THE SOCIAL COSTS OF KEYSTONE SPECIES COLLAPSE: EVIDENCE FROM THE DECLINE OF VULTURES IN INDIA*

Eyal G. Frank†

Anant Sudarshan‡

March 10, 2022

Abstract

The loss of a keystone species can theoretically lead to large social costs because their complex ecosystem interactions may be important for environmental quality. We quantify these effects for the case of vultures in India where they play an important role in removing livestock carrion from the environment. The expiration of a patent for a common chemical painkiller led to its increased use in cattle, unexpectedly rendering carcasses fatal to vultures, leading to a catastrophic and near total population collapse. Using habitat range maps for the affected species, we compare high to low vulture suitability districts before and after the patent for the painkiller expired. On average, all-cause death rates increased by 9.2% in vulture-suitable districts after the vultures nearly went extinct. We find suggestive evidence that feral dog populations and rabies increased, and that water quality deteriorated in the affected regions. These mechanisms are consistent with the loss of the scavenging function of the vultures. Quantifying the costs of biodiversity losses has critical implications for optimal investments into species conservation and rehabilitation.

*We thank Rema Hanna, Michael Greenstone, and Amir Jina for providing access to data. We are grateful for the Tata Centre for Development and the Becker Friedman Institute at the University of Chicago for providing funding in support of the work. We thank the seminar and conference participants at the Indian Statistical Institute, BREN School at UC Santa Barbara, the LSE Workshop in Environmental Economics, and the Department of Zoology at the University of Cambridge for their helpful comments. We thank Sushant Banjara, Alina Gafanova, Animesh Jayant, Simran Karla, Sahila Kudalkar, and Yuerong Zhang for excellent research assistance. All remaining errors are our own.

†Harris School of Public Policy, University of Chicago. Keller Center, 1307 E 60th St. Chicago, IL 60637. Email: eyalfrank@uchicago.edu

‡Kenneth C. Griffin Department of Economics, University of Chicago. Saieh Hall, 1160 E 58th St. Chicago, IL 60637. Email: anants@uchicago.edu

1 Introduction

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

The extinction of species in the wild poses a vexing challenge to the economic ideal of efficiently exploiting natural resources (Dasgupta and Heal 1974; Heal 2000). Although biodiversity loss is argued to be damaging in general (Cardinale et al. 2012a), evidence regarding the effects of specific species declines is sparse, as it is hard to quantify (Frank and Schlenker 2016; Ferraro et al. 2019). Wild population levels can collapse unexpectedly, making it hard to prevent, and even harder to reverse the change. Economic theory has long recognized the complexities involved in carrying out a forward looking cost-benefit analysis in the presence of irreversibilities and uncertainty (Arrow and Fisher 1974; Weitzman 2009). In the case of natural inputs, losses in biological abundance and diversity contrast the costs of averting them with the damages that they can cause (Naeem et al. 2009; Cardinale et al. 2012b; David U Hooper et al. 2012), yet empirical evidence is needed to quantify these magnitudes (Weitzman 1992; Polasky and Solow 1995; Weitzman 1998; Brock and Xepapadeas 2003). Incomplete information regarding the willingness-to-pay for ecosystem functioning presents a challenge in answering whether losses of species in the wild is inefficiently high.

In recent years, there have been several attempts to reintroduce or stabilize wild species in order to sustain their ecosystem functions (Seddon et al. 2007). Known examples are preserving bees for pollination (Winfree et al. 2007), bats for insect suppression (Frank 2021), and wolves for reducing deer densities (Raynor et al. 2021). Determining the optimal rate of reintroduction or restoration of wild species presents a challenging problem for policymakers to solve. Once a population has been depleted, whether it is worthwhile to restore depends on the benefits that are lost as long the species is absent in the wild, any defensive expenditures in response to their decline, and on the costs of recovery. Consequently, policymakers face an allocation problem of balancing species preservation with mitigating the effect of their decline (Metrick and Weitzman 1998; Tschirhart 2009).

