The Social Costs of Keystone Species Collapse: Evidence From The Decline of Vultures in India

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Abstract

Scientific evidence has documented that we are undergoing a mass extinction of species, caused by human activity. However, allocating conservation resources is difficult due to scarce evidence on damages from losing individual species. This paper studies the collapse of vultures in India, triggered by the expiry of a patent on a painkiller. Our results suggest the functional extinction of vultures—efficient scavengers who removed carcasses from the environment—increased human mortality by over 4% because of a large negative shock to sanitation. We quantify damages at $69.4 billion per year. These results suggest high returns to conserving keystone species such as vultures.

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1 Introduction

> "[D]isgusting" - Charles Darwin, observing a vulture off the deck of the *Beagle* in 1835

We are in the midst of the sixth mass extinction in the history of the planet, likely induced by human activity (Ceballos et al. 2015). Since 1900, 477 vertebrate species have become globally extinct in the wild, at a rate about a hundred times higher than the 'background' level estimated between the five previous mass extinctions (Pimm et al. 2014; Jaureguiberry et al. 2022). Local extinctions, where a species disappears from the wild in a part of the world, are even more common (Kuussaari et al. 2009; Wan et al. 2019). Well before local extinction, severely deteriorated wildlife populations may no longer be capable of filling their role in the ecosystem—resulting in what ecologists refer to as "functional extinctions" (Valiente-Banuet et al. 2015; Carmona et al. 2021).

These facts set the stage for a thorny policy challenge. Wildlife levels can collapse quite rapidly, with trajectories that are difficult to predict or reverse. Curtailing or regulating economic activity, or investing in conservation initiatives, might protect or restore some species populations. Unfortunately, since it is impossible to prevent every extinction, conservation policy must solve a crucial targeting problem—which of the many endangered species should we protect or restore? This question is difficult to answer because although biodiversity loss is arguably damaging in general (Cardinale et al. 2012), estimates of the effects of losing *specific* species on human well-being are sparse.[1] Despite this lack of evidence, several policies focus on preventing the extinction of species. In the United States, leading examples are the Endangered Species Act, Marine Mammals Protection Act, Migratory Bird Treaty Act, and the

[1] In contrast, we know much more about the impacts of non-biological aspects of the environment, such as the costs of pollution (Chay and Greenstone 2003; Currie and Walker 2011; Ebenstein 2012; Zivin and Neidell 2012; Schlenker and Walker 2015; Currie et al. 2015; Ebenstein et al. 2017; Deryugina et al. 2019; Keiser and Shapiro 2019; Marcus 2020), or changes in weather conditions (Schlenker et al. 2006; Deschênes and Greenstone 2007; Deschênes et al. 2009; Schlenker and Roberts 2009; Dell et al. 2014; Costinot et al. 2016; Fujiwara et al. 2016; Hsiang et al. 2017; Proctor et al. 2018; Corno et al. 2020; T. A. Carleton et al. 2022).

Magnuson–Stevens Fishery Conservation and Management Act, with similar laws passed in other countries—for example, Natura 2000 in the European Union. Globally, nations have committed to the goal of preserving biodiversity by signing the Convention on Biological Diversity, and establishing the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.[2] Without species-specific evidence of damages from extinction, policymakers find themselves in the undesirable situation of having to allocate scarce resources towards a few lucky winners with little sense of the magnitude or even sign of the social benefits of their choices.

The costs of species extinction are hard to estimate for several reasons. First, the effect of a catastrophic collapse cannot in general be recovered by studying the impact of marginal changes.[3] Second, causal evidence is hard to produce because we often possess very little data on species population counts and experimental estimates are unavailable because manipulating ecosystems can be both unethical and infeasible (Frank and Schlenker 2016; Ferraro et al. 2019). Third, the number of potentially endangered species is large, forcing us to target not only conservation, but also evaluation efforts.

In this paper, we study the sudden and catastrophic collapse of vulture populations across the Indian subcontinent, making progress on all three fronts. First, we use a local functional extinction to study the costs to society of a catastrophic collapse of vultures in India, caused by the introduction of the painkiller diclofenac to treat cattle. The disappearance of vultures resulted in the loss of sanitation services that these birds had previously provided through scavenging dead livestock. We provide evidence of a meaningful increase in human mortality after vultures died out and were no longer removing carcasses from the environment. Although this analysis is retrospective, local functional extinctions are more easily reversed than global extinction in the wild, en-

[2] The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services is to biodiversity as the Intergovernmental Panel on Climate Change is to climate change.

[3] Economic theory has long recognized the conceptual and practical difficulties involved in carrying out a forward-looking cost-benefit analysis in the presence of uncertainty, irreversibility, and catastrophic tail risks (Arrow and Fisher 1974; Weitzman 2009).

abling evidence of this type to constructively influence conservation policy in extinction areas, and protection of vultures in parts of the world where they still provide scavenging services.

Second, we overcome the causal inference challenges associated with estimating social costs by drawing upon empirically and theoretically grounded measures of habitat suitability developed by ecologists. Specifically, we use a differences-in-differences approach comparing changes in mortality in areas with habitats that had high vs low vulture-suitability, before and after a near-total decline in bird populations due to an unintentional, unexpected, and rapid poisoning event in which vultures became exposed to the painkiller diclofenac. Habitat definitions in this setting provide an indicator for regions where the population change is expected to have been large.[4] We find that districts that were highly suitable to vultures saw an average increase in all-cause human death rates of 4.7% in the years following their sudden collapse. This number is measured relative to areas that were always poorly suited to vultures and thus much less affected. Our results hold up to multiple robustness checks and specifications, and to an alternative triple-difference approach that exploits the fact that negative effects are likely to be concentrated in districts that had *both* vultures and large livestock populations. The effect size we obtain implies an average of 104,386 additional deaths a year relative to a population of 430 million people in our main sample. Using an India-specific value of statistical life of $665,000 (Nair et al. 2021), this implies mortality damages of $69.4 billion per year.

Lastly, the example of vultures suggests that one way to target evaluation, conservation, and protection efforts is to focus on what are known as *keystone species*—those that help "hold the [eco]system together."[5] Keystone species are seen as being crucial to the functioning of an ecosystem, sometimes providing unique services, such that if they are removed, the effects on the ecosystem are potentially large (Paine 1969; Power et al. 1996; Hale and Koprowski 2018).

[4] In Appendix Section A.2 we provide evidence from 376 bird species that habitat suitability measures are indeed a strong predictor of population.

[5] A short National Geographic explanation of keystone species is available online. URL: https://education.nationalgeographic.org/resource/keystone-species.

In India for instance, vultures have provided critical environmental sanitation services. The 2019 livestock census in India reported a population of over 500 million animals, more than any other country in the world. Vultures are extraordinarily efficient scavengers and farmers historically relied on them to quickly remove livestock carcasses (D. L. Ogada et al. 2012). As vultures died out, the scavenging services they provided disappeared too, and carrion were left out in the open for long periods of time creating a large negative sanitation shock.

Related Literature Our work links to several strands of the economics and ecology literature. We build on a theoretical foundation in ecology that explores how declines in species that perform important ecosystem functions can have effects beyond their immediate ecosystem (Dirzo et al. 2014; Hooper et al. 2005; Estes et al. 2011; Martin et al. 2013; Ceballos et al. 2015; J. E. M. Watson et al. 2016; Luis et al. 2018; Dainese et al. 2019; Schmeller et al. 2020). We quantify the impact of a catastrophic shock to a keystone species with evidence on mechanisms. Economic theory shows that this type of estimate is essential for a meaningful cost-benefit analysis of conservation policy (Weitzman 1992; Solow et al. 1993; Weitzman 1993; 1998; Nehring and Puppe 2002; Brock and Xepapadeas 2003). Our approach offers an alternative to back-of-the-envelope approaches that have valued global ecosystem and natural capital at nearly twice the output of the global economy (Costanza et al. 1997). Such approaches have been criticized as an "Audacious bid to value the planet" (1998). Furthermore, our use of a natural experiment overcomes some of the limitations inherent to contingent valuation methods (Daily et al. 2000; Heal 2000), as discussed in Hanemann (1994) and Carson (2012).

We also join a nascent strand of the economics literature that has provided empirical evidence on the value of biodiversity. Using variations in environmental suitability, Alsan (2015) studied the long-term effects of the tsetse fly on agricultural production and political institutions. More recent papers study how farmers increase their use of insecticides to substitute for the loss of pest control following declines in insect-eating bats (Frank 2021); how air pollution increases after tree die-offs caused by the emerald ash borer (Jones and McDermott 2018); the importance of tree shade to human health (Jones 2019); and how reintroducing wolves can change the behavior of deer and reduce deer-vehicle-collisions (Raynor et al. 2021). Other related work in economics has focused not on the impacts that keystone species have on human well-being, but on how technology and trade can play a role in their decline (Taylor 2011), how anticipated scarcity can lead to extinction (Kremer and Morcom 2000), or even actively promote extinction (Mason et al. 2012).

Finally, we add to a body of work outside the economics literature on the vulture collapse in the Indian sub-continent. Vibhu Prakash et al. (2012), Cuthbert et al. (2014), and Galligan et al. (2020) document the magnitude and spatial extent of the loss of vultures and investigate whether restrictions on the veterinary use of diclofenac have aided recovery.[6] To the best of our knowledge, the closest paper to our work is Markandya et al. (2008) who use a back-of-the-envelope calculation to estimate the extent to which the population of feral dogs might increase in the absence of vultures and thus the potential mortality costs due to increased rabies. This calculation relates to one of several mechanisms through which the loss of vultures might affect mortality, with other channels including water pollution and increased spread of infectious diseases. In this paper, we collect panel data at the district level to test whether the decline in vultures had a detrimental effect on health outcomes, and leverage baseline variation in vulture suitability to identify the full causal effect of their decline on mortality.

The remainder of this paper is organized as follows. In Section 2 we describe the role of vultures as scavengers and outline the mechanisms through which their disappearance might impose costs on society; followed by the cause of the sudden population collapse of vultures in India. In Section 3 we describe the

[6] The Indian government banned diclofenac for veterinary use in 2006 but the widespread diversion of diclofenac doses meant for humans may have rendered this regulation relatively toothless. In 2015, diclofenac was restricted to single dose injections for humans and a court battle continues on a complete ban. Unfortunately, close derivatives such as the drug aceclofenac remain legal and new evidence shows they have similar harmful impacts on vultures because they quickly metabolize to diclofenac (Chandramohan et al. 2022).

sources of data we use in this paper. In Section 4 we outline the econometric approach we use and present different specifications that we take to the data. In Section 5 we present our estimates of the mortality impacts of losing vultures. We also present supporting evidence on the hypothesized mechanisms and a summary of different robustness checks and alternative specifications. In Section 6 we benchmark the effects of losing vultures against other environmental or sanitation shocks and include an assessment of the costs of replacing their ecosystem services with technology (incinerators). We conclude in Section 7.

2 Vultures as Ecosystem Sanitizers

The ecological and epidemiological dynamics of scavengers, pathogens, and infectious diseases help explain the causal link between diminishing vulture populations and human health. While some animal species will feed on carrion if available, for vultures, it is the only source of food. As a result, vultures have evolved as very efficient scavengers. High stomach acidity—up to a hundred times more acidic than the stomach of humans—reflects one of the key adaptations that allows vultures to safely consume carrion, and also results in most bacteria not surviving their digestive system (D. L. Ogada et al. 2012; Roggenbuck et al. 2014).

Vultures are uniquely effective at reducing a carcass to its bones, and can consume the carrion of an entire cow within 40 minutes (D. L. Ogada et al. 2012).[7] Other scavenging species such as dogs and rats not only leave the flesh behind and therefore do not solve the sanitation problem, but also transmit various diseases including rabies. Recent experimental evidence confirms that vultures do not have a good functional replacement in the ecosystem (Hill et al. 2018).

The historic presence of large and stable vulture populations simultane-

[7] We use previously published numbers on the meat consumption of vultures, and on the mean weight of cattle in India to estimate that vultures removed roughly 27.5 million cow carcasses a year. We walk through this back-of-the-envelope calculation in more detail in Section A.10 of the Online Appendix.

ously reduced pathogen and bacteria concentrations in the environment, and crowded out other scavengers such as dogs and rats that transmit disease. (Moleón et al. 2014). In settings with very limited access to expensive animal incinerators—itself perhaps an equilibrium outcome of the free sanitation provided by vultures—the role of vultures is particularly important. In place of incinerators, "animal landfills" have emerged on the outskirts of population centers across India (Sanjayan 2013). Anecdotal accounts describe how with vultures no longer available, the rotting meat and its scent build up, attracting feral dogs.[8] The combination of dogs and rats serving as vectors of infectious diseases, and being far less efficient scavengers than vultures, make carcass dumps a breeding ground for disease (D. L. Ogada et al. 2012).

Livestock agriculture also becomes a source of water pollution once farmers need to dispose of dead animals themselves (Engel et al. 2004; Kwon et al. 2017). A 2016 Supreme Court ruling in the state of Uttarkhand recognized that animal carcass dumping in water bodies is an ongoing problem, even in water bodies that are considered sacred: "It is tragic that the Ganga, which has since time immemorial, purified the people is being polluted by man in numerous ways, by dumping of garbage, throwing carcass of dead animals and discharge of effluents" (Sharma and Singh 2016).

Finally, the interaction of widespread dairy cultivation with cultural practices regarding dead animals has resulted in a historically large reliance on scavengers in India. Restricting the amount of carrion and the time it remains in open fields is of particular importance in India due to the prevailing social norms regarding the handling of meat. Hindus will not consume cows, whereas Muslims will not consume animals not killed according to *halal*.

We summarize the interactions between vultures, mammalian scavengers, environmental quality, and public health in Figure 1. Within the ecosystem interaction group of vultures, mammalian scavengers (dogs and rats), and livestock carrion, the former two are competing for the food source (dead

[8] As Dr. Asad Rahmani, Director of the Bombay Natural History Society, put it: "Now there are dogs. They eat anything, live or dead. There are dogs on the ground but the skies are empty" (Subramanian 2011).

animals). Greater availability of dead carrion supports larger populations of both scavenger types, efficient (vultures), and inefficient (dogs and rats). Because both types compete for the same food source, each type indirectly limits the population growth of the other type.

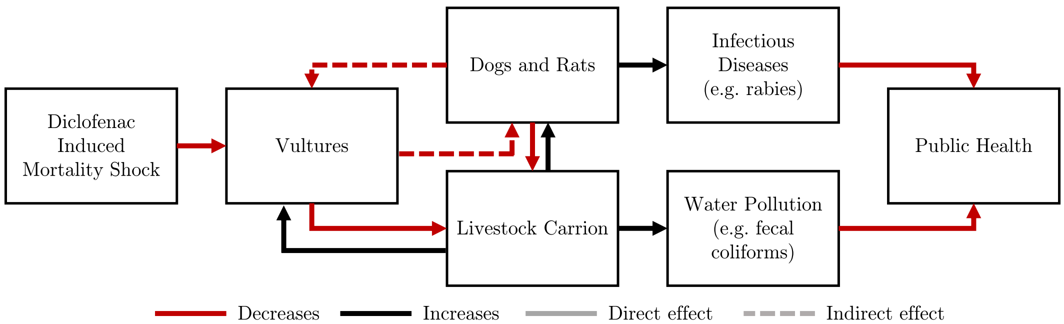
In the absence of vultures, livestock farmers and municipalities can utilize either labor-intensive or capital-intensive substitutions. Farmers can exercise deep burial but given the number of livestock animals this adds high labor costs. Since these costs are private while the costs of disposing of animals in carcass dumps or water are socialized, it is not surprising that deep burial remains uncommon. Livestock carrion can be disposed of using specially designed incinerators, yet they are expensive to buy and operate and require a reliable mechanism for making sure that farmers transport dead animals to them. According to a 2020 report by India's Central Pollution Control Board, India has yet to adopt livestock incinerators as a substitution for vultures: "Very few cities have carcass utilization plants and incinerators. One such carcass utilization plant is installed in Delhi and incinerator is under installation in Chandigarh" (Central Pollution Control Board 2020).

In other words, there are well-defined mechanisms at work that imply that removing vultures from the ecosystem may lead to worse environmental quality, inefficient scavengers, animal-borne diseases, more carrion rotting in the open or thrown into water bodies, and an increase in infectious disease vectors.

2.1 The Sudden Population Collapse of Indian Vultures

Vultures were once a ubiquitous sight across India with a population that may have exceeded 50 million birds. In the course of a few years in the second half of the 1990s, the number of Indian vultures in the wild fell by over 95%. Today, the three species that made up the bulk of the population are all critically endangered with a few thousand birds left in the wild. The decline of vultures in India is the fastest of a bird species in recorded history and the largest in magnitude since the extinction of the passenger pigeon in the United States.

Figure 1: Schematic Relationship of Ecosystem Interactions & Environmental Quality



Notes: The figure summarizes the key components of the coupled natural-human system: (i) ecosystem interactions between vultures, dogs and rats, and livestock carrion; (ii) the impacts that mammalian scavengers and carrion have on environmental quality and public health. Red lines denote a decreasing effect, while black lines denote an increasing effect. Solid lines reflect a direct effect, while dashed lines reflect an indirect (reduced form) effect.

The cause of vultures' death was initially mysterious.[9] It was only in 2004 that research showed that several species of vultures would develop kidney failure and die within weeks of digesting carrion with even small residues of the chemical diclofenac (Oaks et al. 2004).[10]

This discovery was a surprise because diclofenac was (and still is) a common painkiller, harmless to human beings, and widely prescribed for people across the world. Indeed, the drug itself is decades old, even at the time, first introduced in 1973 by Ciba-Geigy (now Novartis). It has since become the most widely used non-steroidal anti-inflammatory drug in the world and is prescribed as a painkiller for many conditions (Altman et al. 2015).

What changed in the early 1990s was that for the first time, the *veterinary* use of diclofenac became feasible and economically viable because of the entry of cheap generic brands made by Indian companies. These generics accompanied the expiry of a patent long held by the pharmaceutical company Novartis (Subramanian 2015). Once farmers began treating their cattle with diclofenac,

[9] At the time, conjectures ranged from the emergence of an unknown new disease, pesticide accumulation, and even deliberate poisoning by western countries (Subramanian 2015)

[10] We use the term kidney failure for clarity. The more medically correct terms are renal failure and visceral gout.

the carcasses of their livestock retained trace amounts of the drug, becoming deadly to vultures.

We draw on multiple sources of data and identify 1994 as the first year in which diclofenac was widely used to treat livestock. Anecdotal accounts place the timing of the patent expiration in the early 1990s (Subramanian 2015). We confirm this using formal patent records and approval for a generic version granted to Novartis in 1993 by the US Federal Drug Administration. Survey evidence also identifies 1994 as the first year when farmers in India began using this drug, previously prescribed only to humans, to treat their livestock (Cuthbert et al. 2014). In addition, we purchased pharmaceutical sales data from the company IQVIA which shows a dramatic growth in the entry of Indian drug manufacturers around this time (see Figure 3a and Online Appendices C and D for more detail).

Reports of vulture declines rapidly followed the veterinary use of diclofenac. Field observations in 1996 found only half of the 353 nesting vulture pairs recorded in 1984 in Keoladeo National Park outside Delhi (Subramanian 2011). After Dr. Vibhu Prakash, at the time a PI in the Bombay Natural History Society, communicated his findings, colleagues reported similar patterns they thought were simply idiosyncratic to their study sites. Population declines were so rapid that in 2000, all three species were classified as critically endangered. The Indian government eventually banned the veterinary use of diclofenac in 2006 (Vibhu Prakash et al. 2012; D. L. Ogada et al. 2012). However, surveys conducted up to 2018 document rampant illicit use of diclofenac in livestock, including by diverting human doses (Galligan et al. 2020). As a result, vulture populations in India have never recovered.

As vultures died out, the scavenging services they provided disappeared too, and carrion were left out in the open for long periods of time. Ecologists have argued that this may have led to an increase in the population of rats and feral dogs, which are a major source of rabies in India. Rotting carcasses can also transmit pathogens and diseases, such as anthrax, to other scavengers. In addition, these pathogens can enter water sources either when people dump carcasses in rivers or because of erosion by surface runoff (Vijaikumar et al. 2002; R. T. Watson et al. 2004; Markandya et al. 2008; D. Ogada et al. 2016). These cascading effects imply that the decline of vultures may have resulted in an extraordinarily large, negative sanitation shock to human populations.

3 Data

In this section, we briefly summarize the data sources that we use in our analysis. We also use the raw data to provide descriptive evidence of the growth of diclofenac, the decline of vultures, and possible effects on mortality. Throughout the analysis, we use districts and states held at their 1981 borders (see Online Appendix C.4 for more on this).

3.1 Vulture Habitat Ranges

Our empirical strategy (described in more detail in Section 4) relies on exploiting geographic variation in the prevalence of vultures before their collapse. Unfortunately, we are unaware of any tabulation of vulture populations in different parts of the country before their collapse, a state of affairs that is common for most non-human species.

Therefore, to determine where vultures used to exist, we obtain maps from BirdLife International (BLI) on the species distribution ranges of all bird species (BirdLife International and Handbook of the Birds of the World 2018). We extract the range maps for vulture species and perform two spatial calculations with the 1981 district boundaries (GADM 2018): (i) whether the district intersects with the range map, and (ii) the area of overlap between the range map and the district (see Figure A.3 for a summary of the distribution of these values). We use the area of overlap to calculate the share of area for each vulture species in each district. Our approach assigns each district a suitability category for diclofenac-affected vultures by dividing the mean overlap of species ranges into terciles. This provides us with a proxy for the abundance of vultures and their prevalence across the district. Figure 2 shows the spatial distribution of the classification into high and low suitability categories for diclofenac-affected vultures. In Appendix Section A.2, we provide an extensive review of the ecology literature as well as a set of original validation exercises used to confirm the quality of this proxy. Briefly, we collect data on over 400 bird species in North America for which *both* population counts and habitat range maps are available. We recalculate our habitat overlap measures for each of these species and find a tight relationship between habitat overlap and population counts. An additional benefit of this approach is that it is less dependent on functional form assumptions previously used in the economics literature to relate environmental suitability to outcomes of interest (Alsan 2015).

Finally, although we use habitat suitability scores for our empirical specifications, it is possible to gain some sense of how vulture populations changed by relying on citizen science reports. The Global Biodiversity Information Facility (GBIF) database (GBIF 2024), aggregates multiple reporting sources of data, including some scientific studies and citizen science reports.[11] We calculate the share of reports of diclofenac-affected vultures relative to other bird species that have non-zero observations each year from 1990 to 2005. Figure 3b, shows a decline in this share, with a trend break that follows the veterinary use of diclofenac in 1994. Unfortunately, these data cannot be used for reliable empirical estimates of the rate of decrease of vultures because once it became known that they were growing rare in the wild, bird enthusiasts would have dedicated more effort to documenting residual birds. In the Appendix, we add a second piece of indicative evidence of the decline of vultures by reproducing a set of survey results that counted vultures along 70 road transects five times between 1992 to 2007 and (Prakash et al. 2007). In Figure A.1, we plot the data from these surveys—they show a decline by about three orders of magnitude over this period.

[11] Previous work has used citizen science data from eBird records to examine the effects of air pollution or the COVID-19 pandemic on bird populations (Liang et al. 2020; Madhok and Gulati 2022).

3.2 Sales & Product Entry of Pharmaceuticals in India

We purchased data from IQVIA on the sales of drugs across India from 1991 to 2003 (IQVIA 2003). The data include information about the main active ingredient, the concentration, usage (topical, oral, or injection), as well as data on the quantity sold, value sold, and the year when the product was launched. Sales of rabies vaccines and of dicoflenac-based painkillers are of particular interest in the context of this paper.

In Figure 3a, we plot both the price and quantity sold of injectable painkillers containing diclofenac. We see that prices dropped dramatically over a short period of time such that by 1996, the mean price begins to stabilize at less than half of its level in 1991. Meanwhile, diclofenac sales increased by almost ten-fold from 1991 to 2003. Although these data largely correspond to medical sales, the sharp fall in price that we observe helps explain the reported entry of diclofenac into the veterinary market in 1994 (Cuthbert et al. 2014). We plot data on injections as that is the version of the drug that is most commonly used to treat animals.[12]

3.3 Health Outcomes

We use mortality data at the district level from the Vital Statistics of India (VSI), reported as part of the Civil Registration System (CRS) (Office of the Registrar General 2005). The data include information regarding live births, deaths from all causes, and infant deaths. Most districts have areas defined as either rural or urban, and the data are reported separately. Areas classified as urban are not necessarily similar to a city, and might simply be denser villages. An area is officially classified as urban if it has a population above 5,000 people, and if more than 75% of men work in non-agricultural jobs (Burgess et al. 2017).

The CRS data yield an unbalanced sample of districts because these records

[12] The IQVIA data does not provide cumulative sales in India because it collects data from a sample of pharmacies. Thus we focus attention here on trends, and changes in those trends.

could not be obtained for some state-years early in the time period we study. To rule out any composition effects over time, our preferred estimates all use a restricted sample of 153 districts for which we have a fully balanced panel from 1988 to 2005. That said, we also estimate additional specifications using the full unbalanced sample and this does not substantively affect our results.

