tr>
<td>Petite manchotière (Crozet Is.)</td>
<td>0</td>
</tr>
<tr>
<td>Pointe géologie</td>
<td>0</td>
</tr>
<tr>
<td>Sum</td>
<td>1.000</td>
</tr>
</tbody>
</table>

Table 2 presents and compares the characteristics of the predictors mentioned above. In addition, three other predictors have been used in order to improve the fit in the pre-treatment period. First, following Abadie et al. (2010) we use in the SCM our outcome variable, Penguins, at the beginning and at the end of the treatment. We also include the value of the year 1973, which is the midpoint of our pre-treatment period.

Table 2: Predictors values for Falklands vs. Synthetic Falklands

<table>
<thead>
<tr>
<th></th>
<th>Falkland Is.</th>
<th>Synthetic</th>
</tr>
</thead>
<tbody>
<tr>
<td>$I_{s0}(1965)$</td>
<td>4.249753</td>
<td>7.748908</td>
</tr>
<tr>
<td>$I_{s0}(1973)$</td>
<td>13.19682</td>
<td>24.02375</td>
</tr>
<tr>
<td>$I_{s0}(1981)$</td>
<td>128.7052</td>
<td>114.0932</td>
</tr>
<tr>
<td>lnCatch</td>
<td>11.44039</td>
<td>3.796783</td>
</tr>
<tr>
<td>HumanPop</td>
<td>2</td>
<td>0.92</td>
</tr>
<tr>
<td>lnDist</td>
<td>7.535297</td>
<td>8.009445</td>
</tr>
<tr>
<td>Mean _ SST</td>
<td>6.641716</td>
<td>4.335399</td>
</tr>
<tr>
<td>Dev _ SST</td>
<td>-0.0887411</td>
<td>-0.1450599</td>
</tr>
<tr>
<td>MaxEggs</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td>Survival</td>
<td>0.93</td>
<td>0.93</td>
</tr>
<tr>
<td>FirstBreed</td>
<td>6</td>
<td>6</td>
</tr>
</tbody>
</table>

As can be seen in Table 2, Falklands and Synthetic Falklands are relatively close in terms of predictor values but differ in a few of them (albeit not dramatically). First, the Falkland Islands are the only permanently inhabited location – other than scientific or military bases. Consequently, human activity is necessarily more developed on the Falklands than in its Synthetic (see HumanPop and lnCatch values in Table 2). Second, they were very few king penguins on the Falklands at the beginning of the pre-treatment period (6 breeding pairs observed in 1966) contrary to other locations included in the Synthetic Falklands\(^{16}\) which explain the

\(^{16}\) For example, 4,000 breeding pairs or more were observed in 1965 at Petite Manchotière, Macquarie Islands and Kerguelen Islands
difference observed for the variable $ln(Av\_Pen)$.

Figure 3 displays the central result of our paper. The solid line represents the ‘treated unit’, and the dashed one stands for Synthetic Falklands. The treatment (i.e., the Falklands War) is highlighted by a vertical dashed line for the year 1982. First of all, there is a good fit between Falklands and Synthetic Falklands during the pre-treatment period. The goodness of fit of the SCM is measured by the root mean squared prediction error (RMSPE). A small RMSPE means that the penguin population in the Falklands during the pre-treatment period is correctly replicated by its Synthetic. In our case, we have a $RMSPE = 8.406418$ which is relatively small regarding to our unit of analysis, it provides confidence for the interpretation of our results.