We study this question in the context of the catastrophic collapse of vulture populations in the Indian subcontinent. The decline of vultures in India is important not just because of the scale of the collapse, but also because of the role they have in the provision of sanitation and public health. Vultures are efficient scavengers and farmers rely on them to quickly remove livestock carcasses (D. L. Ogada et al. 2012). Across the Indian subcontinent, vulture populations fell sharply after farmers began administering their livestock with a painkiller called diclofenac, made widely available after its patent expired in 1993 (Oaks et al. 2004; Cuthbert et al. 2014; Subramanian 2015). The cause of death was as deadly as it was initially mysterious, and it was only in 2004 that research showed that certain vulture species would develop kidney failure and die within weeks after digesting a carrion with even small residue of diclofenac (Oaks et al. 2004).

In this paper, we show that the collapse in vulture populations has led to a significant increase in human mortality. Following the onset of large-scale diclofenac use in 1994, districts that were highly-suitable for diclofenac-affected vulture species saw an average increase of 9.2% in all-cause death rates, relative to low-suitability districts. Urban areas, with a large supply of animal carcasses on their outskirts, and higher population densities, saw a higher adverse effect relative to rural areas. In conjunction with the higher mortality rates, we also document suggestive evidence for an increase in dog populations in the high-vulture-suitability districts, and an increase in the use of rabies vaccines following the collapse of vulture populations. This is consistent with anecdotal evidence regarding large increases in animal bites and mortality from rabies (Subramanian 2015), which India suffers a high-prevalence of (Braczkowski et al. 2018). To further establish the mechanisms that connect vultures and public health, we provide evidence that water quality deteriorated in the high-vulture-suitability districts, consistent with greater availability of rotting carrion in the environment.

The setting of the vultures in India provides a vivid example of biodiversity loss as an unforeseen externality. Once too numerous to count, with a population in the tens of millions, the number of vultures in the wild fell by over 95% within a few short years in the second half of the 1990s. Today just a few thousand vultures survive in India, with the three primary species all critically endangered. 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 infectious diseases (Moleón et al. 2014).

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.[1] 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).

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. In India, the interaction of widespread dairy cultivation with cultural practices regarding dead animals has resulted in a historically large reliance on scavengers. The sanitizing function vultures perform prevents livestock carcasses from rotting in open fields, and from transmitting their pathogens and diseases, such as anthrax, to other scavengers. Deadly pathogens can also get eroded by surface runoff and end up in the drinking water supply (Vijaikumar et al. 2002; R. T. Watson et al. 2004; Markandya et al. 2008; D. Ogada et al. 2016). Restricting the amount of carrion and 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

[1] 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.

consume animals not killed according to *halal* (Subramanian 2011).

While other scavenging species exist, they are imperfect substitutes for vultures. 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). Non-vulture scavengers are just not as effective, as they tend to leave some of the flesh behind (Subramanian 2011). 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). In the absence of vultures, the composition of species that feed on carcasses changes towards species that are a reservoir of pathogens and diseases to which humans are susceptible to.

The limited availability of infrastructure such as animal incinerators have 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 builds up, attracting feral dogs (Subramanian 2011). Dogs and rats are vectors of infectious diseases and much less efficient scavengers than vultures, a combination that makes carcass dumps a breeding ground for disease (D. L. Ogada et al. 2012).

Notwithstanding the existence of these mechanisms, it is empirically challenging to quantify the value vultures could provide if restored. This is of fundamental importance in determining the resources that should be allocated to preservation or recovery. The circumstances of their rapid near-extinction in India create a unique natural experiment that allows us to quantify the public health benefits of the vulture population, identified off a large and non-marginal change in their population. Specifically, the expiry of the diclofenac patent and the consequent flooding of the Indian market with low-cost generic versions of the drug, created a plausibly exogenous shock that differentially affected regions of the country, based on their ex-ante population of diclofenac-susceptible vulture species.

Because vultures were so abundant across India it was never deemed necessary to survey them and maintain a population count database. In the absence of data on the population densities, we leverage habitat suitability to infer treatment intensity. We classify districts into vulture suitability terciles using well-established species range distribution maps produced by BirdLife International (BLI), considered the global authority on birds and their habitats. Explicitly, we overlap each district with the habitat range maps for each species, calculate the mean area overlap for the three diclofenac-affected vultures, and divide into terciles. This is similar to how Alsan (2015) uses suitability for the tsetse fly in Africa, but our approach does not rely on strict functional form assumptions.