Using the classification into high and low suitability for the diclofenac-affected vultures, we plot changes in the mean population-weighted all-cause death rate for the balanced sample in Figure 3c, relative to 1993. We observe an increase in mortality in the high-vulture-suitability districts following the introduction of veterinary diclofenac. However, no similar change in magnitude or trend is observed in the lowest suitability category. The habitat suitability groups trend similarly quite strongly in the years leading to the collapse in diclofenac-affected-vulture populations, yet diverge from each other following the onset of diclofenac use in livestock—the cause of the vulture collapse. While high-suitability districts exhibit a break from their 1988 to 1993 trend, low-suitability districts maintain the same mean death rate from 1988 to 2005.[13]

An important limitation of CRS data in India is that many vital statistics events go unrecorded, and as a result, the CRS under-reports the true magnitude of mortality. We adjust for this when interpreting our empirical results and discuss this further in Section 4.

3.4 Livestock Census

In addition to a population census and an industrial census, India also reports a livestock census. The data include counts of different livestock animals such as cattle, sheep, etc. We use the data from 1987 and 1992 (Ministry of Agriculture 1987; 1992)to classify districts as high or low livestock districts at baseline (as above or below the median level), which we use as part of a triple-differences design (see Section 4).

Notwithstanding the name, the livestock census also reports a count of

[13] Our main sample starts in 1988 because reporting of CRS data changed in 1988. See Online Appendix C.5.

dogs at the district level. However, these were only systematically collected for feral dogs starting in 2012.[14] If dog populations are higher in the high-suitability areas for diclofenac-affected vultures, then that is consistent with the anecdotal evidence regarding the increase in feral dogs, animal bites, and rabies cases.

3.5 Water Quality

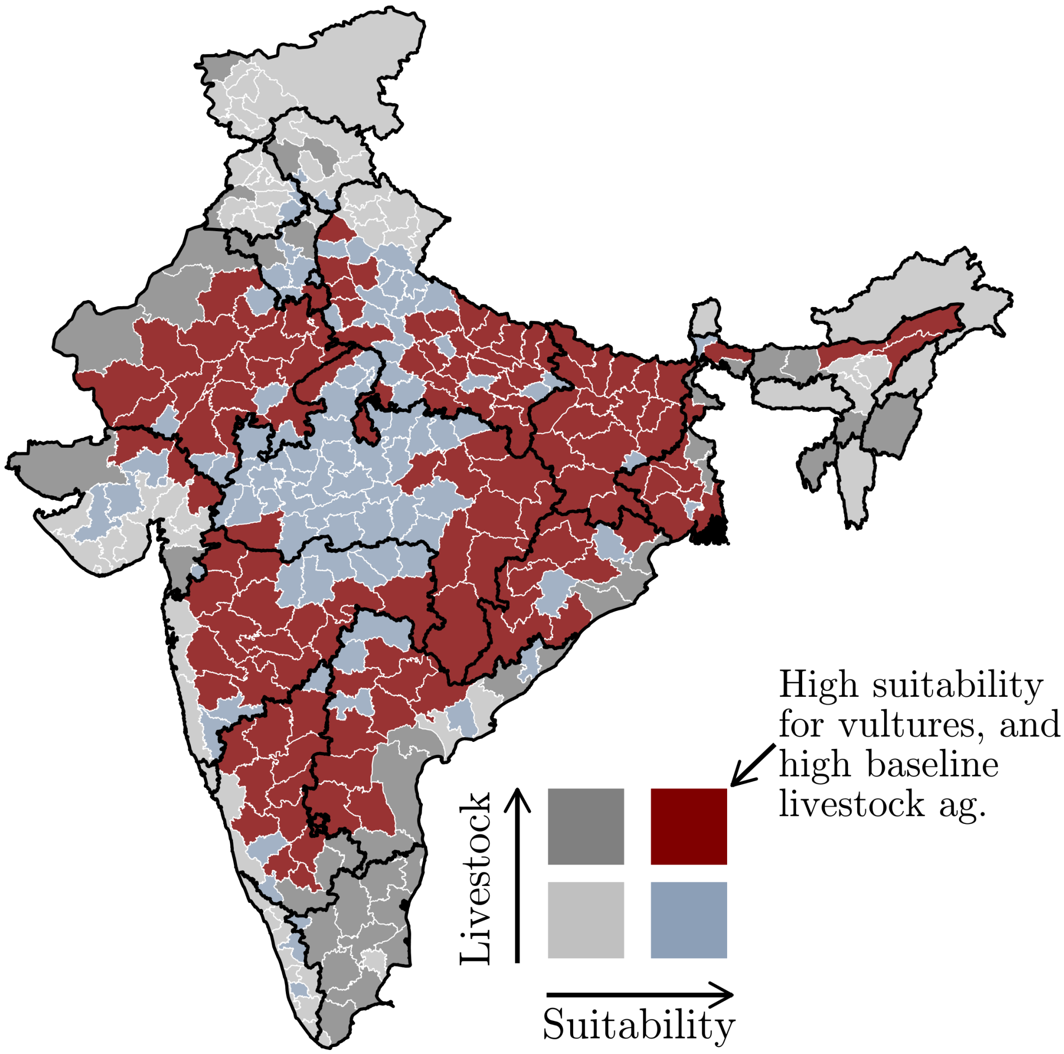
India's Central Pollution Control Board operates a network of water quality monitors covering different surface and groundwater sources. Greenstone and Hanna (2014a, 2014b) draw upon this data and use 489 monitors located at different points along 162 rivers to create an unbalanced district-level panel spanning 1986-2005. We use this dataset for our analysis and more details on its construction are available in the original paper.

3.6 Additional Environmental & Demographic Data

In some of the results, we either include weather controls or demographic data. We obtain weather data from ERA5 reanalysis product (Hersbach, H., and Coauthors 2020). We obtain additional demographic controls from the Socioeconomic High-resolution Rural-Urban Geographic Platform for India (SHRUG) (Asher et al. 2021).

[14] As Markandya et al. (2008) summarize: "Participants in the census were instructed to count dogs owned by households as domestic, and all other dogs, including dogs fed by households but not owned by them as 'other.' Total counts are therefore likely to include the majority of semi-dependent dogs around count households, but may not include a high proportion of truly feral dogs."

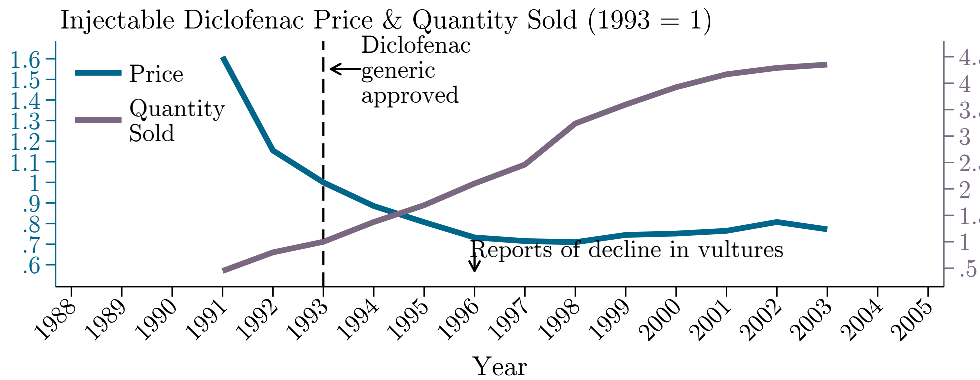
Figure 2: Spatial Distribution of Diclofenac-Affected-Vulture Ranges & Livestock Agriculture



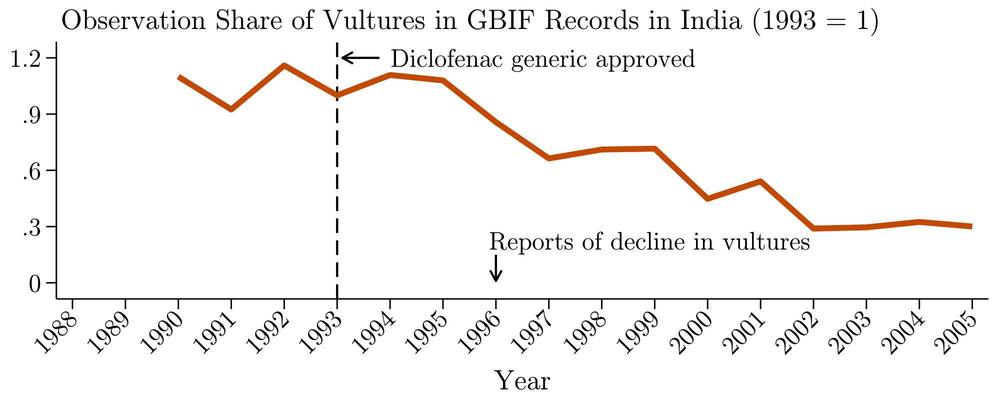
Notes: Districts in India, at their stable 1981 geographic borders, classified as high or low exposure to diclofenac-vulture-collapse, and as high or low baseline livestock agriculture (see Section 3 for more details).

Figure 3: National Trends in Diclofenac Use, Vulture Observations & Death Rates

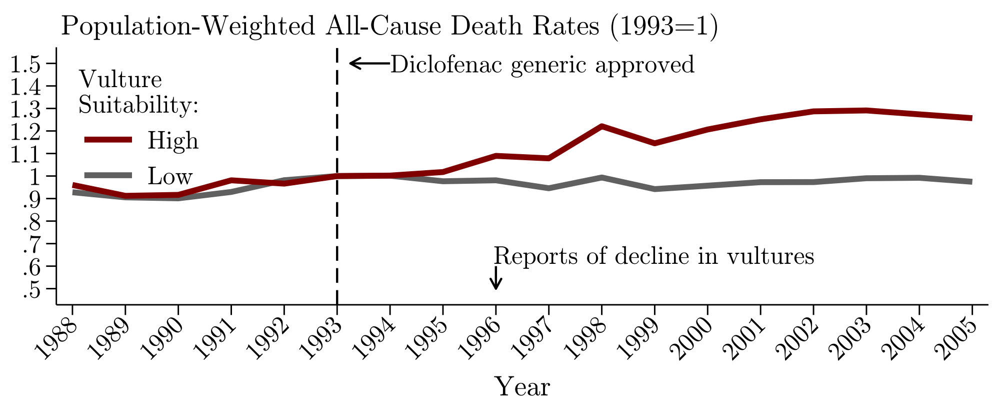
(a) Expansion in Diclofenac Around the 1994 Veterinary Use Onset



(b) Decline in Observations of Affected Vulture Species



(c) All-Cause Death Rates by Vulture Habitat Suitability



Notes: (a) Injectable forms of diclofenac price and sales (Source: MIDASTM, years 1991-2003, IQVIA LTD. All Rights Reserved). (b) The share of vulture reports relative to all bird species that are consistently reported every year. (c) Mean all-cause death rates for balanced districts by vulture suitability classification for diclofenac-affected vultures. Each time series is normalized relative to 1993.

4 The Collapse of Vultures in India as a Natural Experiment

We turn now to our empirical approach. To estimate the causal effect of the collapse in vulture populations on public health, the ideal experiment would randomly assign vultures to different districts across India. This ideal experiment is impossible to conduct. However, the poisoning of vultures from diclofenac residue in livestock carcasses provides a plausibly exogenous and large shock, affecting those areas where vultures were historically prevalent. The timing of this shock was not based on local factors but rather was determined by the expiry of a long-standing international patent, the consequent approval of a generic formulation in 1993, and the introduction of veterinary formulations in 1994. Nor were the effects on vultures anticipated at the time, indeed the connection of the drug to the demise of specific vulture species was only made a decade later in 2004. Finally, diclofenac itself was neither new to humans nor harmful to people or cattle. To this day it remains one of the most widely used treatments for pain and inflammation across the world (Altman et al. 2015).

4.1 Differences-In-Differences Design

We use a difference-in-differences approach to estimate the impact of vultures on health outcomes. We treat the sudden decline in vultures after 1994 as a shock resulting in the removal of a key ecosystem service, thus resulting in lower sanitation and an increased risk of disease, including rabies, following the mechanisms described in Section 2. Using our habitat suitability measures, we then compare districts that had a significant vulture presence with those that did not, before and after the 1994 onset of diclofenac use. The key identifying assumption in this design is that both groups of districts would have seen their health outcomes develop along parallel trends in the absence of the collapse in vulture populations.[15]

[15] This implicitly requires two additional assumptions that we find reasonable. First, that vulture populations were in equilibrium prior to the onset of diclofenac use. Second,

Mortality effects over time: We estimate the following event-study-like regression specification:

$$y_{daszt} = \sum_{\substack{\tau \in \{\underline{T},\ldots,\overline{T}\} \\ \tau \neq 1993}} \beta_\tau (\text{HVS})_d \times \mathbb{1}\{t = \tau\} + \\ \lambda_{da} + \delta_{zt} + \boldsymbol{X_{daszt}\theta} + \varepsilon_{daszt} \quad (1)$$

Our main outcome of interest is the all-cause death rate, y_{daszt}, in district d, rural or urban area a, state s, in zonal council z, and time period t. We denote the treatment variable as HVS, which is a dummy variable that equals one for districts that we classify as having a high pre-collapse presence for the three vulture species affected by the exposure to diclofenac, and zero otherwise. We define high presence as being in the top and middle terciles of our habitat suitability index constructed using the overlap between vulture ranges and district areas and described in more detail in Section 3 (see Figure 2). We interact the treatment variable with year dummies, with 1993 as the baseline (omitted) year since that is when the use of veterinary formulations began (see 2.1 for details)

The coefficients on these interaction terms, β_τ, recover the dynamic response in the outcome variable of interest following the collapse in vulture populations. Each coefficient provides an estimate for the difference between the high and low suitability districts, before and after the collapse. We should expect to see no systematic difference prior to 1993, which would be consistent with the identifying assumption of parallel trends on the counterfactuals. If the decline in vulture populations resulted in deteriorating health conditions, then we should expect to see the coefficients diverge from zero following 1993. The differences between high and low suitability districts could diverge further over time as vulture populations continue to decline, and mammalian scavenger populations increase.

Our comparison of high to low suitability areas will tend to recover a lower

diclofenac was used widely to treat cattle and not only in areas with high suitability for affected vultures.

bound of the effects following the collapse in vulture populations. This is because the districts we classify as low suitability may still be affected to some degree since their baseline vulture populations are unlikely to have been zero.[16] This means that our analysis is leveraging differences in the intensity of the collapse experienced in each district with the control providing an approximation to the ideal counterfactual of zero treatment. In Section 6 we provide more discussion of the likely size of the differential shock in high vs low suitability districts.

Since we are interested in residual variation that is not explained by time-invariant characteristics of districts, or pooled time-trends, we include district-area fixed effects λ_{da} as well as a flexible set of controls for common time trends. District-area fixed effects control for baseline differences in factors such as sanitation, morbidity, mortality, and healthcare access.[17] To further ensure that any observed results are strictly driven by the interaction of vulture suitability and diclofenac use onset, we also include time-varying environmental control variables, $\boldsymbol{X_{daszt}}$. These include flexible degree days in intervals of three-degree Celsius bins, along with precipitation quintiles.

In our primary specification, we control for time trends using zonal council-by-year fixed effects. In 1957, India was divided into six zonal councils, where each zonal council contains two to seven states, as defined by their 1981 borders. We also run specifications using state-linear time trends as well as state-by-year fixed effects. These state-level controls additionally guard against the possibility that states that we classify as high-suitability for diclofenac-affected vultures also happened to change (systematically increase) their reporting of mortality outcomes after 1994.[18]

These fixed effect designs also help adjust for known under-reporting in death rates from the CRS since our estimates are based on relative changes

[16] There are only two districts in the data that do not overlap with any of the ranges of diclofenac-affected vultures.

[17] In specifications where we separately examine effects on urban and rural areas we correspondingly allow for separate fixed effects for urban and rural areas in the district.

[18] The cost of using increasingly granular time controls is that we risk absorbing much of our identifying variation.

and not the absolute levels of mortality in the data. In the Appendix, we use an alternative source of more aggregated vital statistics data from India's Sample Registration System (SRS) to show that although the CRS underestimates mortality rates by about a factor of two relative to the SRS, after controlling for state and zonal council-by-year fixed effects, both sources of data allow us to recover similar trends in mortality rates. When reporting estimates in percentage terms, we use the nationally representative baseline mean of all-cause death rates in deaths per 1,000 people between 1988 to 1992 of 10.2 for the entire country, and 7.2 for the census urban area (see Online Appendix C.5 for additional details).

Any unobserved variation is captured by the error term, ε_{daszt}. We allow standard errors to be correlated across years within districts. In our baseline results, we allow standard errors to be correlated across districts up to a distance threshold of 200 km. In the Appendix, we demonstrate that the choice of bandwidth has little effect on the precision of the estimates.

Average Treatment Effects We estimate aggregated versions of Equation 1 to summarize average treatment effects. We define a post-diclofenac use dummy variable that is equal to one from 1994 onward as well as two 'partial period' dummies that take the value one during the years 1994 to 1999 and 2000 to 2005 respectively. These help capture average effects shortly after the diclofenac shock and several years later. We estimate specifications of the following type:

$$\begin{aligned} y_{daszt} = & \beta(\text{HVS})_d \times \mathbb{1}(t \in [1994, 1999])_t + \beta(\text{HVS})_d \times \mathbb{1}(t \in [2000, 2005])_t \\ & + \lambda_{da} + \delta_{zt} + \boldsymbol{X_{daszt}\theta} + \varepsilon_{dszt} \end{aligned} \tag{2}$$

4.2 Heterogeneity in Effect of Vulture Loss

We investigate two dimensions over which we might expect increased negative effects of loss of vultures.

Table 1.
Differences in Observables Prior to the Collapse of Vultures

	(1)	(2)	(3)	(4)
	Group	Means	Δ:(2)-(1)	N
Vulture Suitability	*Low*	*High*		
All-Cause Death Rate[1,2]	5.3	4.2	-1.2	153
	(1.8)	(1.8)	(.32)	
Degree Days Above 30°C[1]	54	66	12	153
	(43)	(35)	(6.8)	
Precipitation ($mm \cdot km^{-2}$)[1]	.25	.12	-.12	153
	(.42)	(.18)	(.044)	
Baseline Livestock[3]	1.6	1.6	.028	153
	(.87)	(.73)	(.15)	
Log(Dissolved Oxygen)[1]	1.9	1.9	.0045	95
	(.18)	(.27)	(.047)	
Log(Fecal Coliform)[1]	7.2	7.4	.25	76
	(2.2)	(1.7)	(.48)	
Pop. Share [1, 24][4]	.42	.51	.097	145
	(.14)	(.08)	(.023)	
Pop. Share [25. 54][4]	.29	.33	.035	145
	(.098)	(.058)	(.016)	
Pop. share [55, 100][4]	.083	.088	.0056	145
	(.029)	(.018)	(.0048)	
Share Literate[4]	.55	.41	-.14	143
	(.13)	(.12)	(.021)	
Water Taps[4,5]	11	13	1.1	145
	(27)	(21)	(2.7)	
Water Wells[4,5]	23	57	34	145
	(25)	(42)	(6)	
Hospitals & Health Centers[4,5]	1.7	2.4	.73	145
	(1.7)	(2.5)	(.34)	
Doctors & Health Workers[4,5]	8.1	9.8	1.7	145
	(7.6)	(8.6)	(1.5)	

Notes: Districts with balanced death rates, 1988-2005. Observations are population-weighted. Robust standard errors are in parentheses.
1: Averaged between 1988 and 1993.
2: Per 1,000 People.
3: Values, in millions, for 1987 and/or 1992.
4: Value for 1991.
5: Per 100,000 People.

Livestock Intensity The mechanisms through which vultures affect mortality (as laid out in Section 2), imply that a key driver of increased mortality is the *interaction* of the disappearance of vultures with the presence of a large supply of animal carrion in the vicinity of human populations. These two conditions exist in districts where livestock populations are high. Conversely, in districts where livestock agriculture is less common, there may be less need for the sanitation services vultures provide and a more muted impact of their disappearance.

The mediating role of livestock in the link between vultures and mortality can be tested through a triple-differences approach. We construct a measure of baseline livestock for each district using the mean of livestock counts in 1987 and 1992 from the corresponding livestock census. Next, we construct a dummy variable, (High Livestock), which takes the value one when the district has above the median level of livestock at baseline. Finally, we run a specification as below:

$$\begin{aligned} y_{daszt} =& \beta(\text{HVS})_d \times \mathbb{1}(t \geq 1994)_t \times (\text{High Livestock})_d + \\ & \lambda_{da} + \delta_{zt} + \boldsymbol{X_{daszt}\theta} + \varepsilon_{dszt} \end{aligned} \tag{3}$$

Urbanization Just as high livestock regions might be more affected by the loss of vultures, so might urban areas. Carcass dumping grounds in India are frequently on the outskirts of towns. The presence of animal landfills near and in census-urban centers has been documented extensively in academic writing and news articles (Kumar et al. 2019; McGrath 2007; Pati 2016; Sanjayan 2013; Senacha et al. 2008; Singh et al. 2013; Van Dooren 2010) (see Online Appendix D for more details). In addition, cattle are frequently reared informally within cities and in peripheral urban villages. Socio-religious injunctions against killing cows mean they are also let loose in towns, where they feed on urban waste, eventually dying within the city. These features are present even in India's capital city of Delhi, where animal waste has also been found to spread through sewage canals and drains (Kumar et al. 2019;

Sanjayan 2013).

The presence of animal remains within urban areas may be especially dangerous because population densities are much higher than in rural parts of the country allowing both infectious and water-borne diseases and rabies to spread more rapidly. To investigate heterogeneity along this dimension we split our sample and re-estimate Equations (2) and (3) separately for outcomes corresponding to urban and rural regions within districts.

5 Results

Figure 3c provides a plot showing the divergence of all-cause death rates between low- and high-suitability districts following the introduction of veterinary diclofenac. In this section, we present the main findings from the DD and DDD estimation showing that following the collapse of vultures, all-cause human death rates increased by more than 4%. After validating that these results are robust to different specifications, sample compositions, and definitions of treatment, we present suggestive evidence in support of the specific mechanisms that link vulture decline with human health.

5.1 Comparing High and Low Suitability Districts

Although our identifying assumptions do not require low-vulture-suitability districts (HVS = 0) and high-vulture-suitability districts (HVS = 1) to be balanced at baseline, it is nevertheless informative to compare the two. Table 1 compares the outcome variable and a number of additional covariates for these two groups.

The mean all-cause death rate between 1988 and 1993 was higher by 1.2 deaths per 1,000 people in the low-vulture-suitability districts (HVS = 0) relative to the high-vulture-suitability districts (HVS = 1). At the same time, there is no difference in the mean number of livestock animals as recorded in the livestock censuses of 1987 and 1992. This is consistent with the possibility that in the early 1990s, districts with low suitability for vultures had similar levels of livestock farming, but had lower environmental capacity to manage the resulting animal carrion waste, potentially resulting in higher mortality.

On other covariates, we should expect districts with high vs low suitability to have different environmental conditions. Indeed we find that districts with high suitability have more warm days, and less precipitation. We do not detect any meaningful differences in baseline water quality or water access. We also do not find that high-vulture-suitability districts had a lower provision of healthcare as measured by the number of hospitals and health centers, as well as doctors and health workers. This comparison helps to rule out the possibility of pre-existing differences in water or healthcare infrastructure being responsible for a future divergence of all-cause death rates in the high-vulture-suitability districts relative to the low-vulture-suitability districts.

5.2 Results for All-Cause Death Rate

In Figure 4, we report the event-study estimation results using Equation 1. High and low suitability districts did not have systematically different trends with respect to death rates between 1988 and 1992, relative to 1993. The parallel trends assumption appears justified.

Following the onset of diclofenac use after 1993 and the first observed signs of large-scale decline of vultures in 1996, we find that death rates from all causes increased in the high-vulture-suitability districts. In 1996, the first year in which the decline in vulture populations gained widespread recognition, the all-cause death rate was higher in the high-suitability districts by 0.65 deaths per 1,000 people. By the end of the sample, in 2005, death rates were higher by about 1.4 deaths per 1,000 people. These reflect an increase of 6.4% and 13.7% relative to the nationally representative mean level of 10.2 deaths per 1,000 in the pre-treatment period, respectively.

Farmers gradually increased diclofenac use after the expiry of the patent. This should have caused the vulture population to decrease over the next few years. This is consistent with both GBIF and transect data (see Figures 3b and A.1). Once vulture populations reach a low equilibrium (functionally extinct in the wild) any further changes in diclofenac use will have no effect on the sanitation services provided by the vultures in the ecosystem. These dynamics would suggest that death rates in high-vulture-suitability regions should first diverge from the low-suitability control over a few years and then flatten out. This is precisely what we see in Figure 4 where an equilibrium treatment effect is reached around 2000, by which time vulture populations were a shadow of their previous levels and designated as critically endangered by the IUCN Red List. Importantly, these patterns would hold only if no compensating adaptive investments were made to replace vultures. This appears to be true—the alternative means of disposal is the use of incinerators and government reports as late as 2020 document their near-total absence (Central Pollution Control Board 2020).