Several remarkable patterns stand out on Figure 3. First, the population decrease taking place in 1982/3 is visible only in the actual, and not in the synthetic Falklands, which provides support for the hypothesis of war mortality among breeding adults. However, a remarkable catch-up effect takes place in the immediate aftermath of this shock, and before the second shock occurring in 1989. For this five-years periods (1984-1988),
the annual population growth rate is even higher in the actual Falkland Islands (about 22%) than in the synthetic Falklands (18%)\(^\text{17}\). In other words, it seems that the war had a positive impact on king penguins’ population growth in the short run. There are two possible explanations for this unexpected result. First, the Falklands War brought about a sharp decrease in the number of tourists visiting the Falklands until the end of the 1980’s (Riley, 1995). Second, landmines laid on beaches by the Argentine military at the outset of the war may have had the unintended consequence of protecting penguin colonies from human disturbances\(^\text{18}\). Thus, this result can be interpreted as evidence of a short-term “refuge-effect” (Dudley et al., 2002; Gaynor et al., 2016). However, this trend must be interpreted with great caution, given the brevity of the period under consideration.

The second shock occurring in 1989 (attributable to chick mortality) has led to a more dramatic decline in the breeding population: following this shock and the subsequent one occurring in 1993, the difference between the counterfactual and the observed populations reaches substantial levels during the 1990s. Even barring 1993 – which is an outlier with regard to the overall trend – the population is on average about 2.15 times higher in the synthetic Falklands than in the actual Falklands during the first half of the decade.

These differences between actual and counterfactual population seem to be driven by the two one-off shocks rather than by a long-term change in population parameters (e.g., increased adult mortality rate or lowered breeding success): recall that the growth rate is higher in the real Falklands during the early pre-treatment period. After 1990, the synthetic population displays a more sustained growth than the actual population (19% vs. 15% between 1990 and 1995) but again, given the shortness of the period, it is difficult to draw valid inferences. Overall, there is no conclusive evidence that the Falklands War has permanently affected king penguins’ population parameters.

### 5.2 Placebo tests

In order to assess the credibility of our results we conduct two placebo tests. First, we reassign the treatment of interest to a year other than 1982 (Abadie et al., 2015). We restrict the sample to the period 1965-1981 and set the treatment to 1975: we can see in Figure 4 that a shock in 1975 seems to have no

\(^{17}\) Thanks to the weights computed in Table 1, these numbers are found by computing the value of the outcome variable for each year.

\(^{18}\) See for example [https://www.bbc.com/news/magazine-39821956](https://www.bbc.com/news/magazine-39821956) The fact that warfare, by reducing human activities, may favor biodiversity is not new in the literature, as discussed above (see Section 2). Similar phenomena have been observed for other animal populations, see e.g. the case of the Persian leopard in the northern Iran-Iraq frontier. See [https://news.nationalgeographic.com/news/2014/12/141219-persian-leopard-iran-iraq-land-mine/](https://news.nationalgeographic.com/news/2014/12/141219-persian-leopard-iran-iraq-land-mine/).
effect in the sense that the curve of the Falklands and synthetic Falklands remain close to each other. This increases our confidence that our findings are not a mere statistical artifact. It should be noted that due to the placebo test procedure, our pre-treatment period is restricted to 11 years which can partly reduce the reliability of the results.\textsuperscript{19}

\begin{figure}[h]
\centering
\includegraphics[width=\textwidth]{figure4}
\caption{In-time Placebo Falklands war (1975)}
\end{figure}

We also conduct a second placebo test initially suggested in Abadie et al. (2010). The general idea is the following: by comparing the difference of synthetic Falklands and Falklands with the differences of other synthetic donors and their observed values, we can evaluate if the results observed in Figure 3 are large enough or could have been obtained by chance alone. This placebo tests is passed if i) trends in other placebo sites are random (\textit{i.e.} we should have positive and negative differences) and ii) if the difference between synthetic and actual values of the treated case is greater than others (Abadie et al., 2010, 2015). As can be seen in Figure 5, both conditions are fulfilled in our case: first, differences between synthetic and actual values follow no discernible pattern among donors (grey lines). Second, the Falklands Islands (black line) display the most extreme value with the exception of one (positive) outlier.