Using a difference-in-differences strategy, we compare high to low-suitability districts, before and after the onset of diclofenac use. In our analysis, we focus on district level data on death rates from all causes. Our main finding is an increase of 0.85 deaths per-1,000 people, reflecting an increase of 9.2%, relative to a nationally representative mean. While a 9.2% increase in mortality is a large and meaningful increase, we find that it is on par with a four to six additional hot days above 34°C. Despite this large divergence in death rates between the high and low vulture suitability districts, we find that they are mostly balanced across observable characteristics in the years prior to the collapse of vulture populations. One notable exception is that the low-vulture-suitability districts had a higher death rate, suggesting that in the absence of large vulture populations those districts were exposed to a higher infectious disease burden.

To further investigate the mechanisms through which a decline in vultures can affect public health, we study the channel of dog population densities. Attacks by feral 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 (Subramanian 2011). Anecdotal accounts often describe feral dog populations increasing by a third after competition over food sources with vultures subsided.[2] As part of the livestock census, India records counts of cattle, sheep, and other domesticated farm animals, as well as counts of dogs. Starting with the 2012 livestock census, the data also include counts of feral dogs. We use the cross-sectional variation in the feral dog counts and compare it to the mean

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

habitat overlap with the diclofenac-affected-vulture speices. We find that dog population are much higher in 2012 in the high-vulture-suitability districts. In addition, using data on rabies vaccines, we document a sharp increase in their use in 1997, which agrees with the timing of a sharp increase in diclofenac sales, a decline in vulture observations in the wild, and the timing of the increase in all-cause death rates.

As previous work has demonstrated, livestock agriculture acts as a source of water pollution, especially when needing to dispose of dead animals (Engel et al. 2004; Kwon et al. 2017). We use data on annual measurements of water quality and find evidence of degraded water quality in the most affected regions in our sample, across a variety of outcomes that have been linked to animal agriculture, and the absence of scavengers (Swift et al. 1979; Santori et al. 2020; Brundage 2021). We find more than a doubling of fecal coliforms in water bodies around urban areas, indicating potential leeching of additional pathogens and bacteria into drinking water.

The analysis in this paper relies on the collapse in vultures to be unforeseen, and not a direct consequence of our outcomes of interest. These assumptions are supported by the fact that the use of diclofenac expanded after its patent expired in 1993, and the connection to the demise of vulture species was only made in 2004. A remaining potential threat to our identification strategy is differential reporting of mortality that is systematically correlated with the timing and location of the decline in vultures.

To reject that the reporting of deaths is increasing more in the high-suitability regions, we verify the findings are robust to the inclusion of state-linear time trends, as well as state-by-year fixed effects. We further leverage baseline variation in livestock farming as part of a triple-differences design to verify it is the combination of high-suitability with high-density of livestock that leads to an increase in mortality following the collapse in vulture populations. As differential pre-trends could also mean that our key identifying assumption is violated, we include results from a longer panel with more time periods, yet fewer balanced districts. Even in the extended sample, we do not observe systematically different pre-trends in death rates across the different vulture suitability categories. Finally, we verify our results are not sensitive to exclusion of any specific district, or state, and that even when we relax the requirement of a fully balanced sample, we recover qualitatively similar estimates.

Our work adds to a growing body of literature in economics that uses quasi-experimental settings to estimate the social costs of biodiversity losses. This builds on a theoretical foundation in ecology that explores how declines in species that perform important ecosystem functions can have effects beyond the interactions within the ecosystem (Dirzo et al. 2014; D U 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). Drawing on variation in ecosystems to construct research designs, these recent empirical findings demonstrate how key predictions from ecology can be tested. The resulting well-identified evidence contributes quantified estimates that previous theoretical work in economics highlighted as important for the optimal management of biodiversity as a resource (Weitzman 1992; Solow et al. 1993; Weitzman 1993; 1998; Nehring and Puppe 2002; Brock and Xepapadeas 2003). Earlier work by Banerjee et al. (2010) used an invasive species as an income shock to study the effect it has on adulthood outcomes. Using variation in environmental suitability, Alsan (2015) studies the long-term effects that result from the presence of the tsetse fly on agricultural production and political institutions. Lange et al. (2009) and Bloome et al. (2017) study the economic and demographic implications of the invasion of the Boll Weevil, a crop pest that plagued cotton fields in the US South.