We turn next to our aggregate specifications in Equation 2. Table 2 contains these results both with and without temperature and rainfall controls. The model in Panel A, column 1 aggregates over the year-by-year coefficients in the event-study by using a single post-dummy for years after 1993. On average, death rates are higher by 0.91 deaths per 1,000 people. column 2 breaks this down into averages for the 1994 to 1999 period and the equilibrium period (2000 to 2005) as in Equation 2. We estimate precise increases in the all-cause death rate by 0.52 and 1.26 deaths per 1,000 people in the two periods (Panel A, column 2). These models control for zonal-council-by-year fixed effects capturing regional factors that might change death rates including regional and national macro-economic factors.

One concern we may have is the possibility of differential reporting of death rates in high vs low suitability districts beginning after 1994 that may not be fully captured by zonal trends. To control for this, in Panel A Column 3, we use a specification that includes linear time trends for each state, which is the level at which the civil registry reporting system is administered. These controls soak up some of our variation, in particular in the period where treatment effects are also growing over time. However, our finding for equilibrium outcomes remains qualitatively similar, with a fairly precise point effect of 0.48 additional deaths per 1,000 people. This reflects a 4.7% increase relative to the nationally representative mean level between 1988 and 1992 of 10.2 deaths per 1,000 people, as reported in the SRS data.[19] We regard this as our preferred specification for estimating equilibrium elevated death rates due to the disappearance of vultures. Finally, we report results using state-by-year fixed effects in column 4. This absorbs more variation but our results remain broadly similar.[20]

As we discuss in Section 4.2 urban areas might have faced a larger sanitation shock due to their proximity to carcass dumps, significantly higher population density, and network infrastructure such as drains allowing pathogens and waste to spread rapidly. Using the urban-rural breakdown of reported district death rates, we re-estimate all models for urban areas only and report results in columns 5-8 of Table 2. Across all specifications, we find that urban areas experienced a larger increase in death rates relative to the combined sample.[21] For our preferred specification including state-linear trends (columns 3 and 7), urban death rates increase by 0.68 per 1000 people after reaching equilibrium (2000-2005). This compares with an estimate of 0.48 in the combined sample.

5.3 Long-Difference Models

In the main analysis described above, we balance our panel to require that each district in the panel reports death rates every year from 1988 to 2005. This limits the number of districts in our sample. After we hold districts in their 1981 geographic borders, there are 340 districts in our sample. Of these, 153 districts have fully balanced data in the combined urban and rural sample.[22]

[19] Using the CRS data allows us to recover level differences, but a correct interpretation of the relative change requires using the nationally representative baseline from the SRS data. See sections C.2 and C.5 for additional details.

[20] Because we hold districts fixed at their 1981 borders, the use of state-year dummies results in aggregating some districts to their state level. As a result, three states are fully absorbed by the state-by-year fixed effects.

[21] An area is officially classified as urban by the Census if it has a population above 5,000 people, or if more than 75% of men work in non-agricultural jobs. Thus census urban regions include areas that may look closer to a dense and large village than a large city.

[22] Many districts are missing at least a year of data, and in the case of the state of Uttar Pradesh, we are missing data for all districts from 1996 to 1999. We went through con-

Table 2.
All-Cause Death Rate, per 1,000 People

Panel A. Without Weather Controls

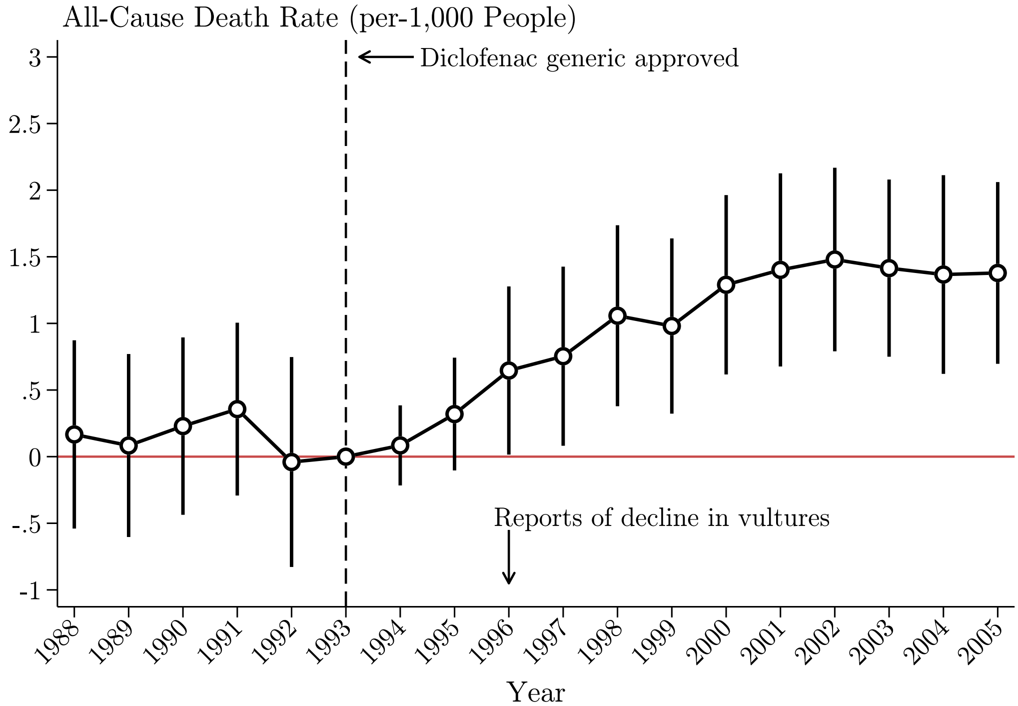
	Combined Sample ($\overline{Y} = 10.2$)				Census Urban Sample ($\overline{Y} = 7.2$)			
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
HVS×Post-1994	0.91				1.04			
	(0.14)				(0.27)			
HVS×[1994, 1999]		0.52	0.13	0.21		0.68	0.35	0.34
		(0.15)	(0.15)	(0.14)		(0.30)	(0.26)	(0.22)
HVS×[2000, 2005]		1.26	0.48	0.40		1.34	0.68	0.63
		(0.19)	(0.16)	(0.16)		(0.30)	(0.23)	(0.24)
R^2	0.74	0.75	0.77	0.80	0.67	0.68	0.70	0.76
N	2,754	2,754	2,754	2,700	2,808	2,808	2,808	2,754
Clusters	153	153	153	150	156	156	156	153

Panel B. With Weather Controls

	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
HVS×Post-1994	0.85				1.04			
	(0.15)				(0.25)			
HVS×[1994, 1999]		0.51	0.18	0.19		0.72	0.40	0.32
		(0.15)	(0.15)	(0.15)		(0.29)	(0.26)	(0.22)
HVS×[2000, 2005]		1.17	0.45	0.38		1.32	0.67	0.64
		(0.19)	(0.17)	(0.17)		(0.25)	(0.22)	(0.25)
R^2	0.75	0.75	0.78	0.81	0.68	0.69	0.71	0.76
N	2,754	2,754	2,754	2,700	2,808	2,808	2,808	2,754
Clusters	153	153	153	150	156	156	156	153
Zonal Council-by-Year FE	X	X	X		X	X	X	
State-Linear Trends			X				X	
State-by-Year FE				X				X

Notes: Estimation results for the specification in Equation (2). Comparing high-vulture-suitability (HVS) to low-vulture-suitability districts, after the collapse of the affected vulture populations. When we include state-by-year fixed effects (columns 4 and 8), three states get dropped as they have no district-level data. Reported means of 10.2 and 7.2 deaths per 1,000 people are for the pre-treatment period of 1988 to 1992. Sample includes balanced district-level data from 1988 to 2005. All regressions include district fixed effects. Observations are population-weighted. We report Conley standard errors that are serially correlated at the district level, and are allowed to be spatially correlated up to 200km.

Figure 4: All-Cause Death Rates DD Estimation Results



Notes: Estimation results from Equation (1) showing coefficients and 95% CIs. The regression compares the high- to low-suitability vulture districts around the timing of the vulture population collapse. Sample includes all districts (combining census urban and rural areas) with balanced data from 1988 to 2005. The regression includes district and zonal council-by-year fixed effects. Observations are population-weighted. We calculate Conley standard errors that are serially correlated at the district level, and are allowed to be spatially correlated up to 200km.

We are able to use a larger sample of districts by estimating a long differences model (Burke and Emerick 2016). Using long differences allows us to overcome issues with missing data in the middle of the panel, and allows us to take averages during pre- and post-treatment periods to address uneven reporting in those periods. The important modification is that we limit the sample to a pre-treatment period of 1990 to 1995, and a single post-treatment period of 2000 to 2005. With a relaxed requirement that districts only have

siderable efforts to fill in any missing years of data. See the Online Appendix for full documentation.

non-missing data in these two periods, we are able to include as many as 324 districts (relative to 153) in combined urban and rural specifications, and as many as 279 districts (relative to 156) when separating urban areas.

The results remain similar to those from the fully balanced panel. In Table 3, we report the results from estimating the long differences model, similar to the specification in Equation (2). Across the larger sample that uses data from almost all the districts in the sample, we find precisely estimated increases in death rates of 0.68 deaths per 1,000 people for the baseline specification, which includes zonal council-by-year fixed effects (Table 3, column 3, Panel A).

Estimating state-level trends poses more of a challenge once we relax the requirement for the panel to be balanced as some districts enter and exit the sample. For our preferred specification with state-linear trends, as well as when including state-by-year fixed effects, we recover smaller and imprecise estimates when using data from both urban and rural areas (Table 3, columns 3 and 4, Panel A). However, as before, when separately estimating effects in census urban areas, the magnitude of the estimated effect remains meaningful and precise when including either state-linear trends or state-by-year fixed effect (Table 3, Panel B, columns 4-7). Lastly, we also use the long-differences model to validate that the result is not sensitive to the inclusion of time-varying district-level controls (see Table A6).

5.4 Investigating the Role of Livestock

We turn next to the role of livestock in increasing the value of the sanitation services provided by vultures. In Table 4, we report results from the triple-differences specification in Equation 3. We find that following the collapse in vulture populations, high-vulture-suitability districts that also had a high level of livestock at baseline showed a significantly higher increase in death rates, relative to districts with below-median livestock populations.[23] This gap widens further when restricting the sample to urban areas (Table 4, columns 3

[23] We still expect some increase in mortality in high-vulture-suitability districts after the collapse, even in the low livestock at baseline districts because those districts had below-median, and not zero levels, of livestock.

Table 3.
All-Cause Death Rate Long-Differences Estimation Results

Panel A. Combined Sample ($\overline{Y} = 10.2$)	(1)	(2)	(3)	(4)	(5)	(6)	(7)
HVS×Post-2000	1.23	0.72	0.68	0.26	0.16	0.17	0.16
	(0.22)	(0.19)	(0.23)	(0.15)	(0.14)	(0.14)	(0.14)
R^2	0.72	0.73	0.85	0.77	0.90	0.79	0.90
N	1,836	3,696	648	3,696	648	3,589	628
Clusters	153	324	324	324	324	314	314

Panel B. Census Urban Sample ($\overline{Y} = 7.2$)	(1)	(2)	(3)	(4)	(5)	(6)	(7)
HVS×Post-2000	1.23	1.04	1.01	0.65	0.62	0.61	0.62
	(0.27)	(0.25)	(0.37)	(0.17)	(0.19)	(0.17)	(0.19)
R^2	0.64	0.65	0.84	0.69	0.90	0.75	0.90
N	1,872	3,193	558	3,193	558	3,087	538
Clusters	156	279	279	279	279	269	269
Balanced	X						
Zonal Council-by-Year FE	X	X	X	X	X		
State-Linear Trends				X	X		
State-by-Year FE						X	X
Collapsed Sample			X		X		X

Notes: Estimation results for the specifications in Equation (2). The regressions compare the high to the low vulture suitability districts in the post-vulture collapse period (2000 to 2005) to the pre-vulture collapse period (1990 to 1995). Column 1 reports the results from the balanced sample from 1988 to 2005. Columns 2 to 7 use districts with unbalanced data, as long as the district has non-missing data in both the pre- and post-periods. Columns 1, 2, 4, and 6 maintain the district-year panel structure, and columns 3, 5, and 7 collapse the data to pre- and post-periods using population weights to obtain a weighted mean of the all-cause death rate in each period. Reported means of 10.2 and 7.2 deaths per 1,000 people are for the pre-treatment period of 1988 to 1992. All regressions include district fixed effects. Observations are population-weighted. We report Conley standard errors that are serially correlated at the district level, and are allowed to be spatially correlated up to 200km.

and 4). These results are consistent with the hypothesis that the main driver of mortality after the collapse in vulture populations is the presence of a large supply of animal carrion that is not effectively scavenged, rather than simply the decline in vultures themselves.[24] We present results from decomposing the triple-differences into two DD comparisons in Table A3, showing that the interaction of high-livestock with post-collapse has a meaningful effect on the all-cause death rate only in the high-vulture-suitability sub-sample.

5.5 Sanitation Channels: Dogs, Rabies, Water Quality

Over our period of interest, India has limited information on the number of feral dogs, the prevalence of rabies, or water quality outcomes. We made an effort to collect available data on all three of these outcomes to explore whether they provide supporting evidence for the key mechanisms that might link a decline in vulture populations to adverse health outcomes (Section 2).

Feral Dogs and Rabies When vultures decline, the reduced competition for carrion allows the population of mammalian scavengers, such as rats and dogs, to increase, which can further spread infectious diseases. Dogs in particular are a major cause of animal bites and rabies infections (Radhakrishnan et al. 2020).

Starting in 2012, India began collecting data on feral dogs as part of its livestock census. In Figure 5, we plot the correlation between the binned values of feral dogs, in log points, and the mean habitat overlap with diclofenac-affected vultures. We observe a strong association between the degree of habitat suitability and feral dog counts. These suggestive findings are consistent with the anecdotal reporting of increasing dog counts following the decline in vultures. However, as the data are only from 2012, they do not allow us to reject that feral dog populations were already higher in the high-vulture-suitability districts even before the collapse of vulture populations.

[24] This analysis also offers another way to flexibly control for local time trends by subtracting average time trends in the low baseline livestock agriculture group.

Table 4.
DDD Results for All-Cause Death Rate

	Combined Sample ($\overline{Y} = 10.2$)			Census Urban Sample ($\overline{Y} = 7.2$)		
	(1)	(2)	(3)	(4)	(5)	(6)
HVS×Livestock×Post-1994	0.60	0.56	0.18	1.17	1.19	0.17
	(0.26)	(0.32)	(0.28)	(0.45)	(0.44)	(0.58)
HVS×Post-1994	0.49	0.46	0.17	0.29	0.32	0.42
	(0.21)	(0.29)	(0.21)	(0.37)	(0.36)	(0.47)
Livestock×Post-1994	0.05	0.06	0.09	-0.15	-0.15	0.58
	(0.20)	(0.31)	(0.22)	(0.43)	(0.40)	(0.48)
Zonal Council-by-Year FEs	X	X		X	X	
State-by-Year FEs			X			X
Weather Controls		X	X		X	X
R^2	0.74	0.75	0.81	0.66	0.67	0.75
N	2,754	2,754	2,700	2,790	2,790	2,736
Clusters	153	153	150	155	155	152

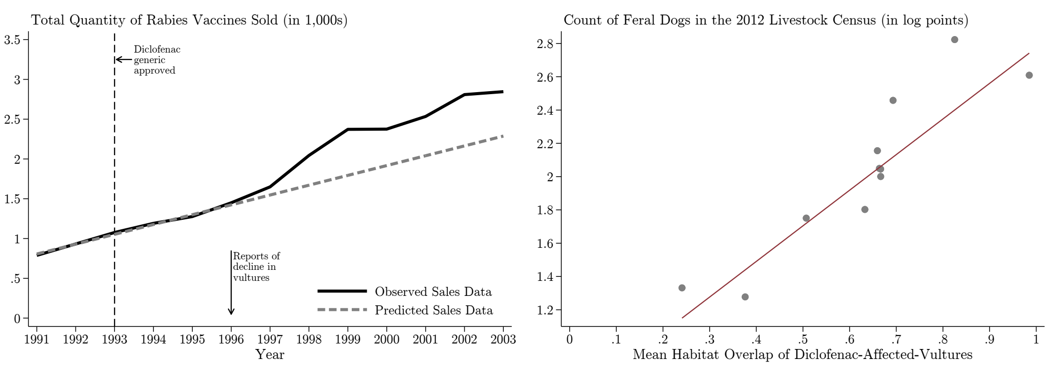
Notes: Estimation results for the specification in Equation (3). The DDD estimation compares the districts that are high-vulture-suitability (HVS), and utilizes the additional sub-group of high-livestock at baseline. Using all livestock animals, we define the high-livestock dummy as being above the median at baseline, using the mean of the 1987 and the 1992 livestock censuses. Sample includes balanced district data, combining urban and rural areas (columns 1 to 3), or only urban areas in the districts (columns 4 to 6), from 1988 to 2005. All regressions include district fixed effects. Reported means of 10.2 and 7.2 deaths per 1,000 people are for the pre-treatment period of 1988 to 1992. Observations are population-weighted. We report Conley standard errors that are serially correlated at the district level, and are allowed to be spatially correlated up to 200km.

We also purchased national-level data on the sales of rabies vaccines from IQVIA. These vaccines are administered as a live-saving treatment after an animal bite, although there are sadly many people in India who still die from rabies because they delay reporting to hospitals.[25] In Figure 5a, we observe a sharp increase after 1996 in the quantity of rabies vaccines sold.

Figure 5: Suggestive Evidence for Feral Dog Mechanism

(a) Sales of Rabies Vaccines

(b) Feral Dogs (2012) VS. Vulture Suitability



Notes: (a) National-level data on all rabies vaccines sold from 1991 to 2003. The solid black line shows the total sold quantity, and the dashed gray line shows a linear trend using the data from 1991 to 1995. (b) District-level data on feral dogs was counted for the first time during the 2012 livestock census.

Water Quality Disposal of dead livestock is a known water pollution source (Engel et al. 2004; Kwon et al. 2017), and water quality deteriorates in the absence of scavengers (Swift et al. 1979; Santori et al. 2020; Brundage 2021). This concern has been noted in the specific setting of the vulture collapse in India: "As there were hardly any vultures left, the carcasses were not disposed of. When the animals died in rivers or other bodies of water, water quality was affected and water sources compromised" (Hugo 2021).

We use data on the water quality outcomes that are most directly linked to a larger presence of carrion when disposal by scavengers declines: namely

[25] Chatterjee (2009) estimated that 36% of global deaths from rabies still occur in India.

dissolved oxygen and fecal coliform.[26] Interpreting the magnitudes we obtain from the water pollution data should be done with caution because monitoring station readings are often unbalanced, and include different water bodies such as lakes, rivers, and wells.

We find evidence of lower dissolved oxygen and higher fecal coliform—consistent with the predictions in the ecological literature and public health literature following the decline in vultures. In Table 5, we report results from a triple-difference specification using water quality as an outcome variable and separately examining urban vs rural outcomes. We find that water quality deteriorates in the urban subsample (columns 2, 3, and 4). Dissolved oxygen drops by 12% in the DDD comparison (Panel A, column 2), while dropping by 7% in the urban subsample (Panel A, column 4). To verify that geographic composition is not driving the results, we use a balanced sample of monitoring in rivers, and recover a 10% reduction in dissolved oxygen.[27] Fecal coliforms more than double in water samples using either the DDD or DD comparison (Panel B, columns 2 and 4). Even though we observe year-on-year and after versus before 1994 variation in the sample that is similar to the magnitude of the change in fecal coliform we report here (see Online Appendix A.14.1 for more details), our emphasis is on the sign of the effect, and that we can reject changes that are smaller than 64%.

In Online Appendix Table A8, we also report increases in biological and chemical oxygen demand, albeit imprecisely estimated. We also find that turbidity declines, which is consistent with previous findings on scavengers increasing turbidity in aquatic environments because they dissect the carrion into finer pieces (Santori et al. 2020).

[26] The higher availability of organic matter decomposing in the water consumes oxygen, lowering the amount of dissolved oxygen. The higher availability of carrion that were not fully consumed by scavengers increases the availability of gut pathogens, such as fecal coliform.

[27] Water quality measurements from river monitoring stations reflect 76.7% of the water quality sample.

Table 5.
District Water Quality DD & DDD Estimates

Panel A. Log(Dissolved Oxygen)	U&R			U
	(1)	(2)	(3)	(4)
HVS×Urban×Post-1994		-0.122	-0.102	
		(0.035)	(0.028)	
HVS×Post-1994	0.004	0.046	0.043	-0.076
	(0.019)	(0.027)	(0.015)	(0.025)
Urban×Post-1994		0.066	0.093	
		(0.030)	(0.027)	
$\overline{Y}_{1988-1993}$	1.92	1.92	1.96	1.89
R^2	0.71	0.71	0.62	0.74
N	4,349	4,349	1,649	2,073
Clusters	220	220	80	139
Panel B. Log(Fecal Coliforms)				
	(1)	(2)	(3)	(4)
HVS×Urban×Post-1994		1.199	2.195	
		(0.360)	(1.005)	
HVS×Post-1994	0.294	-0.111	-0.903	1.132
	(0.287)	(0.340)	(0.492)	(0.341)
Urban×Post-1994		-0.474	-0.564	
		(0.291)	(0.464)	
$\overline{Y}_{1988-1993}$	6.86	6.86	6.93	6.85
R^2	0.78	0.78	0.65	0.83
N	3,344	3,344	986	1,578
Clusters	200	200	48	120
Balanced (Rivers Only)			X	

Notes: Estimation results for DD and DDD specifications. Each regression includes district-by-area-by-type fixed effects where area is either urban or rural, and type is the water body type (well, river, or lake). In addition, each regression includes year fixed effects. Sample consists of district-level data for census-urban (U) and census-rural (R) areas, from 1988 to 2004. Observations are population-weighted. We report Conley standard errors that are serially correlated at the district level, and are allowed to be spatially correlated up to 200km.

5.6 Sensitivity Analysis & Robustness Checks

We evaluate the robustness of the main results in several ways and report outcomes in the Online Appendix. First, we further examine the presence of pre-trends in the data by extending the sample to cover 1981 to 2005, and verify that we recover similar estimates (Figure A6). To better account for other factors that could be changing over time at the state level we confirm that including state-linear time trends or state-by-year fixed effects produces qualitatively similar findings to those in the event-study results (Figure A7). We also use census data to test for differences in per capita hospitals and health centers, as well as doctors and health workers, between the two groups of districts before and after the collapse (Table A9). We are unable to reject the hypothesis that there are no differences. We also run a battery of placebo tests using a variety of different outcomes and fail to detect meaningful differences. (Figure A12).

We also explore whether an alternative method of identifying treatment status affects our results by using a habitat suitability model. Habitat suitability models use data on the presence of the species of interest along with environmental conditions to generate predictions regarding the suitability of a habitat for the specific species. In short, the model first links geographic data on the presence of species to environmental conditions, and then uses the inferred relationship to classify the suitability of other geographic areas.[28] We use the BIOCLIM model, which is a well-established model in the ecological literature (Booth et al. 2014), to generate suitability scores for the diclofenac-affected-vultures, and calculate the mean suitability score across the three species (see Online Appendix, Section A.12 for a full description of the methods and results).

Using the suitability scores from the BIOCLIM model, we generate two classifications of high and low suitability. One that splits the suitability score into terciles, defining the third and second tercile as high suitability, and an-

[28] The habitat range maps produced by BLI, which we use to classify districts into high or low-vulture-suitability, also rely on a habitat suitability model but combine it with expert knowledge and other unpublished records.

other where we define high suitability as being above the median suitability score. We plot the change to the classification of districts along with the event-study analysis in Figure A9, and report the average treatment effects in Table A7. For both of the alternative classifications, we estimate an increase of more than 0.5 deaths per 1,000 people. This analysis confirms that our results are not driven by a specific functional form for the vulture suitability, and that the results are not sensitive to the exact definitions of the treatment and control groups.

We further examine the sensitivity of the results to compositional changes in the sample by estimating two leave-one-out versions of the DD specification in Equation (2). Specifically, we either omit one district at a time, or one state at a time. We plot the resulting narrow distribution of the estimated treatment effects in Figures A13 and A14. Lastly, we perform a permutation inference analysis, where we randomly assign treatment status and re-estimate the DD specification in Equation (2) (Fisher 1966; Barrios et al. 2012; Young 2019). We obtain distributions that are centered around zero, where the estimated effect from the non-randomly assigned treatment is in the right tail of the distribution.

6 Benchmarking Mortality Effects

An effect size of 0.48 deaths per 1,000 people (Table 2, Panel B, column 3) implies an average of 104,386 additional deaths a year relative to a population of 430 million people in the main sample. Using an India-specific mortality risk reduction value (or value of statistical life) of $665,000 implies mortality damages of $69.4 billion per year. These effect sizes are substantial but so is the sanitation shock in question.