\textsuperscript{19} However, it is superior to the 10 years pre-treatment period recommended in Costalli et al. (2017).
5.3 Other tests

We run two other tests: the “leave-one-out” test distribution and the ratio of the post-treatment Root Mean Squared Prediction Error (RMSPE) to the pre-treatment RMSPE. The first one is a sensitivity test introduced by (Abadie et al., 2015). It consists in iterating over the model to leave out one real donor (i.e. a donor with a strictly positive weight in the Synthetic Falklands) each time to be sure that the result obtained is not fully driven by one single donor. Concretely, if the new synthetics (labeled ‘Other synthetics’ in Figure 6) are very different from the original Synthetic Falklands, it means that results are not very robust. In our case, it should be noted than when we dropped Heard Islands, the algorithm was no longer able to compute a synthetic. Consequently Figure 6 only presents three new synthetics. Our main results seem robust to the leave-one-out test: although the magnitude of the difference between synthetic and actual population varies quite strongly, the population of the real Falkland Islands is smaller than all synthetic populations, and the original synthetic (including all donors) never takes on extreme values.

Second, following Abadie et al. (2010), we also compute the ratio of the post-treatment RMSPE to the pre-treatment RMSPE in Figure 7. This procedure is a different way to evaluate how large the difference between the Falklands and their synthetic is relative to other donors. In the best-case-scenario, the ratio of Falkland
Islands should be superior to all others. However, three locations present a higher ratio: Heard Islands, Ile aux cochons and Petite manchotière. This result means that these locations have been impacted by strong idiosyncratic shocks in the post-treatment period. It could be questioning for our central result in the sense that Synthetic Falklands depends on Heard Islands (weight: 0.324). Consequently, it draws our attention to the need to increase the number of donors to eliminate the risk of idiosyncratic shocks in a future version of this work.
6 Conclusion

To the best of our knowledge, this paper is the first attempt to evaluate the effect of conflict on biodiversity using the SCM. We built “counterfactual” Falkland Islands as a linear combination of characteristics of other locations in the Antarctic and Sub-Antarctic areas, in order to assess the existence of a causal effect of the Falklands War on marine biodiversity. Our results show that on the whole, the war had a substantial, negative effect on the development of the population under study. There is some tentative evidence that the war temporarily boosted king penguins’ population growth (which might be due to a temporary decrease of human disturbances), but this short-term effect is more than offset by two sharp population decreases occurring at the beginning and at the end of the 1980s – which we respectively attribute to adult mortality and chick mortality during the war. Our results indicate that the population of king penguins breeding on the Falkland Islands would have been more than twice as high at the end of our investigation period if the war had not occurred.

On the flip side, we do not find evidence that the war has had indirect, long-term effects on king penguins population dynamics by permanently deteriorating the local ecosystem. The data does not support the hypothesis of a lasting decrease of population growth rates after the war: the difference between synthetic and actual population sizes seems fully attributable to one-off shocks, and we find evidence of an immediate recovery after both of these shocks.
One limitation of the present study is the fact that the Falklands may be considered an outlier relative to the donor pool. Indeed, it is the only inhabited island (around 2,000 inhabitants in our investigation period), which can create noise in our analysis. In order to circumvent this issue, a next version of the paper we will use South Georgia as another treated unit. Indeed, South Georgia was also directly impacted by the Falklands War (albeit less intensively) but has the advantage of being unoccupied by civilians. Therefore, adding a second treated unit will allow us to obtain more fine-grained results on the impact of the Falklands War. Moreover, we will also try to address the question of how the intensity of fighting have impacted local biodiversity by comparing results from South Georgia with Falklands’ ones. Last, we will also increase the number of donors up to thirteen (instead of ten in the present version) to alleviate issues related to idiosyncratic shocks in the donors pool.
References


Appendix

Appendix A

Figure 8: Population dynamics of penguins in the donor pool
7.2 Appendix B

Figure 9: Index of breeding pairs (base 100, 1980) – all sites