In Online Appendix Section A.10 we carry out an indicative exercise to quantify the size of the sanitation shock. We apportion 40 million vultures across districts in proportion to their habitat overlap score. Using data from the conservation literature on the food requirements of adult vultures we conclude that this population could have removed about 10.4 billion kg of meat per year in places where vultures were located.[29] We calculate a measure of exposure to unscavenged meat by adjusting for area and population and find that treatment districts would have had exposures three times higher than controls (Table A4, columns 1 and 3).

The literature supports large improvements in mortality for other interventions that improve water and sanitation, just as we might expect vultures to do. Geruso and Spears (2018) estimate a reduction in infant mortality rate in India by 8% for a 10% decrease in open defecation. In the context of privatizing water provision to improve sanitation and quality, Galiani et al. (2005) find that child mortality drops by 8%, on average, and as much as 26% in the poorest regions. Cutler and Miller (2005) estimate an even larger drop, of 43%, in infant mortality rates from the improvements to water quality in US cities around 1900. In Mexico, where water chlorination went up from 58% to 90%, Bhalotra et al. (2021) find that child mortality dropped by 45% to 67%. These comparisons are tabulated in the Online Appendix (Table D1).

Other environmental risk factors such as pollution have also been found to have large effects on mortality. Ebenstein et al. (2017) suggest that China's policy of providing free heating coal increased all-cause mortality by 20%-26%. Tanaka (2015) finds that air pollution regulations instituted in Chinese provinces in 1998 reduced infant mortality by 20%. T. Carleton et al. (2022) study the mortality effects of exposure to future high temperatures due to climate change. One of the countries projected to be most negatively affected by heat deaths is India. The estimates in this study suggest an increase in death rates by 0.6 per 1,000 in 2099 under an RCP 8.5 warming scenario (a relatively pessimistic 'business as usual' projection of future emissions and warming). This is comparable in magnitude to our estimate of a 0.48 increase in deaths from losing the sanitation services provided by vultures. Of course, deaths due to heat exposure are only one aspect of climate costs and mortality due to climate change, but the comparison is nevertheless striking and

[29] The average weight of the Indian Gir cow is about 385kg (Felius 1995), so this is about 27 million cow-equivalent carcasses per year. Of course, vultures would obtain their food from multiple sources -cows, other livestock, and non-livestock animals such as dogs.

underscores the importance of keystone species to human welfare.

Incinerator Costs A third way to think about these damages is to consider what it would cost to avoid them. The most straightforward alternative to vultures is to build out a network of incinerators (carcass rendering machines) to dispose of livestock carcasses. Ishwar et al. (2016) carry out a detailed analysis of the costs of operating mechanical incinerators using data from 2014-15. They study a medium-sized incinerator model chosen for use by the government and estimate that it is able to process 5,480 cattle carcasses per year at an annual cost (inclusive of operating costs and amortized capital costs) of INR 8,346,097 ($\sim$ USD 139,000).

In 2019, India's livestock population was over 500 million, with about 300 million of those being cattle (20th Livestock Census). Although it is illegal to slaughter cows in India, they do not survive long after their productive life as milch animals because farmers may set them free, effectively denying them access to sufficient food or medicines. Assuming an average life span of about 10 years suggests an annual burden of about 30 million cow carcasses alone. This number suggests annualized costs of operating a nationwide network of carcass rendering machines of about USD 768 million (2014-15 dollars), solely for cows. This estimate ignores air pollution damages from the incinerators.

These are back-of-the-envelope calculations but it is clear that although using technology to replace vultures would easily clear a cost-benefit test, it is still extraordinarily expensive in its own right. Furthermore, rendering machines require farmers to bring dead animals to them, a big disadvantage over vultures, who will go to where the carcass is located. Indeed Ishwar et al. (2016) note that a state-of-the-art machine located in Delhi was non-functional for years, due to lack of any demand.

Vulture Recovery Finally, we might wonder what it would cost to bring back vultures. We do not venture to place a monetary cost on this option for two reasons. First, a key element of any such recovery would be a successful ban on diclofenac and its derivatives. The leading alternative to this drug is

Meloxicam which is similarly priced but takes much longer than diclofenac to be effective in cattle (roughly four hours against 15 minutes). Second, the most significant hurdle involved in restoring vultures to the point where they might once again provide these services is the time it would take. Vultures, much like humans, reproduce relatively slowly. They mate for life, reach sexual maturity at five years, and lay only one or two eggs each year.

7 Conclusions

We live in an era of mass extinctions, only the sixth in the history of the planet and the first to be induced by human activity. Policies intended to preserve biodiversity exist in countries all over the world, from the US Endangered Species Act to India's Wildlife Protection Act.

Yet the paucity of evidence on the costs of losing specific species has made it difficult to both target conservation or recovery efforts, and to determine appropriate levels of funding. Focusing on keystone species is one way to narrow down what would otherwise be a large set of claimants for policy dollars.

In this paper, we provide evidence on the public health implications of the decline of vultures in India. Using a difference-in-differences strategy, we compare districts with habitats highly suitable for vultures to those that are unsuitable, both before and after the onset of diclofenac use. We find that districts that were affected by the disappearance of vultures—those with highly suitable habitats—saw an increase in human all-cause death rates of at least 4.7%, averaged over 2000 to 2005.[30]

Narrowly, these results may inform current vulture recovery efforts in India, and conservation efforts elsewhere. Vultures are important scavengers in parts of Africa as well as Europe, but their populations are falling and diclofenac is

[30] Beyond mortality, losing vultures may also have other costs we do not measure. On the health side, this includes increased morbidity. Vultures also provide other important services. India's tanning industry once relied on quick removal of carrion by vultures. The Parsi community in India has burial rituals that require vultures to consume the body.

still commonly used in many parts of the world.

More broadly, this paper shows how local extinction events can be used to learn about anthropocentric benefits from biodiversity, potentially allowing us to make better policies before a species goes extinct everywhere in the wild. In addition, the vulture collapse in India provides a particularly stark example of the type of hard-to-reverse and unpredictable costs that must be accounted for when evaluating the introduction of new chemicals into fragile and diverse ecosystems. Although it is easy to be wise after the fact, it is plausible that a counterfactual policy regime in India that tested chemicals for their toxicity to *at least* keystone species might have avoided the collapse of vultures.

In the absence of empirical estimates of the social benefits conferred by different species, conservation policy may be heavily influenced by existence values unrelated to utility. The vulture is not a particularly attractive bird and evokes rather different emotions at first sight than do more charismatic poster animals of wildlife conservation such as tigers and panda bears. Our results suggest that subjective existence values alone may not be the best way to formulate conservation policy.

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The Social Costs of Keystone Species Collapse: Evidence From The Decline of Vultures in India

Online Appendix

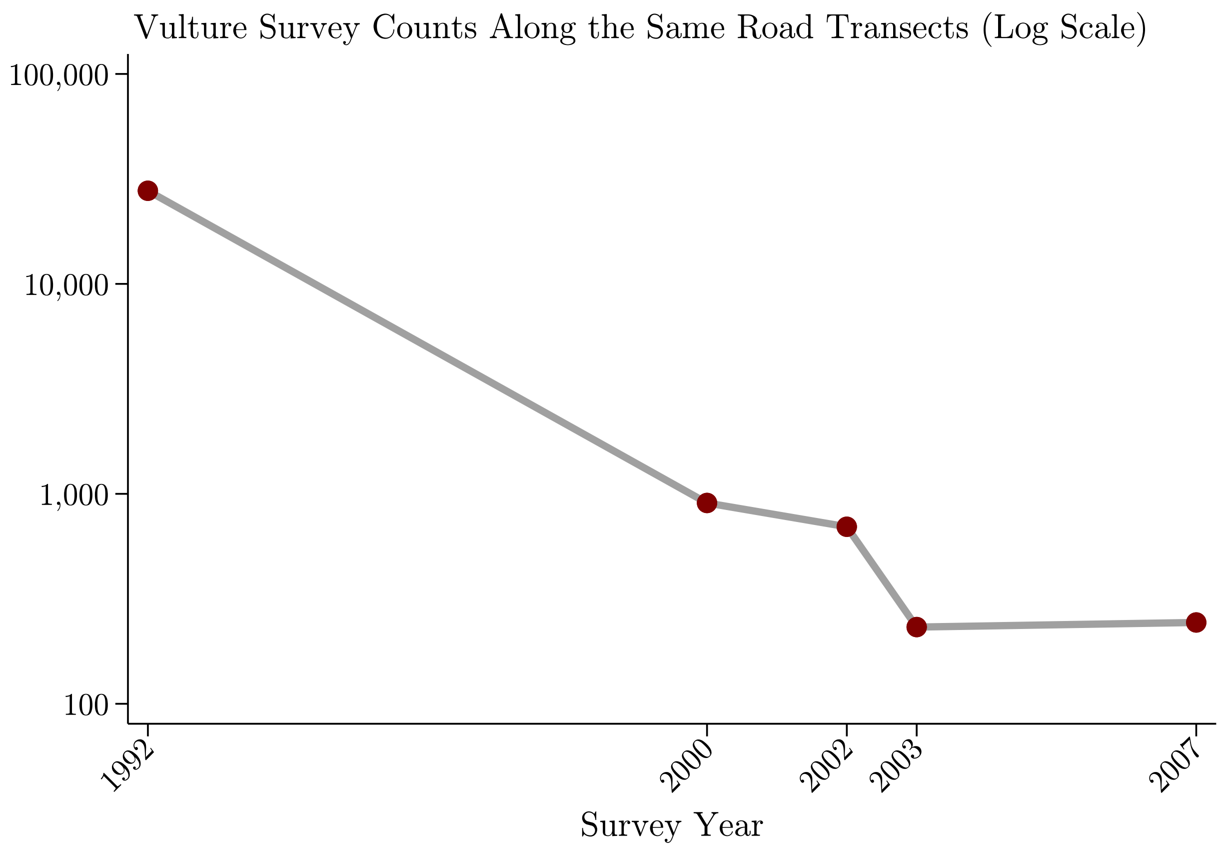
Eyal Frank
Anant Sudarshan

A Additional Results

A.1 Survey Results on Diclofenac-Affected-Vulture Populations

In the main text, we rely on bird observations at a national level to document a decline in vulture populations. However, the reported observations in the Global Biodiversity Information Facility (GBIF) database are likely upward biased as there was likely more attention given to documenting and reporting vultures after it became public knowledge that their populations were in decline. Unfortunately, there are no large-scale repeating surveys of vulture populations as they were always seen as too numerous to count. One exception is a repeating population survey that took place along 70 road transects during the years of 1992, 2000, 2002, 2003, and 2007. That data and survey methodology are reported in Prakash et al. (2007). While some survey years included additional road transects we only use the data from the 70 road transects that were repeatedly surveyed. In Figure A1, we plot the data from the repeated surveys as reported in Prakash et al. (2007), showing a large decline of three orders of magnitude from 1992 to 2007.

Figure A1: Vulture Counts From Repeated Surveys Along Road Transects



Notes: Each dot is the sum of surveyed vultures, in log scale, along the same 70 road transects for the three diclofenac-affected species. Data are reproduced from Prakash et al. (2007).

A.2 Validating Habitat Overlap Serves as a Population Level Proxy

In the analysis, we use the overlap of vulture habitats to define districts as high or low suitability, under the assumption that the degree of overlap captures meaningful information about the vulture population levels before they collapsed. To better support this interpretation of the habitat overlap as a meaningful baseline population proxy, we summarize here references from the ecological literature, as well as original data analysis. First, we present a short literature review regarding habitat suitability modeling—the underlying feature that generates the type of habitat maps used to construct the mean suitability score we use. Second, we combine data for hundreds of bird species on their habitat maps, and compare the habitat overlap to scientifically collected population-level data.

A.2.1 Previous Work on Habitat Suitability Modeling and Population Levels

We have reviewed the literature in the field of ecology that discusses the relationship between the habitat range a species can occupy, and as a result the number of sites a species occupies, to the population level of that species—their abundance. In analyzing 400 bird species across North America, Brown and Maurer (1987) write that the "area of the geographical range provides a measure of the breadth of tolerances and requirements of the individual units, and it characterizes the extent to which each species is able to use the total space available to the biota." While the literature discusses the heterogeneity in the strength of those relationships, it overwhelmingly concludes it is a positive relationship that is widely observed across taxonomic groups (Zuckerberg et al. 2009). Brown (1984) writes that "There appears to be a general relationship between abundance and distribution." Freckleton et al. (2006) state that "Positive abundance-occupancy relationships (a relationship between the number of sites a species occupies and the average density of individuals in occupied sites) are widespread through a range of taxa." Borregaard and Rahbek (2010) go as far as stating that "The positive relationship between a species' geographic distribution and its abundance is one of ecology's most well-documented patterns." Reviewing decades of papers on the topic, a meta-analysis by Weber et al. (2017) concludes that "In all cases we found a significantly positive relationship between abundance and suitability." In particular, the positive relationship between the size of the habitat and the abundance of a species has been observed in numerous studies for various bird populations (Järvinen and Sammalisto 1976; Hengeveld and Haeck 1982; Bock 1984; Lacy and Bock 1986; Brown and Maurer 1987).

A.2.2 Comparing Bird Habitat Overlap With Bird Breeding Survey Data

One of the primary limitations that habitat suitability models are used to resolve is the lack of wildlife population data. Often, such populations are not monitored, and if they are, records might be available across a small geographic range, and measurement is conducted either once or over long (decadal) time periods. As in our setting, prior to vultures declining, there was no population census that would allow us to establish baseline vulture abundance.

A key exception to this scarcity of data is the Breeding Bird Survey (BBS) conducted by the United States Geological Survey (USGS) (Ziolkowski et al. 2022). Since 1966, the USGS (in partnership with Environment Canada's Canadian Wildlife Service) has been collecting data on the presence and abundance of bird species in North America. Unlike other wildlife population data products such as eBird, the BBS relies on the repeated use of sampling protocols along the same road transects, and sampling is conducted by individuals who received training. In 1999, the coverage of the sampling area was greatly expanded. This data product allows us to test the key assumption regarding the correlation between habitat overlap and population levels.

We collect the habitat range maps data from BirdLife International (BLI)—the same source of habitat range maps as we use for the vulture species. We focus on the habitats that intersect with the contiguous US. For each county in the US, we calculate its overlap with the habitat of each bird species, for the species in the BLI repository that have a non-zero overlap with the contiguous US. Then, we use the geocoded data from the BBS to construct the mean count of birds for each species by county over the time period of 1999 to 2019.[31] We were able to match bird counts to habitat overlap scores for 524 species. Of those, we keep in the sample bird species that overlap with at least 30 counties, resulting in a total of 376 unique bird species, and 1,907 unique counties.

There is a clear increasing relationship between bird abundance and habitat overlap. We document this both descriptively, as well as in a cross-sectional regression. In Figure A2, we plot the local polynomial fit over all 376 bird species, across 1,907 counties, documenting the positive relationship between the counts of bird species and their habitat overlap. We fit the local polynomial over all the data, as well as sub-samples that exclude county-species pairs where the overlap is either above 95% or 90%—recovering an almost identical pattern over the overlapping ranges.

We further test the magnitude and precision of this abundance-habitat correlation using a simple cross-sectional regression. We regress the mean log of mean abundance on the continuous measure of overlap share (in percent), as well as quantiles of the overlap measure. In Table A1, we summarize these results for both the full sample, and the sub-sample which excludes overlap values above 95%. For a one percentage point increase in habitat overlap, mean abundance is associated with an increase of 0.68 percent in the full sample (column 1), or of 0.49 in the sub-sample (column 4). We split the habitat overlap score into terciles separately for each species.[32] Locations with high (third tercile) and medium (second tercile) habitat overlap values are associated with a 42% and 28.4% higher mean abundance relative to the first tercile (column 2). These correlations are precisely estimated, allowing us to reject the null hypothesis of no difference between the high and medium terciles. This pattern holds in the sub-sample as well, albeit with smaller coefficients (column 5). Finally, we use the same definition as the one in the paper, where we classify the high and medium terciles as the high suitability locations. We estimate that high suitability locations are precisely associated with 33.3% higher mean abundance in the full sample (column 3), and 19.7% in the sub-sample (column 6). In conclusion, the regression analysis recovers large differences in baseline populations, as reflected in the mean abundance values, across the gradient of habitat overlap.

[31] We exclude data from 2020 to 2021 because bird presence and counts might have responded to the lockdowns during the COVID-19 pandemic. Such responses have been documented in India by Madhok and Gulati (2022).

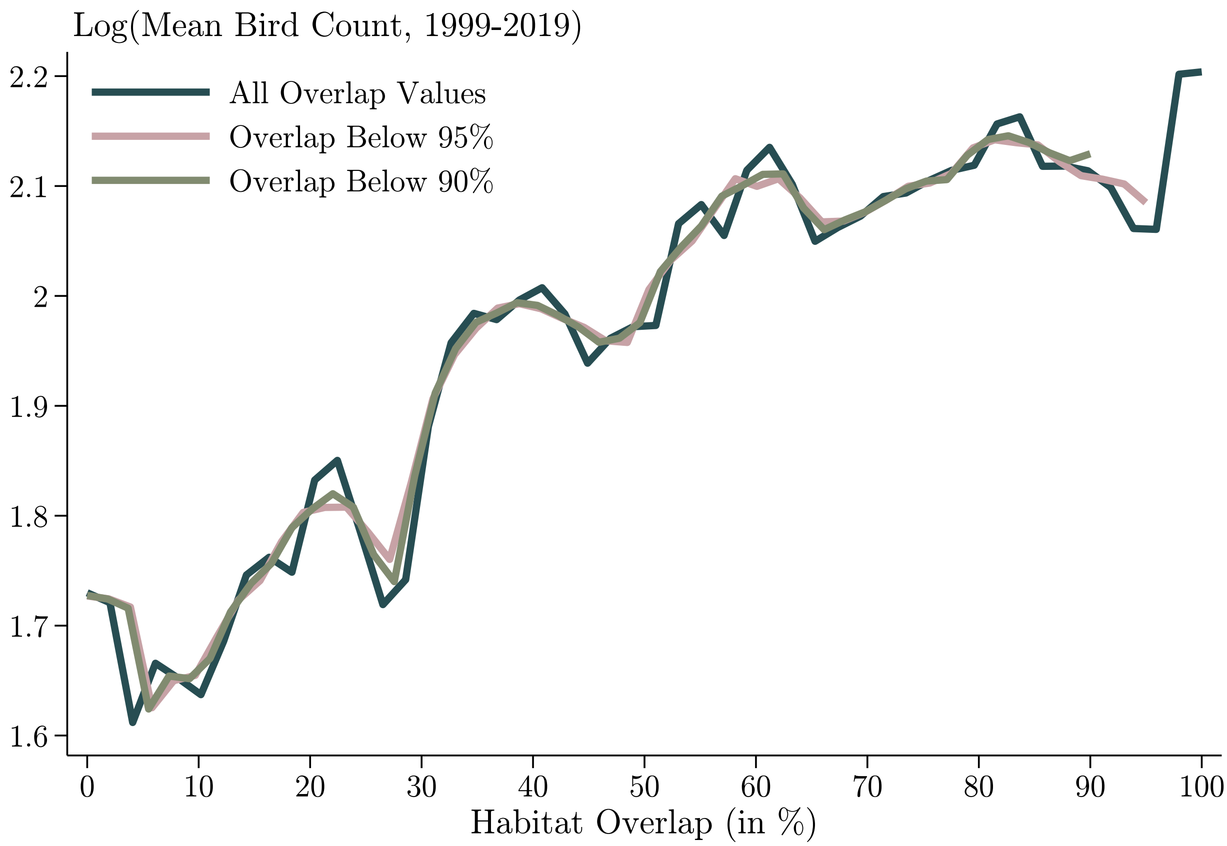
[32] We calculate the terciles over the species-specific distribution instead of the global, all-species, distribution.

Table A1.
Relationship Between Log(Mean Bird Counts, 1999-2019) & Habitat Overlap

	All Overlap Values			Overlap Values $\leq 95\%$		
	(1)	(2)	(3)	(4)	(5)	(6)
Habitat Overlap Share	0.68			0.49		
	(0.05)			(0.04)		
Habitat Overlap Tercile (M)		0.25			0.17	
		(0.02)			(0.04)	
Habitat Overlap Tercile (H)		0.35			0.27	
		(0.03)			(0.07)	
High Suitability (Terciles M or H)			0.29			0.18
			(0.02)			(0.04)
R^2	0.440	0.434	0.434	0.452	0.444	0.444
N	174,267	174,267	174,267	33,062	33,062	33,062
County Clusters	1,907	1,907	1,907	1,818	1,818	1,818
Bird Clusters	376	376	376	371	371	371

Notes: We use data for the contiguous United States on mean bird population counts from 1999 to 2019 (USGS Breeding Bird Survey), which we aggregate by county. We match each county to the habitat overlap of that county to each species, using data from BirdLife International. North America provides a setting where scientific measurement of bird populations has been carried out for a long period of time, allowing us to validate the assumption that habitat overlap provides a useful proxy for baseline wildlife bird populations. Each regression includes county and bird species fixed effects. Standard errors are clustered at both the county and bird species levels.

Figure A2: Local Polynomial Fit for Mean Bird Abundance & Habitat Overlap



Notes: Summarizing the relationship between the mean bird counts (in log points) and the habitat overlap (in percent), for all counties and bird species in the sample (see text for more details).

A.3 Vulture Habitat Overlap & Treatment Assignment

Here we provide more details about the distribution of the mean habitat overlap for the three vulture species—the suitability score—and report results for different choices of assigning treatment status based on the suitability score. In Section A.12, we also consider an alternative approach to constructing the suitability score using the BIOCLIM habitat suitability model.

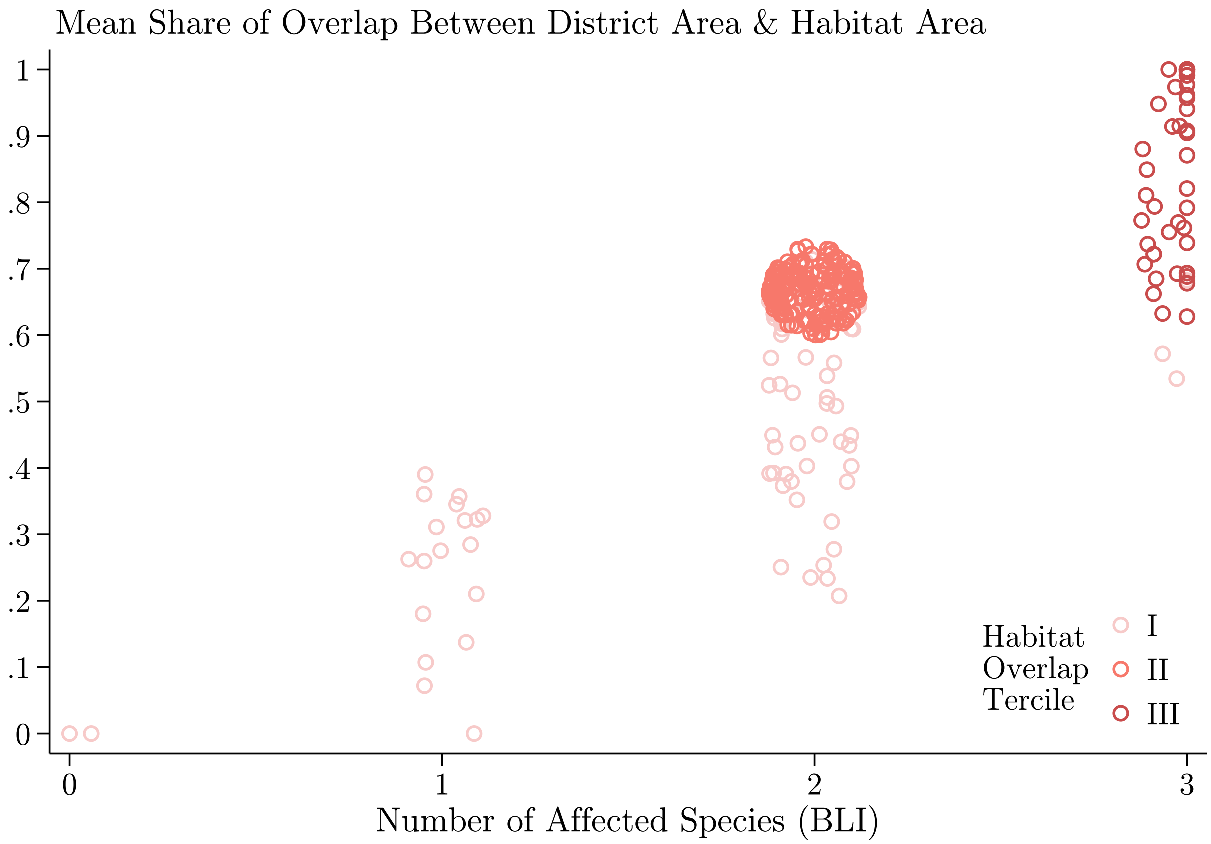
A.3.1 Distribution of Suitability Score and the Number of Species

In Figure A3, we plot the relationship between the mean overlap of the district area and the three affected species, relative to the number of affected species that have a non-zero overlap with the district. Each dot is the suitability score (the mean overlap of the habitat area and the district area) for a specific district. To allow for easier visual inspection, the dots are jittered. We color each dot according to its habitat overlap tercile.

The highest tercile is composed of districts that overlap with all three species, and have mean area overlap between the habitats and the district of 66% and higher. The second tercile is largely composed of districts that overlap almost entirely with two species (hence the bunching around 66%), and two districts that overlap partially with all three districts. Finally, the lowest tercile is composed of districts that overlap with either two, one, or none of the species. The lowest tercile covers the range of zero overlap up to approximately 60% overlap.

In the full sample, for all mainland districts, 111 districts are in the first (lowest) tercile, 184 districts are in the second tercile, and 45 districts are in the third (highest) tercile. In the main sample we use in the analysis, where we restrict the sample to a balanced sample with respect to the all-cause death rate, those numbers are 78, 71, and 4, respectively. In other words, for the main sample, our classification of high and low vulture suitability assigns 75 districts as high, and 78 districts as low suitability.

Figure A3: Mean Habitat Overlap VS. Number of Species



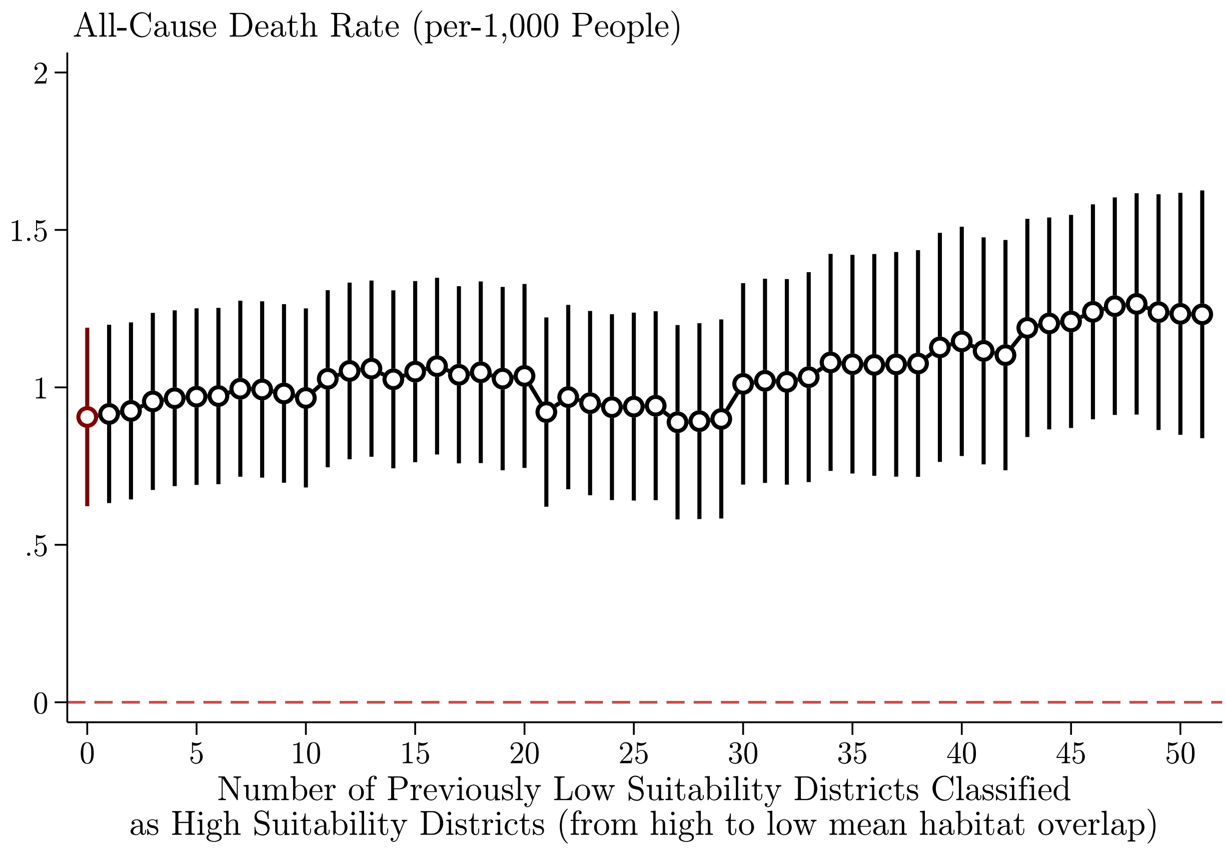
Notes: Summarizing the mean overlap of the habitat area and the district area for each district, relative to the number of species that have their habitats intersect with the district. Each dot is a separate district. Dots are jittered to allow for easier visual inspection.

A.3.2 Sensitivity to Treatment Assignment Definition

Our main classification of treatment assigns districts in the second and third terciles as high vulture suitability districts. However, as seen in Figure A3, there are several districts that have similar mean habitat overlap values, around 0.66. Here we report how sensitive the results are to changing the threshold that assigns treatment status. Explicitly, we use the ranking of the mean habitat overlap in the low vulture suitability group to incrementally shift more and more districts from the control to the treatment group. Each time we re-estimate the regression (that corresponds to the result in Table 2, Panel A, column 1).

The results are robust to shifting districts that were previously in the control group into the treatment group. In Figure A4, we report the coefficients and 95% CIs from repeatedly lowering the treatment assignment cutoff. Across all the results, we still recover a precisely estimated increase in the all-cause death rate. As more districts get reclassified from low to high vulture suitability, the effect size increases slightly, but remains within the confidence interval of the unmodified estimate. This result agrees with the interpretation that even some of the low vulture suitability districts are experiencing treatment, and by comparing high to low vulture suitability districts, we are recovering a lower bound of the effect.

Figure A4: Evaluating Sensitivity to Treatment Assignment of Districts



Notes: Repeating the estimation in Equation (2) but changing the treatment assignment such that more and more low vulture suitability districts are classified as treated. The result in maroon reflect the unmodified treatment assignment rule, which we use in the main text.

A.4 Pooled Estimation Results When Including State-Level Trends

In the main text, we report pooled results for the specification with the zonal council-by-year fixed effects, and then disentangle the pooled coefficient into two periods, 1994-1999 and 2000-2005, when including state-level trends. Here we report pooled results for the specifications with state-level trends.

The pooled estimation recovers large and meaningful increases in the all-cause death rate, even when we include state-linear trends, or state-by-year fixed effects. In Table A2, columns 2-3 and 5-6, we summarize the pooled results that correspond to the results in Table 2, Panel A, columns 3-4 and 7-8. To make comparison easier, we repeat the pooled result from the specification without state-level trends (columns 1 and 4). As expected, the coefficients in Table A2, columns 2-3 and 5-6, are higher than the 1994-1999 coefficients, but lower than the 2000-2005 coefficients. The estimated effect is larger in the census urban sample (columns 5 and 6), relative to the combined sample (columns 2 and 3).

Table A2.
Pooled Estimation Results
All-Cause Death Rate, per-1,000 People

	Combined Sample ($\overline{Y} = 10.2$)			Census Urban Sample ($\overline{Y} = 7.2$)		
	(1)	(2)	(3)	(4)	(5)	(6)
HVS×Post-1994	0.91	0.27	0.31	1.04	0.48	0.50
	(0.14)	(0.13)	(0.14)	(0.27)	(0.22)	(0.20)
R^2	0.74	0.77	0.80	0.67	0.70	0.76
N	2,754	2,754	2,700	2,808	2,808	2,754
Clusters	153	153	150	156	156	153
Zonal Council-by-Year FE	X			X		
State-Linear Trends		X			X	
State-by-Year FE			X			X

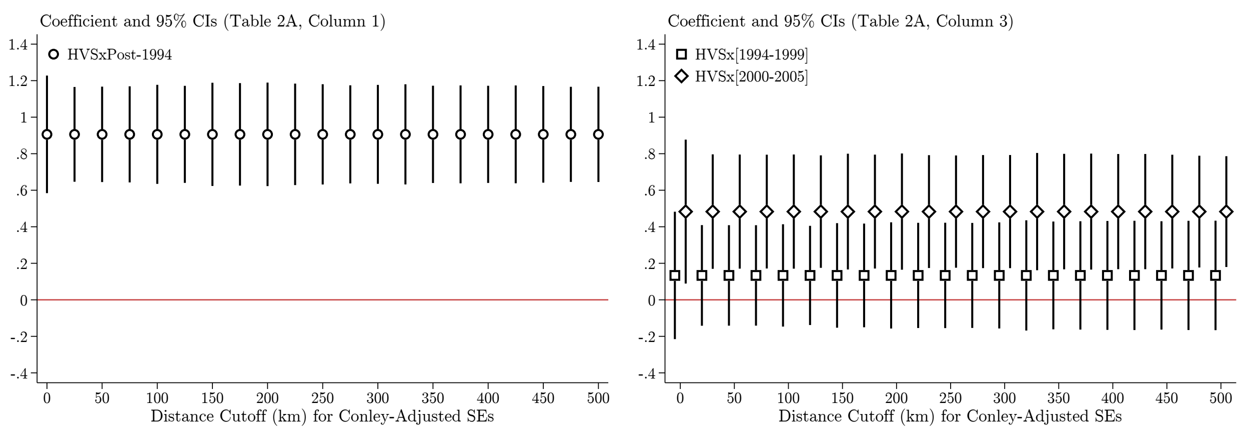
Notes: Estimation results for the specification in Equation (2). Comparing high-vulture-suitability (HVS) to low-vulture-suitability districts, after the collapse of the affected vulture populations. When we include state-by-year fixed effects (columns 4 and 8), three states get dropped as they have no district-level data. Reported means of 10.2 and 7.2 deaths per 1,000 people are for the pre-treatment period of 1988 to 1992. Sample includes balanced district-level data from 1988 to 2005. All regressions include district fixed effects. Observations are population-weighted. We report Conley standard errors that are serially correlated at the district level, and are allowed to be spatially correlated up to 200km.

A.5 Verifying Conley Standard Errors Are Not Sensitive to Bandwidth Choice

The precision of the estimates increases when we adjust for spatial correlation up to 200 km (Conley 1999; Hsiang 2010; Alan Bester et al. 2016). In the main text, as well as in the results reported in the Online Appendix, we report results that allow for spatial correlation up to 200 km. In Figure A5, we confirm that the choice of the bandwidth does not meaningfully change the precision once we adjust for spatial clustering.

Figure A5: Adjusting Standard Errors for Spatial Correlation

(a) Table 2A, Column 1 (b) Table 2A, Column 3



Notes: We report how the Conley standard errors change when we use different bandwidths for the spatial correlation. We narrow the attention to the two results that are the focus in the main text.

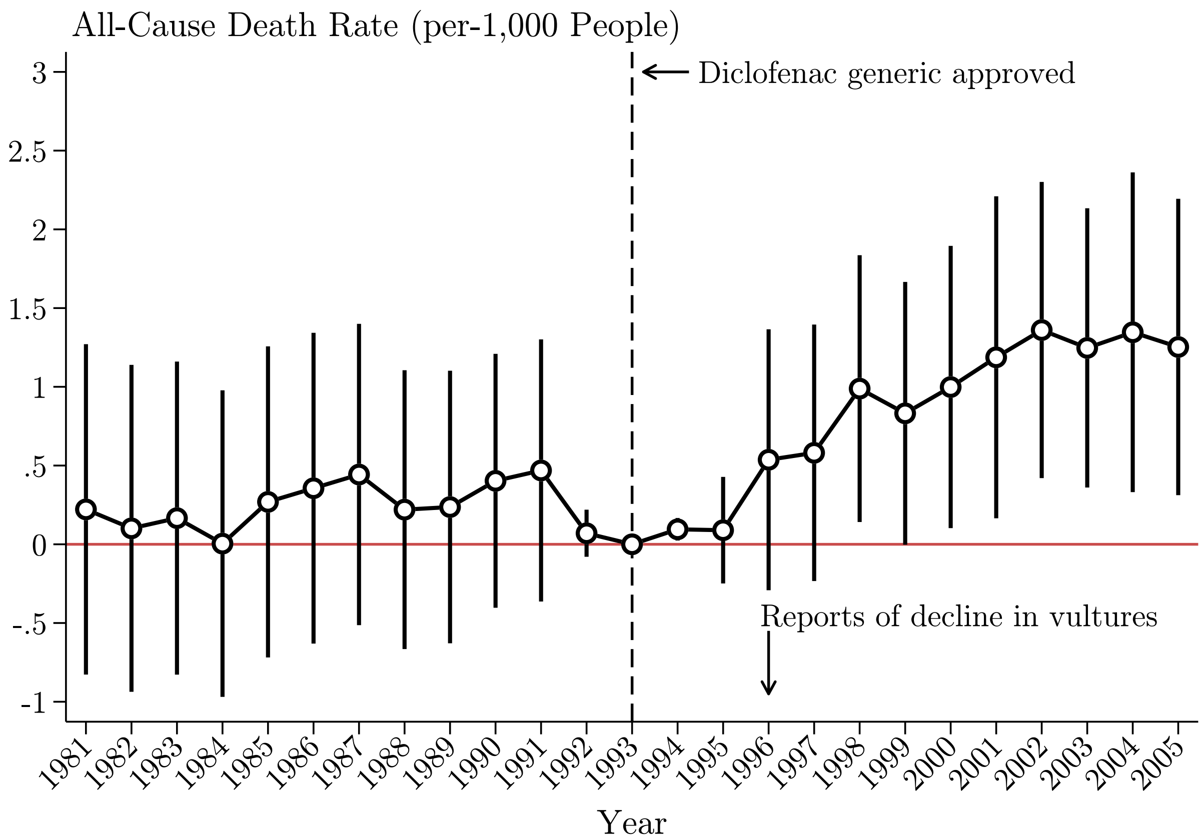
A.6 Extending the Panel to Cover 1981 to 2005

In the main text, we use the data from 1988 to 2005 for two main reasons. First, there was an abrupt shift in the reporting regime in 1988 when the vital statistics started reporting vital event counts instead of rates. We prefer to use data reported under the same regime, as this allows us to fully control the conversion to rates. Second, the number of districts that are fully balanced from 1988 to 2005 is 153, while there are only 104 balanced districts for the 1981 to 2005 period. When extending the panel to the full 1981 to 2005 period, and losing 33% percent of the districts, we recover similar results to those in the main text (Figure A6). Specifically, we do not observe a differential time trend in the years leading to the collapse in vulture populations, and find that death rates increase in the high-vulture-suitability areas only in the years after the collapse.

A.7 Accounting for State-Level Temporal Trends

To account for potential differential trends in the reporting of vital statistics data that systematically change by state, we repeat the estimation in Equation (1) and include either

Figure A6: All-Cause Death Rates DD Estimation Results With Earlier Years

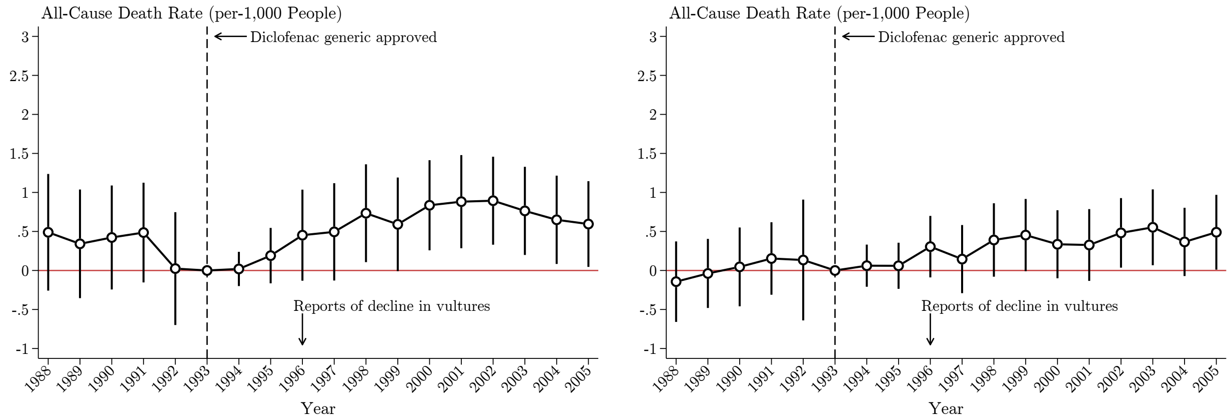


Notes: Estimation results from Equation (1). Comparing the third and second terciles to the first tercile of vulture habitat overlap. Expanding the sample to 1981, while still using a balanced sample, lowers the number of districts from 153 to 104. The regression includes district and zonal council-by-year fixed effects. Observations are population-weighted. We calculate Conley standard errors that are serially correlated at the district level, and are allowed to be spatially correlated up to 200km.

state-linear trends, in addition to the zonal council-by-year fixed effects, or include state-by-year fixed effects. The inclusion of state-level trends potentially absorbs a large share of the signal of interest as there is little sub-state variation in habitat suitability overlap. Even with the inclusion of flexible time trends that vary by state, we recover similar patterns in Figure A7 to those in Figure 4. The divergence in death rates only starts after the vulture populations collapse, yet the magnitude of the effect is smaller. By 2000, all-cause death rates were about 0.5 or 0.3 deaths per 1,000 people higher in the high-vulture-suitability districts when including state-linear trends, or state-by-year fixed effects, respectively.

Figure A7: All-Cause Death Rates DD Estimation Results With State-Level Trends

(a) State-Linear Time Trends (b) State-by-Year Fixed Effects



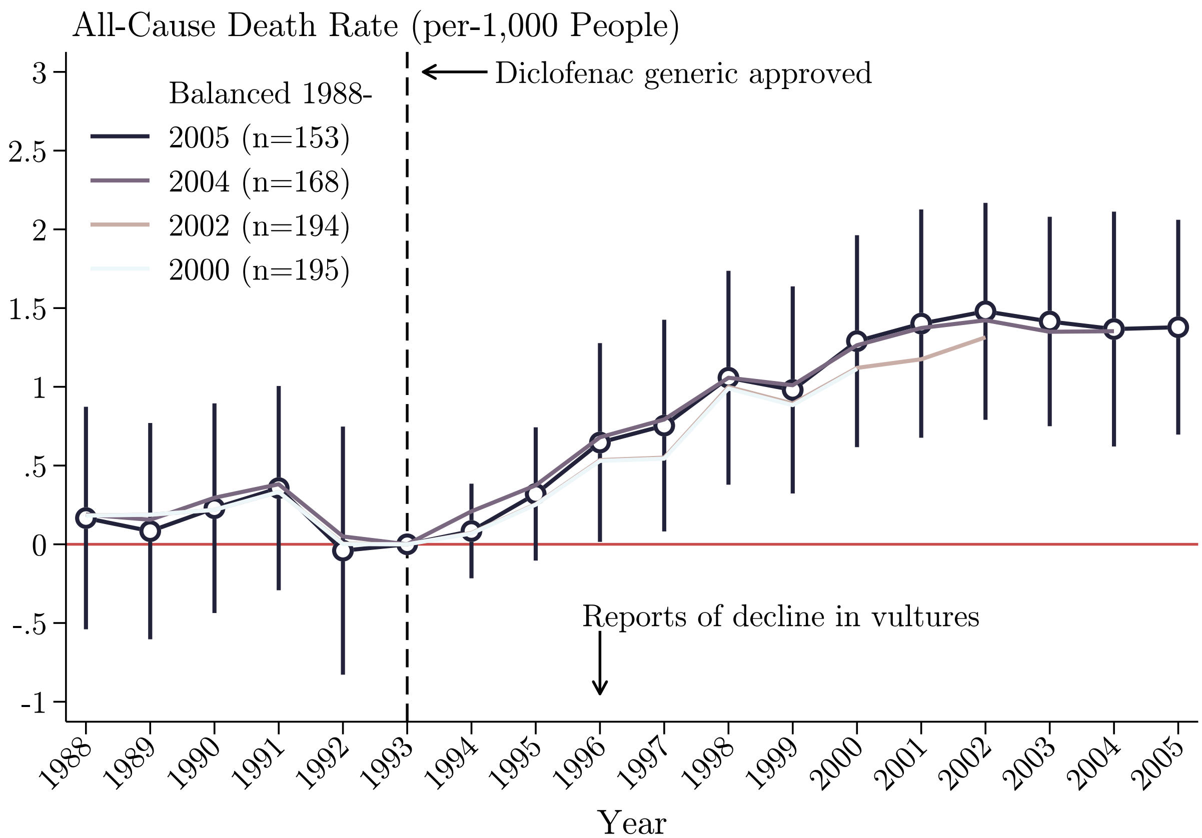
Notes: Estimation results from Equation (1). Comparing the third and second terciles to the first tercile of vulture habitat overlap. All regressions include district fixed effects. The regression in (a) includes zonal council-by-year fixed effects and state-level linear time trends, and the regression in (b) includes state-by-year fixed effects. Observations are population-weighted. We calculate Conley standard errors that are serially correlated at the district level, and are allowed to be spatially correlated up to 200km.

A.8 Increasing the Number of Balanced Districts in the DD Estimation

In the main text, we present results from three different estimation approaches: differences-in-differences (DD), long-differences (LD), and triple-differences (DDD). The DD and DDD estimations use a balanced sample from 1988 to 2005. Because many districts are missing all-cause death rate data at either the end of the sample period, or in the middle of the sample period, our balanced DD sample, for example, covers 153 of the 342 districts (held at 1981 borders). In the paper, we address this gap in coverage by relaxing the requirement for districts to have fully balanced data. Instead, we require districts to have at least one year of non-missing data before 1995 (including), and one year of non-missing data after 2000 (including). We use this sample in a long-differences estimation.

To verify that our results in the DD estimation are not sensitive to the composition of the sample, we run additional analysis where we truncate the sample earlier to allow for more districts to qualify as having balanced data. In Figure A8, we plot the DD estimation results from the main DD sample (same as in Figure 4), along with the event-study estimates of three new samples. In each new sample, we truncate the sample at 2004, 2002, or 2000. This increases the number of balanced districts from 153 to 168, 194, or 195, respectively. The results from each of these new samples track the main results very closely, and are all well within the 95% CIs.

Figure A8: All-Cause Death Rates DD With Different Terminal Years



Notes: Estimation results from the sample used for the DD analysis in the main text (as in Figure 4), with lines showing the event-study estimates of the same specification but balancing the data up to 2004, 2002, and 2000, instead of 2005, with the number of balanced districts of 168, 194, and 195, instead of 153, respectively

A.9 Decomposing the Triple-Differences Estimation

In the main text, we report results from a triple-differences approach, which uses livestock to test whether the mortality response is more pronounced in areas with both high-vulture-suitability and high levels of livestock at baseline. Another way to undertake the comparison embedded in the triple-differences is to separately evaluate two diff-in-diff components. In Table A3, we split the sample into high- and low-vulture-suitability districts. We then carry out a diff-in-diff exercise comparing districts with high vs low levels of baseline livestock, before and after the veterinary use of diclofenac.

This decomposition reveals that it is only in high-vulture-suitability districts that baseline levels of livestock are associated with elevated death rates in the post period (compare Table A3, columns 1 and 2 in Panels A and B). In areas where vultures were expected to be absent or less abundant, there is no differential change in mortality between low and high livestock districts. This is true even when restricting attention to urban portions of districts—here the difference is even more marked (Table A3, columns 3 and 4).

This decomposition analysis also alleviates concerns regarding possible spatial clustering of the vulture suitability as it offers an alternative research design. The diff-in-diff results in Table A3, Panel A, only rely on districts in the more central parts of India, where environmental conditions are similar in terms of vulture habitats. This alleviates concerns that coastal or mountainous districts might drive our results.

A.10 Back of the Envelope Calculation on Annual Cow Removal by Vultures at Baseline

In the main text, we highlight that we are comparing districts that suffered a large loss of vultures to districts that likely experienced a smaller loss of vultures. Here we provide a back-of-the-envelope calculation to better interpret the differences between districts in the treatment and control groups. Each vulture consumes four to six kg of meat each week (Ishwar et al. 2016). In our calculations below, we assume each vulture consumes five kg of meat a week, and therefore 260 kg of meat annually. Scientists have estimated there were between 30 to 50 million vultures before their population levels collapsed (Subramanian 2011). We choose the middle number of 40 million vultures as a baseline population level—resulting in an estimated annual meat consumption of 10.4 billion kg.

The mean weight of the Indian Gir cow is 385 kg (Felius 1995) implying that this amount of meat is the equivalent of removing 27,012,987 cows each year. This number strongly agrees with a separate report that estimates that vultures in India removed 25 million carcasses each year (Ishwar et al. 2016).

Our next step is to disaggregate this national-level approach to the district level. To do so, we first need to assign a baseline vulture population level to each district, and then use the above values to calculate the amount of cow-equivalent carrion removed each year. First, we calculate the total habitat area across all three affected species in each district. Second, we sum up all of the habitat areas across all the affected species and all districts. We then divide the first measure by the second to arrive at a set of district weights that sum up to one. A district will have a higher weight the more area it has that is suitable for vultures.[33] We multiply each weight by the baseline population estimate of 40 million vultures. Then, we multiply this estimated number of vultures in each district by the mean amount of meat (denoted in cow-equivalent carcasses) they remove each year: $\frac{385}{5} \times 52$. Finally, we sum the number of estimated cows removed each year by mean habitat overlap tercile (see Figure A3), and normalize it by geographic area, or by population density.

We summarize the back-of-the-envelope calculations in Table A4. In our treatment group, the sum of estimated removed cow-equivalent carcasses by vultures (the row that sums up the high and middle terciles) is about three times larger than the value in the lowest tercile (columns 1 and 3), or 50% larger than the lowest tercile (column 2). While this is a crude quantification of the removal of carrion by vultures, it highlights that the differences between the treatment and control groups are not trivial, and are in fact quite large and meaningful.

[33] Note that habitat areas can overlap, resulting in the total habitat area in a district being larger than the area of the district. This is not an issue because we normalize by the sum of all habitat areas across the country, which accounts for this potential habitat overlap.

Table A3.
Decomposing the DDD Results by Districts' Vulture Suitability

Panel A. High Vulture Suitability Subsample

	Combined Sample ($\overline{Y} = 10.2$)		Census Urban Sample ($\overline{Y} = 7.2$)	
	(1)	(2)	(3)	(4)
Livestock×Post-1994	0.62	0.57	1.07	1.05
	(0.19)	(0.19)	(0.30)	(0.28)
R^2	0.83	0.83	0.76	0.77
N	1,350	1,350	1,386	1,386
Clusters	75	75	77	77

Panel B. Low Vulture Suitability Subsample

	(1)	(2)	(3)	(4)
Livestock×Post-1994	0.08	0.11	-0.07	-0.12
	(0.21)	(0.21)	(0.48)	(0.48)
R^2	0.68	0.69	0.60	0.61
N	1,404	1,404	1,404	1,404
Clusters	78	78	78	78
Weather Controls		X		X

Notes: Estimation results for a specification similar to Equation (2). The estimation is comparing districts with high to low livestock agriculture at baseline, after the collapse of the affected vulture populations. We repeat the analysis in two subsamples: The districts with high vulture suitability where we expect high baseline livestock to affect health (Panel A), and low vulture suitability where we do not expect baseline livestock to affect health (Panel B). Sample includes balanced district-level data from 1988 to 2005. Reported means of 10.2 and 7.2 deaths per 1,000 people are for the pre-treatment period of 1988 to 1992. All regressions include district and zonal council-by-year fixed effects. Observations are population-weighted. We report Conley standard errors that are serially correlated at the district level, and are allowed to be spatially correlated up to 200km.

Table A4.
Back-of-the-Envelope Calculation on
Annual Cow Removal by Vultures

Tercile	(1)	(2)	(3)
L	7.3	6.4	1,787.2
M	16.7	9.3	3,784.7
H	3.1	11.8	1,904.3
H+M	19.8	9.7	5,471
Total, Unadjusted	X		
Normalized by Area		X	
Normalized by Population/Area			X

Notes: In column 1, units are millions of cows. In column 2, units are cows per squared km. In column 3, units are millions of person-cows per squared km. In column 1, the fourth row simply sums the second and third rows, but in columns 2 and 3, we first sum the removed cows in M and H districts, and then divided by the area or population/area in M and H districts. See text in the Online Appendix for details on calculations and assumptions.

A.11 Examining Heterogeneity Between Census Urban & Rural District Areas

In Table A5, we explore the degree to which death rates respond differently to the collapse in vulture populations in either census urban or census rural areas. Because census urban areas have larger populations, are denser, and are more likely to have an animal landfill site at their outskirts, we expect that a larger portion of the average effect is driven by the census urban areas. When we use the district-level data reported by census urban or census rural areas, we find a higher average treatment effect in census urban relative to census rural areas, but the effects are not statistically different from each other.

A.12 Using Habitat Suitability Model to Define Treated Districts

In the main analysis, we rely on the habitat range maps, as produced by BirdLife International (BLI), to classify districts as either high or low suitability for the diclofenac-affected vultures. One concern is that the maps heavily rely on biased samples and local knowledge which places more weight on populated areas. To alleviate these concerns, and to examine the sensitivity of the classification to the maps by BLI, we estimate our own version of a habitat suitability model (HSM). In general, habitat suitability modeling uses data on the presence records of species along with a range of environmental variables in order to characterize the environmental niche that a species can occupy. An HSM will use observations of polar bears and conclude that cold tundras are a more likely habitat than tropical forests, or that mountain goats are more likely to be found in high-elevation areas than in flat plains.

Table A5.
All-Cause Death Rate, per-1,000 People

Panel A. Census Urban Sample ($\overline{Y} = 7.2$)

	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)
HVS×Post-1994	1.04			0.48			0.50		
	(0.27)			(0.22)			(0.20)		
HVS×[1994, 1999]		0.68	0.72		0.35	0.40		0.34	0.32
		(0.30)	(0.29)		(0.26)	(0.26)		(0.22)	(0.22)
HVS×[2000, 2005]		1.34	1.32		0.68	0.67		0.63	0.64
		(0.30)	(0.25)		(0.23)	(0.22)		(0.24)	(0.25)
R^2	0.67	0.68	0.69	0.70	0.70	0.71	0.76	0.76	0.76
N	2,808	2,808	2,808	2,808	2,808	2,808	2,754	2,754	2,754
Clusters	156	156	156	156	156	156	153	153	153

Panel B. Census Rural Sample ($\overline{Y} = 11$)

	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)
HVS×Post-1994	0.79			0.11			0.22		
	(0.17)			(0.16)			(0.16)		
HVS×[1994, 1999]		0.35	0.32		-0.06	-0.02		0.13	0.12
		(0.17)	(0.17)		(0.17)	(0.18)		(0.17)	(0.18)
HVS×[2000, 2005]		1.21	1.08		0.40	0.35		0.30	0.27
		(0.23)	(0.22)		(0.19)	(0.19)		(0.18)	(0.19)
R^2	0.73	0.73	0.74	0.75	0.76	0.76	0.80	0.80	0.80
N	2,916	2,916	2,916	2,916	2,916	2,916	2,862	2,862	2,862
Clusters	162	162	162	162	162	162	159	159	159
Zonal Council-by-Year FE	X	X	X						
State-Linear Trends				X	X	X			
State-by-Year FE							X	X	X
Weather Controls			X			X			X

Notes: Estimation results for the specification in Equation (2). Comparing high-vulture-suitability (HVS) to low-vulture-suitability districts, after the collapse of the affected vulture populations. When we include state-by-year fixed effects (columns 7 to 9), three states get dropped as they have no district-level data. Reported means of 7.2 and 11 deaths per 1,000 people are for the pre-treatment period of 1988 to 1992. Sample includes balanced district-level data from 1988 to 2005. All regressions include district fixed effects. Observations are population-weighted. We report Conley standard errors that are serially correlated at the district level, and are allowed to be spatially correlated up to 200km.

We use the well-known BIOCLIM HSM that was first developed in 1984 (Booth et al. 2014). The model uses data on the presence of a species, and links those records to local bioclimatic variables such as the elevation, temperature, and precipitation. The model uses weather data from several seasons on the mean, max, and min values. Overall, the standard application uses 19 such variables. Combining the data on the bioclimatic variables and presence records, the model constructs the convex hull of environmental conditions that appear to be beneficial for the presence of the species. Using that classification, the model then projects that convex hull back into geographic space to construct suitability scores. The higher the score, the more likely the area is a suitable niche for the species.

We use observation records from eBird and from the Global Biodiversity Information Facility (GBIF) to construct the BIOCLIM suitability scores. We then take the mean level of the suitability scores across all three affected species, and use it to define high and low suitability dummy variables. We either split the suitability score into terciles, defining high suitability as the third and second terciles, or we define the high suitability dummy as being above the median suitability score.

Using these alternative definitions of the treated districts, we re-estimate the specifications in Equations (1), (2), and (3). We report the maps showing the classification of districts, along with the event-study results in Figure A9, and the average treatment effects in Table A7. Across the two alternative treatment classification schemes, we recover similar magnitudes for the change in all-cause human death rates following the collapse in vulture populations. This helps us to reject that our analysis is extremely sensitive to the exact classification of districts in either treatment or control status.

A.13 Long-Differences Results With Time-Varying Controls

In the main text, we report results for the long-differences model in Table 3. This allows us to include additional districts even if they do not have fully balanced data. In Figure A12, we report a series of precisely estimated zero differences for a variety of outcomes between the high and low vulture suitability districts, before and after the collapse of the vulture populations. Here we combine the two and use the time-varying variables we use in Figure A12 with the long-differences model.

We recover similarly meaningful and precisely estimated increases in the all-cause death rate for the high vulture suitability districts after the collapse. In Table A6, we report the estimation results of the long-differences model when including each set of time-varying controls, as well as when we include all the time-varying controls in the same regression. The estimation results are fairly insensitive to the included controls.

A.14 Additional Water Quality Parameters

Here we report additional results on water quality for biological and chemical oxygen demands (BOD and COD), as well as turbidity. In general, as the demand for oxygen in the water system increases with more substances that react with it, we see dissolved oxygen levels decline (as seen in Table 5), as well as increasing levels of BOD and COD. Because BOD only captures biological uses of oxygen, it will be below the COD level which captures

Table A6.
Long Differences Results With Controls

	(1)	(2)	(3)	(4)	(5)	(6)	(7)
HVS×Post-1994	0.40	0.48	0.45	0.60	0.62	0.48	0.50
	(0.20)	(0.20)	(0.20)	(0.20)	(0.21)	(0.21)	(0.20)
Age Shares	X						X
Village Infrastructure		X					X
Literacy Rate			X				X
Employment Rate				X			X
Employment Shares					X		X
Healthcare Access						X	X
R^2	0.92	0.90	0.89	0.88	0.88	0.89	0.93
N	638	626	626	626	606	634	604
Clusters	319	313	313	313	303	317	302

Notes: Estimation results for the specification in Equation (2). Sample includes all districts with non-missing data in the collapsed pre- and post-collapse sample (combined sample of census-urban and census-rural areas). We include a series of control variables for which we have observations both before and after 1994. Column 1 controls for population age shares in five-year intervals. Column 2 controls for the share of villages that have roads, have medical facilities, have educational facilities, have communications infrastructure, have electrical power infrastructure, and have access to irrigation. Column 3 controls for the literacy rate. Column 4 controls for the total employment rate. Column 5 controls for employment shares in agriculture, manufacturing, and services. Column 6 controls for the number of doctors, health workers, health centers, and hospitals per 100,000 people. Column 7 combines all the controls from columns 1 to 6. Each regression includes district and zonal council-by-year fixed effects. Observations are population-weighted. We report Conley standard errors that are serially correlated at the district level, and are allowed to be spatially correlated up to 200km.

Table A7.
Results for All-Cause Death Rate Using
BIOCLIM Classifications ($\overline{Y} = 10.2$)

Panel A. High & Medium Suitability Score Terciles

	(1)	(2)	(3)	(4)
HVS×Livestock×Post-1994			0.70	0.62
			(0.27)	(0.26)
HVS×Post-1994	0.62	0.54	0.13	0.10
	(0.15)	(0.12)	(0.18)	(0.17)
Livestock×Post-1994			-0.05	-0.02
			(0.22)	(0.22)
R^2	0.73	0.74	0.74	0.74
N	2,754	2,754	2,754	2,754
Clusters	153	153	153	153

Panel B. Above Median Suitability Score

	(1)	(2)	(3)	(4)
HVS×Livestock×Post-1994			0.77	0.75
			(0.27)	(0.29)
HVS×Post-1994	0.62	0.55	0.01	-0.03
	(0.15)	(0.15)	(0.20)	(0.23)
Livestock×Post-1994			0.11	0.10
			(0.17)	(0.18)
R^2	0.73	0.74	0.74	0.74
N	2,754	2,754	2,754	2,754
Clusters	153	153	153	153
Weather Controls		X		X

Notes: Estimation Results for the specification in Equations (2) and (3). The treatment classification uses predicted suitability scores for the diclofenac-affected-vultures from the BIOCLIM habitat suitability model. We either split the suitability score into terciles and define treated districts as the third and second terciles (Panel A), or split districts as above or below the median suitability score, and define treated districts as those above the median (Panel B). Sample includes balanced district data, combining urban and rural areas, from 1988 to 2005. Reported mean of 10.2 deaths per 1,000 people is for the pre-treatment period of 1988 to 1992. Observations are population-weighted. We report Conley standard errors that are serially correlated at the district level, and are allowed to be spatially correlated up to 200km.

both organic and inorganic uses of oxygen. We should expect to see both BOD and COD levels increase with greater availability of carrion in the environment.

Turbidity is a measure of water quality that generally shows improvement in water quality as it goes down. However, in the case of a decline in scavengers, turbidity declines as well. This is because scavengers tend to increase turbidity through the act of tearing carrion flesh. As shown in other aquatic environments, the absence of scavengers reduces turbidity (Santori et al. 2020).

In Table A8, we report results that are consistent with the above predictions, albeit, imprecisely estimated. BOD and COD values increase in the high vulture suitability district after the onset of diclofenac use in livestock. This effect is entirely driven by the census urban districts (columns 2, 3, 5, and 6), similar to how the decline in dissolved oxygen and increase in fecal coliform was as well (see Table 5). Turbidity declines in water bodies monitored in census urban districts (columns 8 and 9), which is consistent with previous findings on declines in scavenger populations.

Table A8.
District Water Quality DD & DDD Estimates

	Biological Oxygen Demand			Chemical Oxygen Demand			Turbidity		
	U&R		U	U&R		U	U&R		U
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)
HVS×Urban×Post-1994		1.5			11.4			-6.4	
		(0.7)			(3.1)			(4.9)	
HVS×Post-1994	0.7	0.2	1.8	1.8	-2.2	9.6	-0.6	1.3	-5.5
	(0.4)	(0.4)	(0.7)	(1.7)	(1.8)	(2.8)	(3.5)	(4.2)	(4.6)
Urban×Post-1994		-0.6			-6.9			-0.2	
		(0.5)			(2.5)			(3.4)	
$\overline{Y}_{\leq 1993}$	4.01	4.01	5.03	25.32	25.32	28.61	36.44	36.44	40.30
R^2	0.74	0.74	0.75	0.71	0.71	0.75	0.79	0.79	0.78
N	4,337	4,337	2,062	4,144	4,144	1,967	3,600	3,600	1,671
Clusters	221	221	140	217	217	135	208	208	129

Notes: Estimation results for DD and DDD specifications. Each regression includes district-by-area-by-type fixed effects where area is either urban or rural, and type is the water body type (well, river, or lake). In addition, each regression includes year fixed effects. Sample consists of district-level data for census-urban (U) and census-rural (R) areas, from 1988 to 2004. Observations are population-weighted. We report Conley standard errors that are serially correlated at the district level, and are allowed to be spatially correlated up to 200km.

A.14.1 Additional Analysis for Fecal Coliform

In the main text, we report a large increase in fecal coliform in urban areas that were highly suitable for vultures, following the vulture population collapse—more than doubling the concentration, allowing us to reject an increase below 64%. At first glance, this might appear to be an implausible effect magnitude. Here, we provide a more descriptive analysis to help contextualize this specific result. First, in Figure A10, we plot the distribution of the logged values of fecal coliform in our sample, a comparison of the logged fecal coliform values in India relative to states in the United States in 2004, as well as the year-on-year and post- versus pre-collapse changes.

We proceed to estimate a quantile regression, for the deciles of the logged values of fecal coliform, and examine whether the estimated DD treatment effect meaningfully varies across the distribution. Because the results in the main analysis are noisy, our focus here is more descriptive—do we observe any evidence of a gradient across the distribution? In Figure A11, we observe an effect size close to zero through almost the entire distribution when using the census urban & census rural sample. However, in the census urban sample, we see suggestive evidence for higher increases in logged values of fecal coliform at higher decile values of the distribution. The analysis in Figures A11 along with the summary statistics in Figure A10 provide three important insights that are helpful when interpreting the results in Table 5: (i) fecal coliform levels in India are high, much higher than potential priors based on high-income countries; (ii) there is support in the data for large fluctuations, consistent with magnitudes we report in the DD estimation; and (iii) the reported effect on fecal coliform is predominately driven by census urban areas, and mostly by areas that experience high levels of fecal coliform pollution.

A.15 Evaluating Changes to Healthcare Access

Changes to healthcare access and utilization could also explain changes in mortality. This presents a threat to our identification strategy if healthcare access and utilization changed differentially between the high- and low-suitability districts after 1994. In Table 1, we document that the two groups of districts show no systematic difference in the number of hospitals and healthcare centers, or in the number of doctors and healthcare workers in 1991.

Here we use data from the 2001 and 2011 censuses to test whether those healthcare access metrics changed after 1994 in the high- relative to low-vulture-suitability districts. In Table A9, we report estimates that show no difference between the two groups of districts. This finding holds when we use the same set of districts as in the main analysis, or if we use the full set of districts that appear in the census. This result alleviates concerns that our main finding is capturing changes to the healthcare infrastructure that are somehow correlated with the location and timing of the vulture collapse.

A.16 Evaluating Changes to District Characteristics

We expand on the previous analysis of healthcare access and add several other placebo outcomes that should not be affected by the collapse in vulture populations. For each

Table A9.
Estimation Results for Healthcare Access

	Main Sample		Census Sample	
	Per-Capita Hospitals & Health Centers	Per-Capita Doctors & Health Workers	Per-Capita Hospitals & Health Centers	Per-Capita Doctors & Health Workers
	(1)	(2)	(3)	(4)
Residuals	0.12	1.52	0.14	3.59
	(0.28)	(2.52)	(0.27)	(5.05)
$\overline{Y}$	1.80	17.87	1.78	21.29
R^2	0.769	0.721	0.678	0.584
N	449	449	972	972
Clusters	153	153	324	324

Notes: Estimation results for the specification in Equation (2). The sample uses data from the Indian census on the number of hospitals, health centers, doctors, and health workers in 1991, 2001, and 2011, and converts them to per capita rates. The results in columns 1 and 2 are for the districts that have fully balanced death rate data and are used in the main analysis. The results in columns 3 and 4 are for all the balanced districts in the census data. Each regression includes district and zonal council-by-year fixed effects. Observations are population-weighted. The reported mean for the outcome is the population-weighted mean. We report Conley standard errors that are serially correlated at the district level, and are allowed to be spatially correlated up to 200km.

outcome, we have at least one year of data before, and one year of data after the collapse. We summarize the results in Figure A12, where we do not find that alternative explanations in the form of diverging employment or district infrastructure are consistent with the data. The overall differences are often very small relative to the mean of each outcome, and even when they are precisely estimated they move in the direction that would suggest improving health conditions in the treatment group.

A.17 Sensitivity Analysis Using Jackknifing

Because we use population weights in the analysis, it is possible that one very large district (in terms of population) had an increase in mortality or in reporting of vital statistics that happened around the same time as the vulture die-offs. If such a district exists, then it will receive a high weight in the regression, distorting the actual effect, and leading us to incorrectly interpret a spurious effect as a causal one. In order to rule out that our results are driven by an extreme outlier, we repeat the main estimation leaving one district out of the sample each time. The resulting distribution of coefficients in Figure A13 is narrowly centered around the estimate we recover using the full sample. The results from the jackknife procedure allow us to reject that a single district is driving the estimation.

We also conduct the leave-one-out exercise by excluding one state at a time. This allows us to evaluate whether any potential changes in the reporting of vital statistics might be driving the estimated effect in a manner that is not already captured by the inclusion of state-level trends in Figure A7. We recover a narrow distribution of the coefficients with mostly overlapping 95% confidence intervals.

A.18 Permutation Inference Analysis

As an additional robustness test, we also run a permutation inference analysis. Using permutation inference analysis allows us to rule out that our research design is failing to capture any cross-sectional or temporal features that are responsible for the observed effect.

We randomly re-assign the treatment across the districts and re-estimate the effect using the specification in Equation (2), repeating the process 1,000 times. We either fully randomize the treatment dummy across districts and years (full), maintain the same temporal structure but randomly assign districts as either treated after 1994 or not (block), or randomly assign the years that are flagged as treated within the districts that are truly part of the treatment group (within). We plot the permutation distributions in Figure A15, where each one of the distributions is centered around zero. More importantly, the estimated effect from the non-permutation sample is in the far right tail of each distribution, resulting in an exact p-value well below 1%.

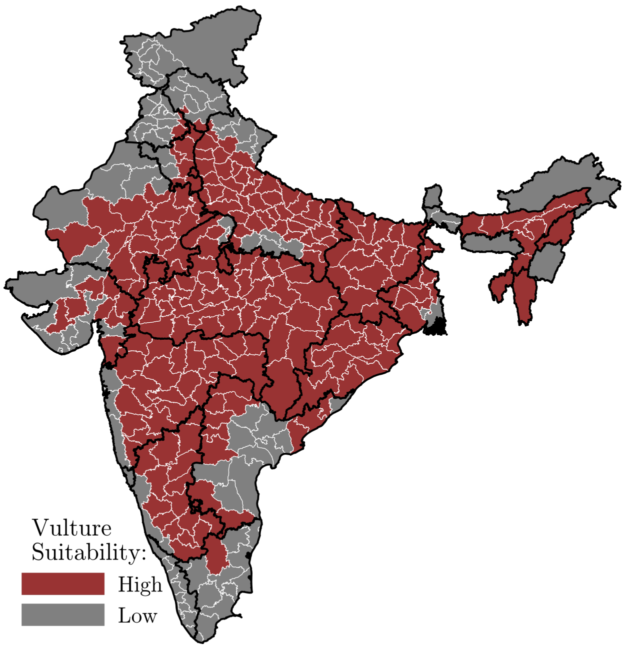
A.19 Robustness to Excluding Coastal Districts

Treatment districts using the habitat classification are mostly in the interior of the country. We might be concerned that including coastal districts might bias our results if death rates were differentially evolving after 1994. To check this, we classify each district using a coastal dummy variable. We summarise the population-weighted mean all-cause death rates for the low-suitability districts separately for coastal and interior districts. Then, we re-estimate the event-study specification by excluding those districts from the sample. We report our results in Figure A16 using versions of Figure 3C and Figure 4 from the main text. We find that qualitative patterns remain unchanged. It even appears that death rates in the coastal districts were slightly increasing over time, which attenuated our estimates in Figure 4, as seen in the variation of that analysis, which excludes coastal districts.

Figure A9: Classifying Treated Districts Using the BIOCLIM Habitat Suitability Model

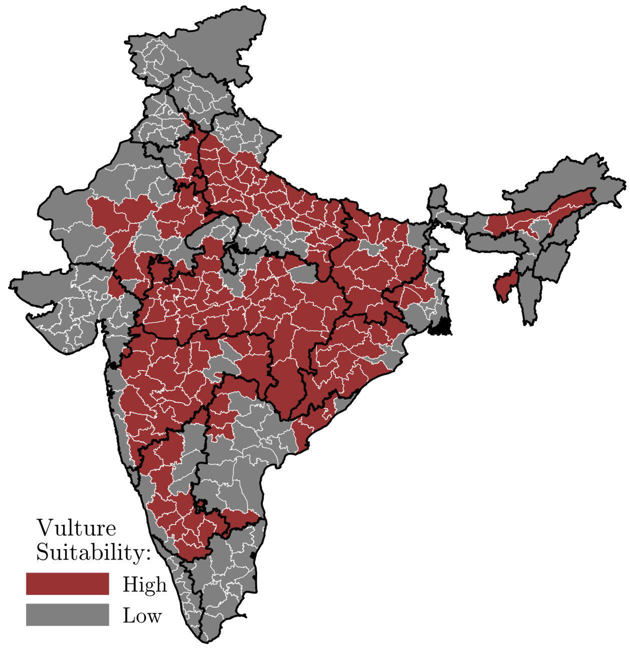
Suitability Score Terciles III & II

(a)

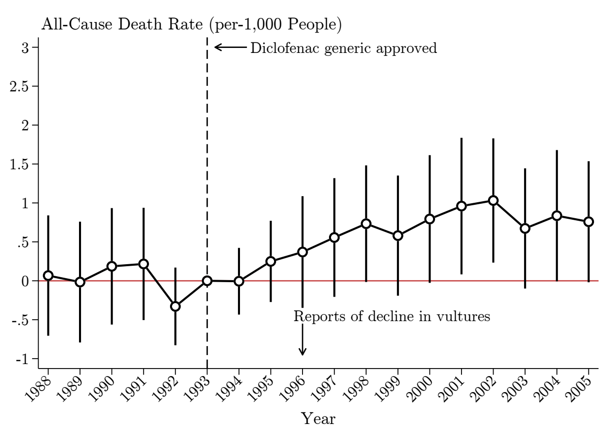


Suitability Score Above Median

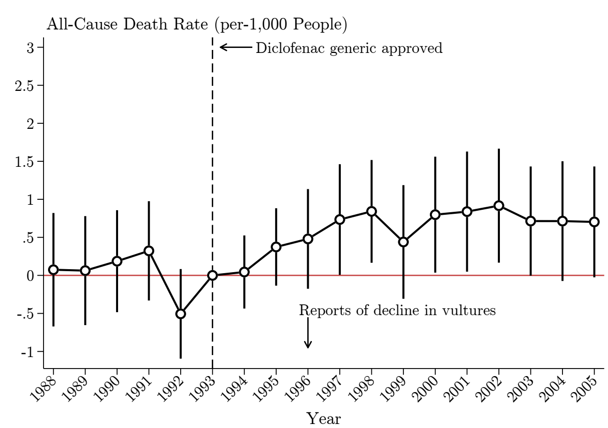
(b)



(c)



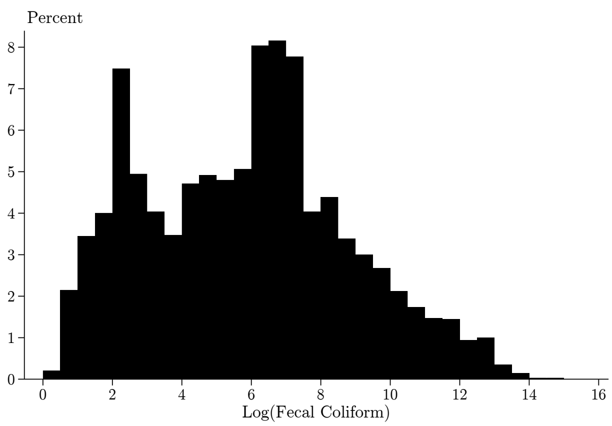
(d)



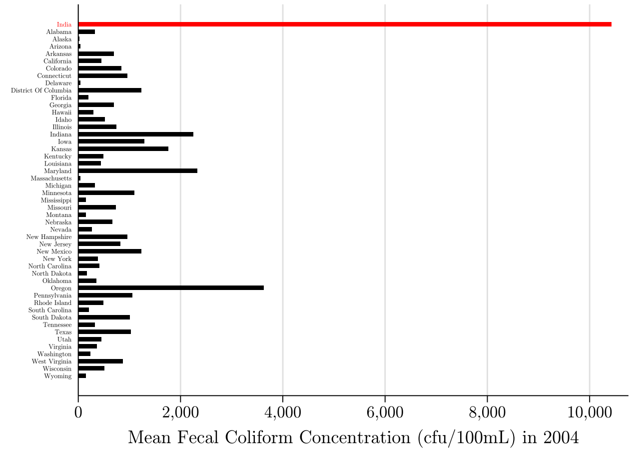
Notes: The treatment classification uses predicted suitability scores for the diclofenac-affected vultures from the BIOCLIM habitat suitability model. We either split the suitability score into terciles and define treated districts as the third and second terciles (a and c), or split districts as above or below the median suitability score, and define treated districts as those above the median (b and d).

Figure A10: Summary Statistics Regarding Fecal Coliform Pollution in India

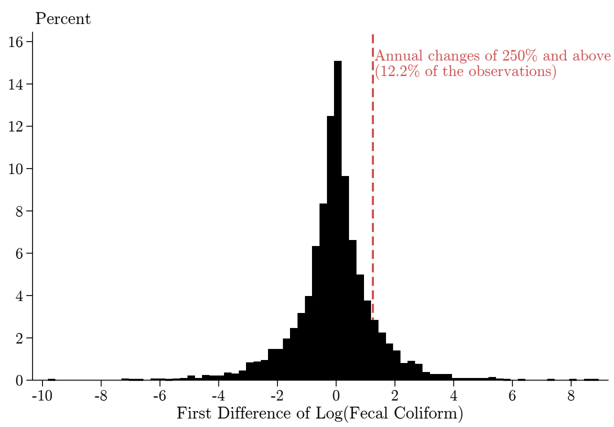
(a) Distribution of Log(Fecal Coli.)



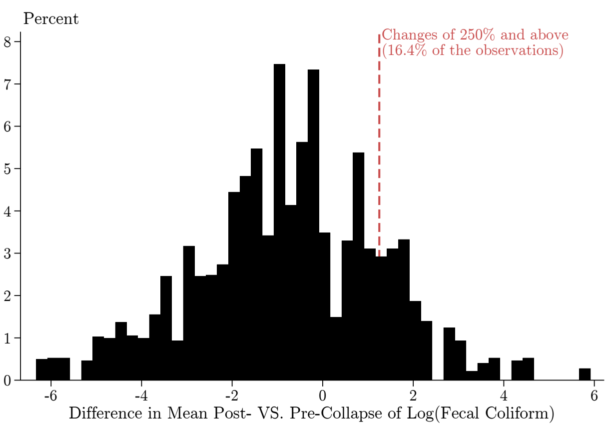
(b) 2004 Comparison India VS. Unite States



(c) Year-On-Year Changes



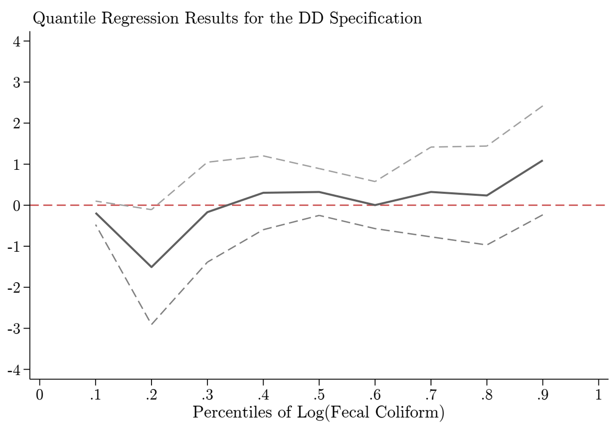
(d) Changes Post- VS. Pre-Collapse



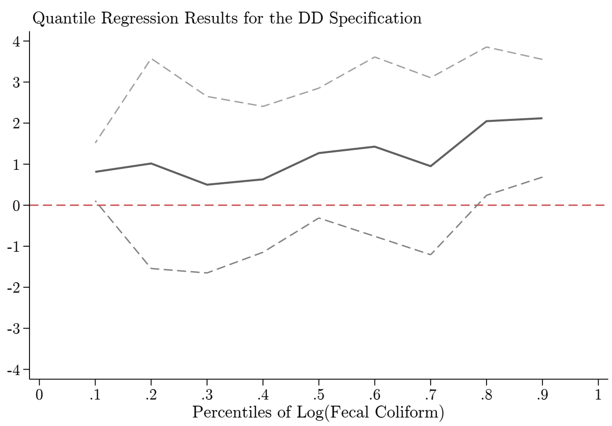
Notes: Summary statistics on fecal coliform data in the sample (panels a, c, and d), as well as a comparison of mean fecal coliform concentration between India and the United States in 2004 (the last year of our water quality data).

Figure A11: Quantile Regression Results for Log(Fecal Coli.)

(a) census urban & census rural Areas

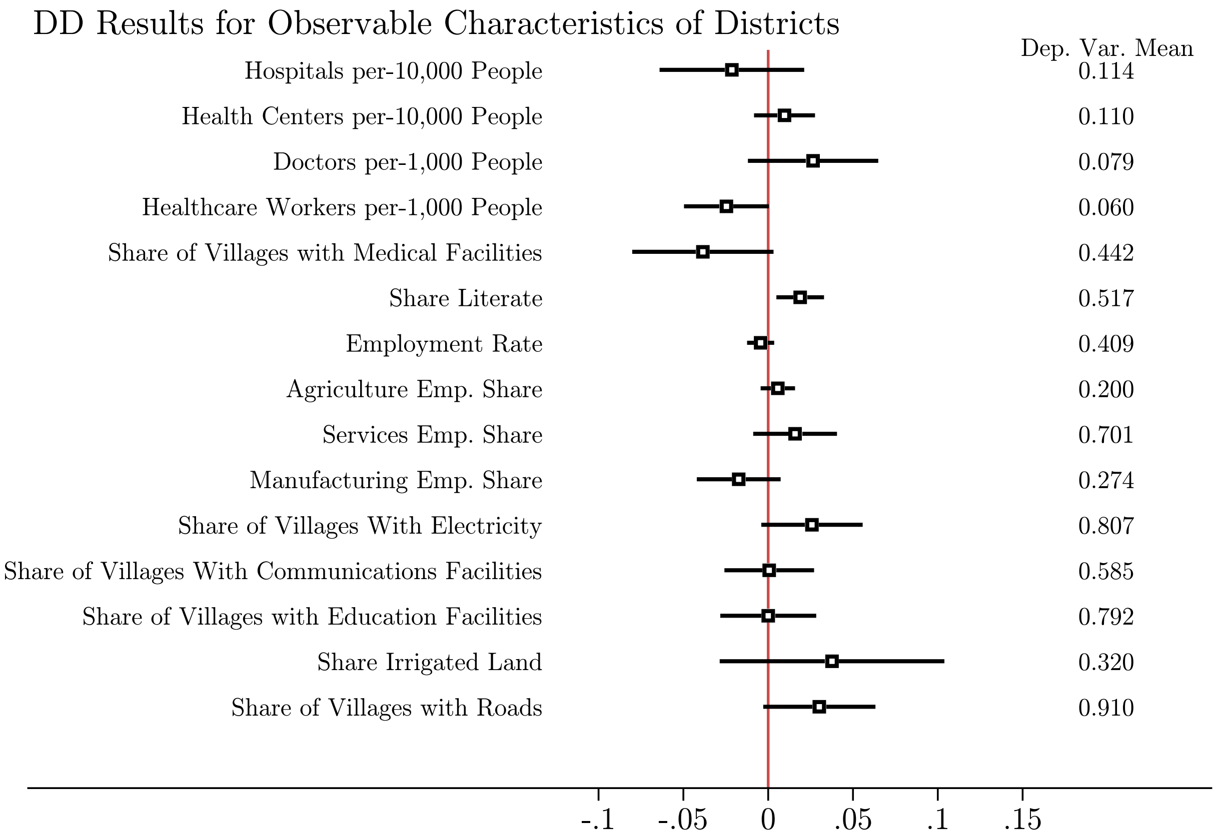


(b) census urban Areas Only



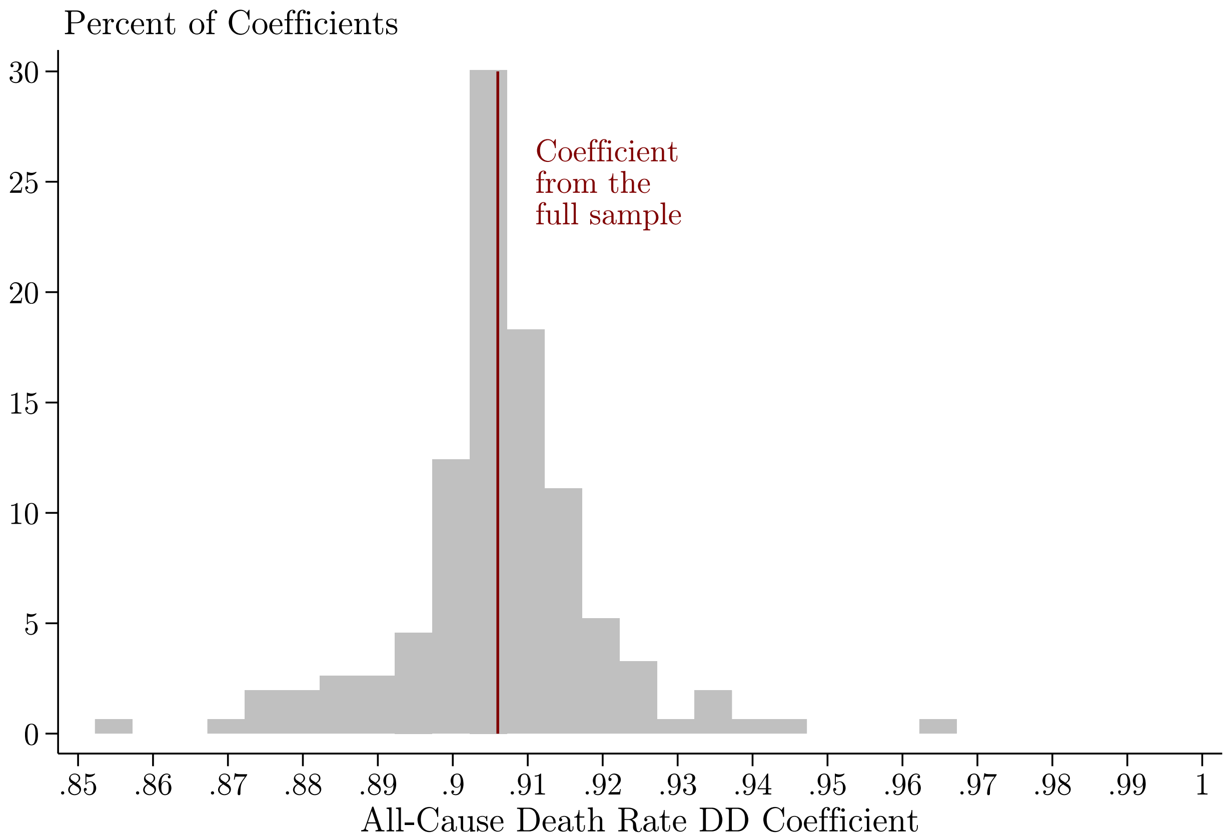
Notes: Quantile regressions, estimated at decile values of the logged fecal coliform distribution. 95% CIs are calculated using robust standard errors.

Figure A12: Summary of Placebo Results



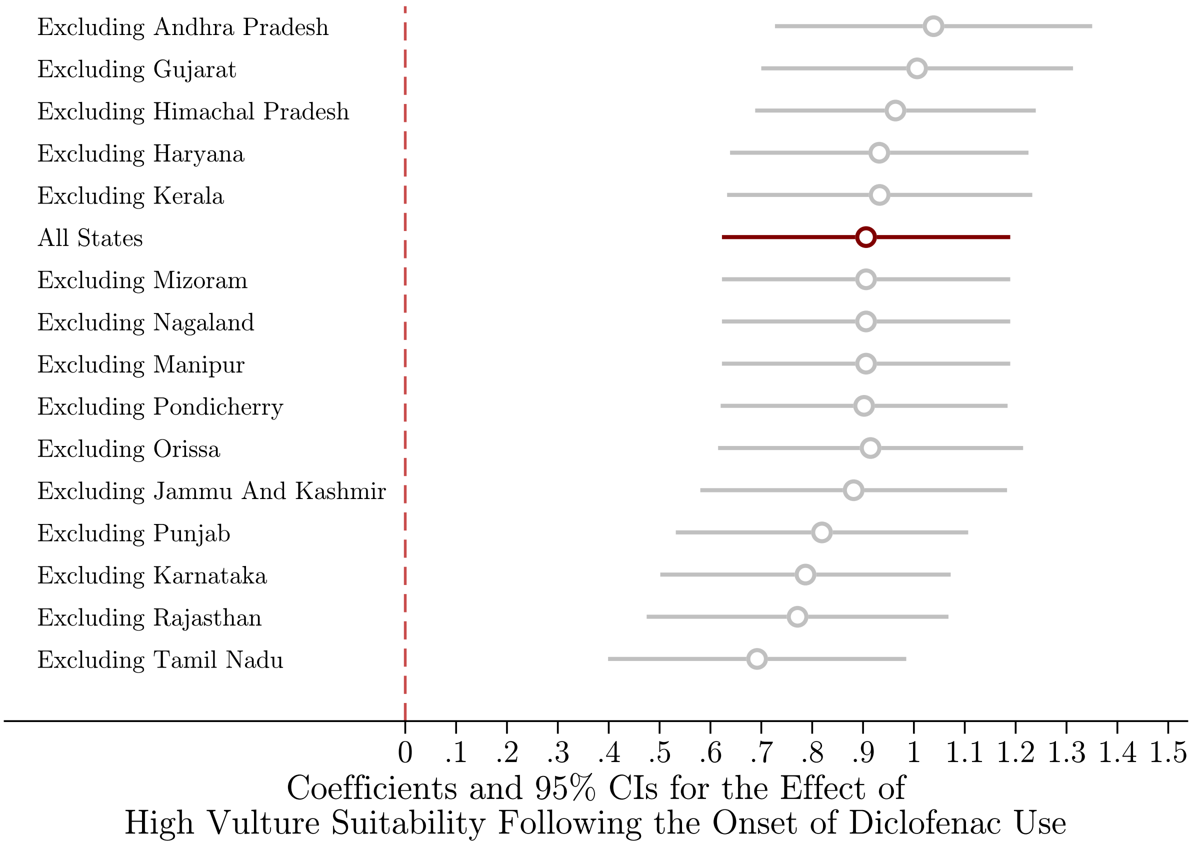
Notes: Estimation results for the specification in Equation (2). Each regression includes district and zonal council-by-year fixed effects. The sample includes all the districts in the balanced sample reported in the main analysis. Observations are population-weighted. We report 95% CIs that are calculated using Conley standard errors, which are serially correlated at the district level, and are allowed to be spatially correlated up to 200km.

Figure A13: Distribution of Leave-One-District Out DD Estimation Results



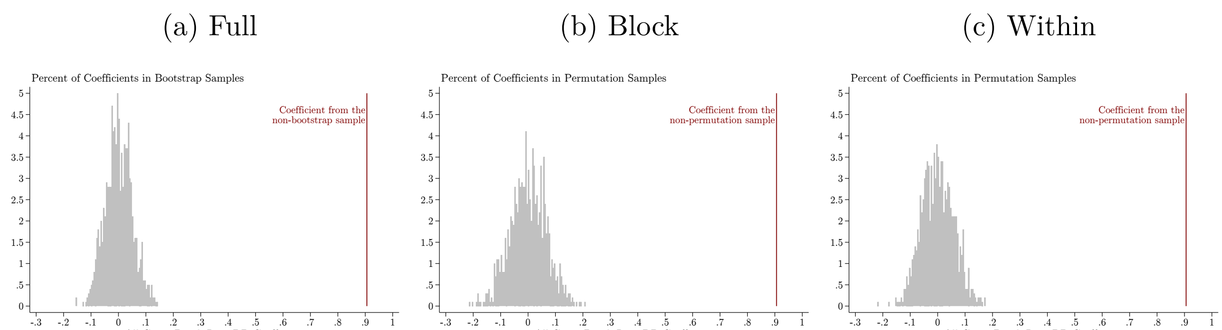
Notes: The distribution of coefficients from repeating the estimation in Equation (2) when leaving one district out each time. The vertical line shows the coefficient from the full balanced sample.

Figure A14: Distribution of Leave-One-State Out DD Estimation Results



Notes: The distribution of coefficients and 95% CIs from repeating the estimation in Equation (2) when leaving one state out each time. The maroon line shows the coefficient and 95% CI from the full balanced sample.

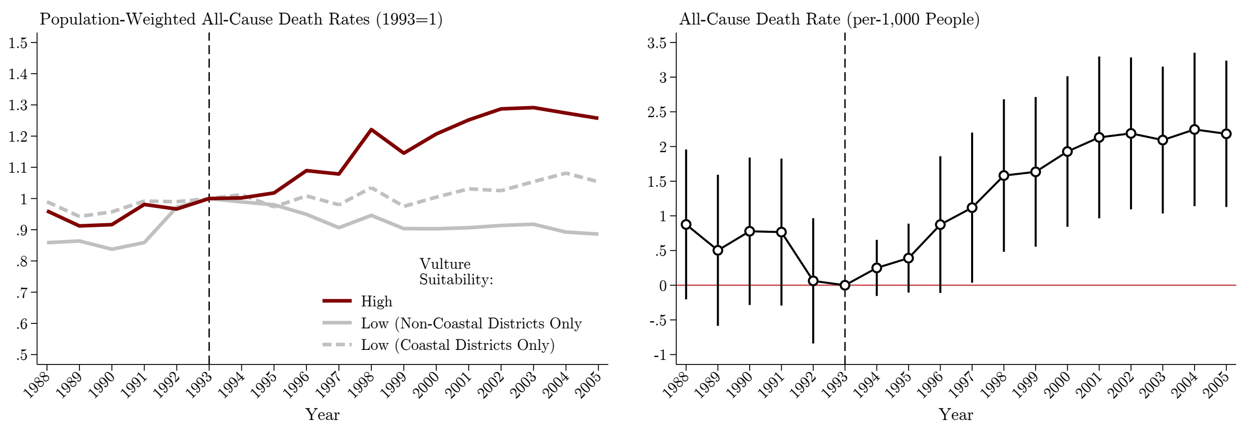
Figure A15: Permutation Inference DD Estimation Results



Notes: Distribution of coefficients from permutation samples where the treatment is randomly assigned. The vertical line shows the estimated coefficient from the non-permutation sample.

Figure A16: Robustness to Excluding Coastal Districts

(a) Highlighting Coastal Districts (b) Excluding Coastal Districts



Notes: Variations on Figure 3C and Figure 4 from the main text. In Panel (a), we plot the mean all-cause death rate for three groups: (i) non-coastal treatment districts, (ii) non-coastal control districts, and (iii) coastal districts. In Panel (b), we exclude coastal districts from the sample and repeat the main DD event-study estimation.

B Diclofenac Use Onset

In her book chapter discussing the decline of vultures in India, Subramanian (2015) writes that "Diclofenac had been restricted as the intellectual property of pharmaceutical titan Novartis, but when the patent expired around 1990, India's generic drug industry, coupled with a thriving black market, flooded the country with cheap highly potent diclofenac." (p. 178). To better establish the timeline of when diclofenac use became prevalent in the livestock sector in India, we looked for evidence on the exact timing of the expiration of the patent. In Figure B1, we include three annotated extracts from Federal Drug Administration (FDA) records and documentation. Combined, these show that there was a change in 1993 pertaining to the patent Novartis had regarding diclofenac, and that the code associated with that change is associated with approval for a generic version of the drug.

Recall surveys were conducted by Cuthbert et al. (2014) in 2004 with 29 veterinary clinics in India. Among the questions asked, veterinary professionals were asked about when they began offering certain non-steroidal, anti-inflammatory drugs to livestock farmers. The summary of the responses reported a median onset year for diclofenac of 1994.

Figure B1: FDA Documents Regarding Diclofenac & Generic Drug Approval

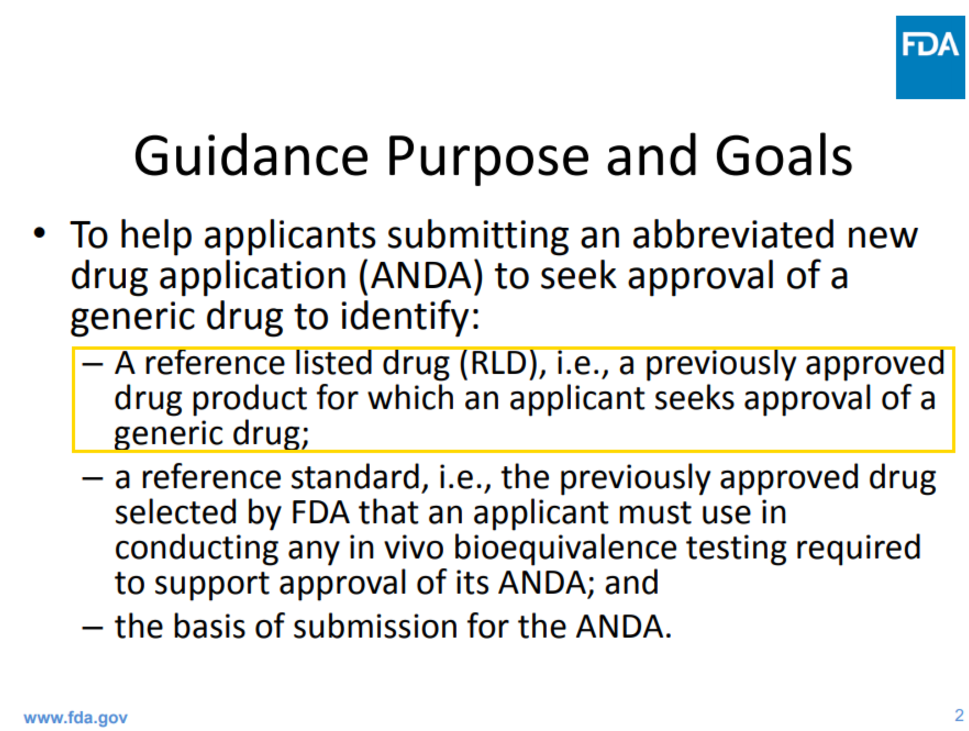
(a) Change to Novartis' Diclofenac Patent in 1993

		DICLOFENAC POTASSIUM					
		CAPSULE;ORAL					
		DICLOFENAC POTASSIUM					
	@	STRIDES PHARMA	25MG	A210078	001	Dec 03, 2019	Jun DISC
		TABLET;ORAL					
		CATAFLAM					
	+ @	NOVARTIS	50MG	N020142	002	Nov 24, 1993	Jan CRLD
		DICLOFENAC POTASSIUM					
AB		AMICI	50MG	A076561	001	Mar 18, 2004	Oct CAHN
AB		ANDA REPOSITORY	50MG	A076561	001	Mar 18, 2004	Sep CAHN
>D> AB	!	MYLAN	50MG	A075463	001	Jul 26, 1999	Nov CAHN
>A> AB	!	RK PHARMA	50MG	A075463	001	Jul 26, 1999	Nov CAHN
AB		RUBICON	50MG	A075229	001	Nov 20, 1998	Aug CAHN

(b) Change Code CRLD

CFTG	Change. A TE Code is added when a first time generic for an innovator is approved.
CMFD	Change. The product is moved from the Discontinued Section due to a change in marketing status.
CMS1	Change. Miscellaneous addition to list.
CMS2	Change. Miscellaneous deletion from list.
CPOT	Change. Potency amount/unit.
CRLD	Change. Reference Listed Drug
CHRS	Change. Reference Standard
CTEC	Change. Therapeutic Equivalence Code
CTNA	Change. Trade Name
DISC	Discontinued. The Rx or OTC listed product is not being marketed and will appear in the discontinued section in the next edition.

(c) Documentation Regarding RLD Changes



Source: Panels (a) and (b) were obtained from "APPROVED DRUG PRODUCTS WITH THERAPEUTIC EQUIVALENCE EVALUATIONS," 40th Edition. This document can be downloaded from: https://www.fda.gov/media/72973/download (Accessed on: 12/15/2020). Panel (c) was obtained from "Draft Guidance for Industry: Referencing Approved Drug Products in ANDA Submissions". This document can be downloaded from: https://www.fda.gov/media/102266/download (Accessed on: 12/15/2020).

C Data

C.1 BirdLife International Species Distribution Maps

We requested access to the geodatabase with all the digitized maps for all bird species maintained by BirdLife International (BLI). Access is provided for non-commercial uses.[34] The data include information about whether the species are extant or extinct, along with discrete categories regarding the likelihood of the two. The data also include information on whether the species is native or not, and whether their presence is yearly, during the breeding season, or other form of seasonality.[35]

We extract the maps for all vulture species in India. We consider the areas where they are labeled as extant, probably extant, possibly extant, and possibly extinct. We include ranges classified as possibly extinct as those still reflect potential presence in the past thirty years. For each district, we calculate the overlap of the habitat area, and repeat this for each species. This provides us with three overlap values for the three diclofenac-affected vulture species. We calculate the mean value of those overlap scores, and use those to assign the suitability category.

The data in the species distribution maps provided by BLI is the most complete source of information regarding the habitat areas of bird species around the world. BLI also assesses the conservation status and extinction risk as part of the Red List, produced by the International Union for Conservation of Nature.[36] BLI uses both published and unpublished sources of information to determine the boundaries of each range. Some unpublished sources of information include specific interviews with local experts, as well as confidential records.[37] The maps are known to err on the side of including areas that might not contain the species (Ramesh et al. 2017). This means that the true distribution of the species is a subset of the area in the distribution map. This could lead us to incorrectly consider districts as treated districts, when in fact they should be classified as control districts, resulting in attenuated estimates.

C.2 Examining the Reporting Accuracy of the CRS Data

One known limitation of CRS data in India is that many vital statistics events go unrecorded, and as a result, the CRS under-reports the true magnitude of mortality. Although there is no alternative to the CRS as far as district-level data is concerned, at the national level a commonly used source of information is the Sample Registration System, which samples less than one percent of the population, but is designed to recover a nationally representative sample (Rao and Gupta 2020).

We obtain the raw SRS records in order to compare the gap in reporting. While we do find that at the national level, the CRS underestimates mortality rates by about a factor of

[34] Application can be filled out at: http://datazone.birdlife.org/species/requestdis
[35] BLI provides a summary of these categories here: http://datazone.birdlife.org/species/spcdistPOS
[36] The Red List is a set of species assessments that classifies species as threatened or non-threatened with respect to extinction risk, across several sub-categories.
[37] Some records are considered confidential as their release might jeopardize the species if they are actively traded in domestic and international wildlife trade markets.

two relative to the SRS, when controlling for state and zonal council-by-year fixed effects, both sources of data allow us to recover similar trends in mortality rates. Specifically, we compare the CRS data to the SRS data in order to evaluate if underreporting of mortality in the CRS data is introducing bias in the trends in addition to underestimating the magnitude. The data in the SRS are reported at the state level. To compare the CRS and SRS, we take a population-weighted mean of the district- or state-level data, respectively, to obtain a national-level estimate for the all-cause death rate. We plot the levels of all-cause death rates, by source of data, by year, in Figure C1.

There is a clear difference in levels (Figure C1, dashed lines) between the all-cause death rate in the CRS relative to the SRS data. The SRS death rate is nearly double the CRS-reported death rate. However, when residualizing the death rates on a set of unit and time fixed effects (Figure C1, reported in the solid lines), the two death rates follow similar trends.[38]

We interpret the agreement between the residualized levels in Figure C1 as evidence that conditional on fixed effects, the CRS data manage to capture similar trends to those in the SRS data. In addition, the results from this comparison also highlight that the correct baseline level that we should use when comparing the relative change in mortality is nearly twice as large, reducing the relative size of the effect when using the CRS mean level by half.

The fixed effect specifications we describe in Section 4 compare changes over time and are robust to several forms of under-reporting. This allows us to recover the level differences in mortality. Interpreting our level estimates *relative* to a baseline level of mortality, using the mean mortality reported in the CRS data is undesirable because it would overestimate the size of relative changes. Consequently, in the interpretation of the analysis, we interpret the magnitude of the coefficients relative to the mean level from the SRS data, which reflects the national-level death rate (see Section C.5 below for details).

C.3 Stable District Boundaries

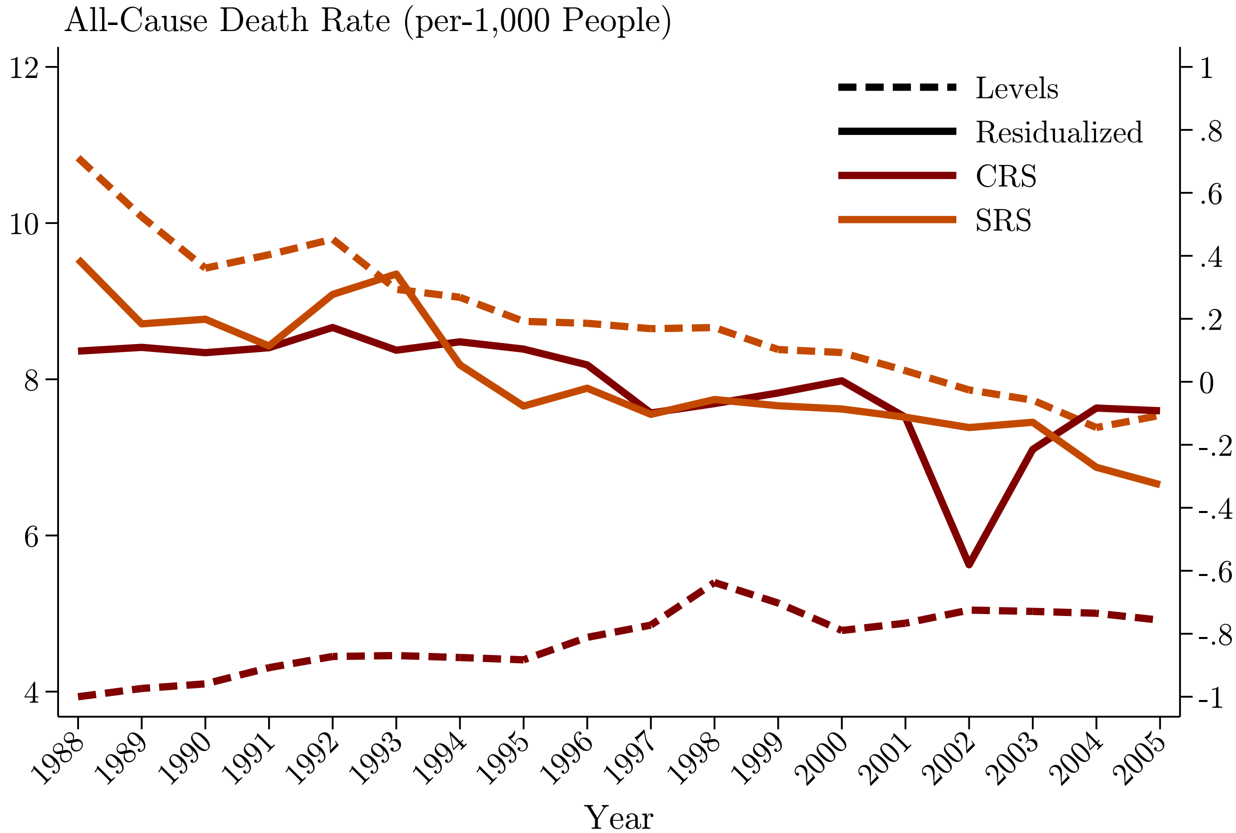
Historically, districts in India underwent considerable changes. Among these changes, some districts were split into new districts, while others had their borders re-drawn. This means that using the administrative definitions of districts, as is, will result in units entering and exiting the sample, and inconsistent geographic ranges over time. To overcome this, we stabilize districts relative to their 1981 borders. In the case that district split, we re-code them as their parent district. In the case where district borders change, we combine different districts as one unit. This builds on previous re-coding work performed in Greenstone and Hanna (2014) and Kumar and Somanathan (2009).

C.4 Changes to CRS Reporting in 1988

The VSI-CRS data experienced a shift in the reporting regime in 1988. From 1981 to 1987, the data are reported as rates, using interpolated population between censuses. From 1988 to 2005, the data are reported as counts. We use population data from the censuses to calculate

[38] Specifically, we include district-by-area or state-by-area, for urban and rural areas, fixed effects, as well as year fixed effects.

Figure C1: Comparing All-Cause Death Rates in CRS & SRS Data



Notes: Data from the CRS and SRS databases on all-cause death rates. District and state-level data are aggregated to the national level using population weights. Death rates are residualized (solid lines) on region (district or state), as well as zonal council-by-year fixed effects.

population growth rates, and use an exponential growth function to interpolate population during intercensal years. We then calculate all-cause death rates using the interpolated population data. In our main results, we use the data from 1988 to 2005 as the earlier data were calculated differently and are perhaps less comparable. In the Online Appendix, we provide the results for the full 1981 to 2005 period.

C.5 Representative National All-Cause Death Rates

As discussed above and in the main text, the CRS data allows us to recover the *differences* within districts over time even if the *levels* are under-reported. To correctly interpret the changes in mortality, we use the SRS sample to recover nationally representative means. We use a report that provides these values, annually, for the entire country, as well as the census urban and census rural areas. In Figure C2, we re-produce the key table in the report which summarizes those values. We use the values from 1988 to 1992 and calculate mean all-cause death rates of 10.2, 11, and 7.2, for all, census rural, and census urban samples, respectively.

Figure C2: Nationally Representative SRS Values

TABLE - 2

Trend in estimated vital rates - India, 1971-92

Year	Birth Rate			Death Rate			Infant Mortality Rate		
	Total	Rural	Urban	Total	Rural	Urban	Total	Rural	Urban
1971	36.9	38.9	30.1	14.9	16.4	9.7	129	138	82
1972	36.6	38.4	30.5	16.9	18.9	10.3	139	150	85
1973	34.6	35.9	28.9	15.5	17.0	9.6	134	143	89
1974	34.5	35.9	28.4	14.5	15.9	9.2	126	136	74
1975	35.2	36.7	28.5	15.9	17.3	10.2	140	151	84
1976	34.4	35.8	28.4	15.0	16.3	9.5	129	139	80
1977	33.0	34.3	27.8	14.7	16.0	9.4	130	140	81
1978	33.3	34.7	27.8	14.2	15.3	9.4	127	137	74
1979	33.7	35.1	27.6	13.0	14.1	8.1	120	130	72
1980	33.7	35.1	27.8	12.6	13.7	7.9	114	124	65
1981	33.9	35.6	27.0	12.5	13.7	7.8	110	119	62
1982	33.8	35.5	27.6	11.9	13.1	7.4	105	114	65
1983	33.7	35.3	28.3	11.9	13.1	7.9	105	114	66
1984	33.9	35.3	29.4	12.6	13.8	8.6	104	113	66
1985	32.9	34.3	28.1	11.8	13.0	7.8	97	107	59
1986	32.6	34.2	27.1	11.1	12.2	7.6	96	105	62
1987	32.2	33.7	27.4	10.9	12.0	7.4	95	104	61
1988	31.5	33.1	26.3	11.0	12.0	7.7	94	102	62
1989	30.6	32.2	25.2	10.3	11.1	7.2	91	98	58
1990	30.2	31.7	24.7	9.7	10.5	6.8	80	86	50
1991@	29.5	30.9	24.3	9.8	10.6	7.1	80	87	53
1992@*	29.0	30.7	23.1	10.0	10.8	7.0	79	85	53

* Provisional
@ Excludes Jammu & Kashmir

Source: Sample Registration System

Notes: Reproduced from the report available on: https://unstats.un.org/unsd/demographic/meetings/wshops/1993_China_CRVS/docs/1993_Doc.26_India.pdf (Accessed August 22, 2023).

D Additional Details on the Empirical Setting

D.1 Additional Details on the Role of Vultures as Keystone Species

The ecological and epidemiological dynamics of scavengers, pathogens, and infectious diseases help explain the causal link between diminishing vulture populations and human health. While some animal species will feed on carrion if available, for vultures, it is the only source of food. As a result, vultures have evolved as very efficient scavengers. Vultures have an extremely acidic stomach, that ranges from just above zero to two pH.[39] In comparison, an average human has a pH level of two in their stomach, making it ten to a hundred times less acidic than that of a vulture. This is one of the key adaptations that allows vultures to safely consume carrion, and also results in most bacteria not surviving their digestive system (D. L. Ogada et al. 2012; Roggenbuck et al. 2014).

Vultures are extremely effective at reducing a carcass to its bones, and can consume the carrion of an entire cow within forty minutes (D. L. Ogada et al. 2012). Other scavenging species are not good substitutes from a sanitation point of view because they leave the flesh behind. Recent experimental evidence confirms that non-vulture species are not able to compensate and functionally replace vultures in terms of scavenging efficiency (Hill et al. 2018). For this reason, the historic presence of large and stable vulture populations simultaneously reduced pathogen and bacteria concentrations in the environment, as well as crowded out other mammalian scavengers such as dogs and rats that transmit various diseases including rabies (Moleón et al. 2014). In the absence of vultures, the composition of species that feed on carcasses changes towards dogs and rats.[40]

The removal of carrion from the environment by vultures becomes more important in low to middle-income countries where these birds have effectively substituted for expensive infrastructure to safely dispose of animal carcasses. The limited availability of infrastructure such as animal incinerators has led to so-called "animal landfills" on the outskirts of population centers across India. Anecdotal accounts describe how with vultures no longer available, the rotting meat and its scent build up, attracting feral dogs (Subramanian 2011). Attacks by dogs are common, and they mostly represent an immediate deadly threat to small children. However, with India being a global epicenter for rabies, any animal bite can result in death (Braczkowski et al. 2018). The combination of dogs and rats serving as vectors of infectious diseases and being far less efficient scavengers than vultures, make carcass dumps a breeding ground for disease (D. L. Ogada et al. 2012).

Livestock agriculture also becomes a source of water pollution when farmers need to dispose of dead animals (Engel et al. 2004; Kwon et al. 2017). A 2016 Supreme Court ruling in the state of Uttarkhand recognized that animal carcass dumping in water bodies is an ongoing problem, even in water bodies that are considered sacred: "It is tragic that the Ganga, which has since time immemorial, purified the people is being polluted by man in numerous ways, by dumping of garbage, throwing carcass of dead animals and discharge of

[39] Acidity is measured on a logarithmic scale. Water has a pH of seven, and lower values are considered more acidic. Acids that are dangerous to come in direct contact with have pH values of four and below.

[40] As Dr. Asad Rahmani, Director of the Bombay Natural History Society, put it: "Now there are dogs. They eat anything, live or dead. There are dogs on the ground but the skies are empty" (Subramanian 2011).

effluents" (Sharma and Singh 2016).

The presence of animal landfills near and in census urban centers has been documented extensively in academic writing and news articles. Senacha et al. (2008) study the concentration of diclofenac residue across different animal dumping sites, and classify many of the sites as residing in urban areas. In their review of the historic importance of vultures in India, Van Dooren (2010) writes that in "urban and semi-urban environments, they found abundant food in carcass dumps, as well as in tanneries, slaughter yards, garbage dumps, and bone mills." They further emphasize that "especially in urban environments, vultures have provided an incredibly valuable service to humans." Singh et al. (2013) begin their paper on the challenges that India is facing with veterinary urban hygiene by writing that: "India is confronted with many hygiene problems in urban areas that are related to animal populations." They continue to describe how dead animals in urban areas are "left to rot in the open" and that "diseases in urban parts of the country have shown a rapid growth, due to the co-habitation of their hosts in areas around animal populations where proper sanitation is not maintained."

Journalists writing on the collapse of vulture populations have also noted the urban landscape as the nexus of vultures and dead animals. McGrath (2007) writes that "In urban areas, haulers take dead animals to official dumps." Sanjayan (2013) writes that "Huge dumps have sprouted near urban centers where thousands of dead cows, along with the occasional horse or camel, are brought to rot." Pati (2016) writes about the civic authorities attempting "to stop the present unscientific disposal of animal carcasses near urban areas."

Finally, the interaction of widespread dairy cultivation with cultural practices regarding dead animals has resulted in a historically large reliance on scavengers in India. Restricting the amount of carrion and the time it remains in open fields is of specific importance in India because of prevailing social norms regarding the handling of meat. Hindus will not consume cows whereas Muslims will not consume animals not killed according to *halal*.

We summarize the interactions between vultures, mammalian scavengers, environmental quality, and public health in Figure 1. Within the ecosystem interaction group of vultures, mammalian scavengers (dogs and rats), and livestock carrion, the former two are competing for the food source (dead animals). Greater availability of dead carrion supports larger populations of both scavenger types, efficient (vultures), and inefficient (dogs and rats). Because both types compete for the same food source, each type indirectly limits the population growth of the other type. Following the decline in vulture populations, which we describe in detail in the next section, environmental quality declines due to the increase in the inefficient scavengers, which lead to more carrion rotting in the open, and the rise in vectors of infectious diseases. Combined, the decline in vultures leads to worse public health outcomes.

D.2 Additional Details on the Decline of Vultures in India

Vultures were once a ubiquitous sight across India with a population that may have exceeded 50 million birds. Today, the three species that made up the bulk of the population are considered critically endangered after declining by more than 95%.[41] Their collapse is

[41] The three common names (and scientific names) of the three affected vulture species are: slender-billed (*Gyps tenuerostris*), white-backed (*Gyps bengalensis*), and long-billed (*Gyps indicus*). There is one ad-

attributed to the chemical residue of the painkiller diclofenac in livestock animals, administered by farmers to treat fevers and inflammations. A vulture that feeds on a carcass with diclofenac residue can develop kidney failure within weeks and die.[42] In Figure 2, we plot the classification of districts according to their baseline habitat suitability for the affected vultures (we explain this classification in detail in Section 4.1).

Diclofenac is an old drug, first introduced in 1973 by Ciba-Geigy (now Novartis). It has since become the most widely used non-steroidal anti-inflammatory drug in the world and is prescribed as a painkiller for many conditions (Altman et al. 2015). However, its use as a veterinary drug to treat injuries, inflammations, and fevers in wounded or sick animals was a much more recent development (Cuthbert et al. 2014; Subramanian 2015). This became possible only when low-cost generic versions of the drug were developed around the time of expiry of the original patent.

Anecdotal accounts place the timing of the patent expiration in the early 1990s (Subramanian 2015). Sales data that we purchased from the company IQVIA shows a dramatic growth in the entry of Indian drug manufacturers around this time (see Figure 3a). In order to more precisely determine the onset of diclofenac use, we draw on additional sources of data. We start with formal records regarding the patent and its expiration. The patent originally belonged to the pharmaceutical company Novartis. Using documents from the Federal Drug Administration regarding drug patents, we are able to trace the first approval for a generic version granted to Novartis in 1993. See the Appendix for additional details. This is consistent with a survey of veterinary clinics conducted by Cuthbert et al. (2014) which indicates the first veterinary formulations in India became available in 1994. With these sources of information, we classify 1994 as the first year in which diclofenac was widely used to treat livestock, and assign this as the year of treatment onset.

Reports of vulture declines rapidly followed the veterinary use of diclofenac. Field observations in 1996 found only half of the 353 nesting vulture pairs recorded in 1984 in Keoladeo National Park outside Delhi. Surveys conducted in 1996 reported dead vultures around the nests, in bushes, and hanging from the trees. By 1999, there was not a single living vulture pair documented at the site (Subramanian 2011). After Dr. Vibhu Prakash, at the time a PI in the Bombay Natural History Society, communicated his findings to colleagues, they reported similar patterns they thought were simply idiosyncratic to their study sites. Population declines were so rapid that in 2000, all three species were classified as critically endangered.

At first, several conjectures were made regarding the potential cause. These included the emergence of a new wildlife disease or the effect of pesticide accumulation, as well as deliberate poisoning by Western countries (Subramanian 2015). It took about a decade to establish the root cause when Oaks et al. (2004) used both autopsy data, and experimental exposure of vultures to diclofenac, to show that even trace amounts of diclofenac in the carcasses that vultures feed on result in lethal kidney failure. As a result, the Indian government banned the veterinary use of diclofenac in 2006 (Vibhu Prakash et al. 2012; D. L.

ditional member of the gyps genus, Himalayan Griffon (*Gyps himalayensis*). However, as their name suggests, they are mostly found in the Himalayas, where they do not depend on livestock carcasses that have diclofenac residue that caused the collapse in the other species.

[42] We use the term kidney failure for clarity. The more medically correct terms are renal failure and visceral gout.

Ogada et al. 2012). However, surveys conducted up to 2018 document rampant illicit use of diclofenac in livestock including through the diversion of human doses (Galligan et al. 2020).

Thus despite the 2006 ban, vulture populations remain a miniscule fraction of what they once were. Recovery is difficult because vultures have a low fecundity. A female vulture will lay at most a single egg each year. Vultures take five years until they reach sexual maturity. Assuming they find a mating pair, construct a nest for six weeks, lay a single egg, and successfully feed and ensure the survival of the offspring for four months, a new vulture gets on the path toward reproducing in about five years (D. L. Ogada et al. 2012).

In the absence of vultures, livestock farmers, and municipalities can utilize either labor-intensive or capital-intensive substitutions. Farmers can exercise deep burial but given the number of livestock animals, this adds high labor costs. Since these costs are private while the costs of disposing of animals in carcass dumps or water are socialized, it is not surprising that deep burial remains uncommon. Livestock carrion can be disposed of using specially designed incinerators, yet they are expensive to buy and operate and require a reliable mechanism for making sure that farmers transport dead animals to them. According to a 2020 report by India's Central Pollution Control Board, India has yet to adopt livestock incinerators as a substitution for vultures: "Very few cities have carcass utilization plants and incinerators. One such carcass utilization plant is installed in Delhi and incinerator is under installation in Chandigarh" (Central Pollution Control Board 2020).

D.3 Tabulation of mortality effects of other WASH interventions

We tabulate the mortality effects of other water and sanitation interventions from the literature to aid benchmarking of the effect of losing an ecosystem service providing the safe and quick disposal of meat from livestock carcasses.

Table D1: Tabulation of mortality effects of other water, sanitation, and air-pollution interventions reported in the literature.

Reference	Country	Year	Intervention	Outcome	Effect Size (Reductions)
Geruso & Spears, 2018	India	1992-2006	10% Reduction in Open Defecation	Infant Mortality	8%
Bhalotra et al. , 2021	Mexico	1991-95	Municipal water chlorination	Under 5 Mortality (Diarrhoea)	45%-67% (diarrhoea) $\sim$ 9.4-13.4% (all-cause)
Watson, 2005	USA (Native-American Reservations)	1968-1998	10% increase in homes receiving sanitation improvements	Infant Mortality	2.5%
Cutler and Miller, 2005	USA	1900-1940	Urban clean-water infrastructure (filtration)	Total & Infant Mortality	16% (TMR), 43% (IMR)
Marcella and Goldin, 2019	USA	1880-1920	Urban sewerage and safe-water treatments	Child & Infant Mortality	33.6% (U5MR), 48% (IMR)
Galiani et al 2005	Argentina	1995-1999	Privatization of municipal water companies	Child Mortality	8%
Tanaka, 2015	China	1998-2000	Air Pollution Regulations	Infant Mortality	20%
Ebenstein et. al. 2017	China	2004-2012	Lifetime free coal-based home heating	Life Expectancy,All-Cause Mortality	-20% to -26% (all-cause), -1.2 years (life expectancy)

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