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); how insecticide use increases in years in which cicadas emerge (Taylor 2021), and exploit the change in insecticide use to study their health impacts (Frank 2021; Taylor 2021); and how reintroducing wolves can change the behavior and density of deer and reduce deer-vehicle-collisions (Raynor et al. 2021). While the exact source differs, these studies exploit a plausibly exogenous shock to an ecosystem to learn about the causal treatment effect of a species of interest.

Previous studies regarding the vulture collapse in the Indian sub-continent have mostly focused on documenting its magnitude, spatial extent, and whether restrictions on the use of diclofenac resulted in a recovery of the affected vulture species (Vibhu Prakash et al. 2012; Cuthbert et al. 2014; Galligan et al. 2020). To the best of our knowledge, the closest paper to our work by Markandya et al. (2008) used nationwide data and performed a back of the envelope calculation suggesting the socio-economic costs are around 34 billion dollars for the period between 1993 and 2006. As part of the calculation, they extrapolated from survey data to calculate that with the increase in available carcasses, based on caloric requirements of dogs, a decline of 10 million vultures is consistent with an increase of more than seven million dogs. Here we use 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 effect of their decline.

In what follows, we summarize the conditions that led to the decline of vulture populations, and proceed to describe the use of the data in our empirical strategy, along with the key findings of the analysis.

2 The Sudden Collapse of Vulture Populations in India

Vultures were once an ubiquitous sight across India. Their collapse is the fastest of a bird species in recent history and the largest in magnitude since the extinction of the passenger pigeon in the United States. While several vulture species are still present in India, the three that made up the bulk of the population are considered critically endangered after declining by more than 95% from population counts in the millions.[3] Their collapse was due

[3] 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 additional 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

to chemical residue of the pain killer diclofenac in livestock animals, after it was administered by livestock farmers. Within weeks, vultures that fed on a carcass with such residue would develop kidney failure and die.[4]

Livestock farmers started using diclofenac, a type of non-steroidal anti-inflammatory drug, to treat wounded animals.[5] Farmers administered diclofenac to their cattle to treat injuries, inflammations, and to help them recover from fevers (Gorman 2004; Subramanian 2015). With the use of diclofenac, livestock animals would recover faster, and would be more docile to manage during their recovery (Cuthbert et al. 2014).

The sudden adoption of diclofenac among livestock farmers was due to the emergence of a generic version of the drug. Anecdotal accounts place the timing of the patent expiration in the early 1990s (Subramanian 2015). 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 its expiration to 1993. See the Appendix for additional details. Further evidence, in the form of recall surveys from 2004, helps to more precisely determine the timing of diclofenac use for veterinary purposes in India. The survey responses found that diclofenac was widely available for sale in veterinary clinics by 1994 (Cuthbert et al. 2014). With these three sources of information, we classify 1994 as the first year in which diclofenac was widely used, and assign it as the year of treatment onset.

Following the large-scale use of diclofenac, reports of vulture declines began to emerge. Field observations in 1996 found only half of the three-hundred and fifty-three 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.

have diclofenac residue that caused the collapse in the other species.

[4] For clarity, we use the term kidney failure, while the literature uses the medical terms renal failure and visceral gout.

[5] Diclofenac is sold in India under the brand name Voveran, whereas internationally it is known as Voltaren (Sahajwalla et al. 1991).

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 (2011).[6]

At first, several conjectures were made regarding the potential cause. Plausible causes were considered to be an emergence of a new wildlife disease or the effect of pesticide accumulation, as well as deliberate poisoning by western countries (2011). However, it was not until 2004 when Oaks et al. (2004) used both autopsy data, and experimental exposure of vultures to diclofenac, that the causal link was established. Even small 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. Ogada et al. 2012). However, surveys conducted up to 2018 documented continued and rampant illicit use of diclofenac in livestock (Galligan et al. 2020).

Despite the 2006 ban on veterinary use, vulture populations are far from fully recovered. This is not surprising given that 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 (Subramanian 2011; D. L. Ogada et al. 2012). The ongoing use, even if at lower capacity, of diclofenac prevents seeing large population gains as vultures can still feed on carcasses with diclofenac residue.

In the absence of vultures, livestock farmers and municipalities can utilize either labor intensive or capital intensive substitutions. Farmers can exercise deep burial, where the livestock carcass is buried in a landfill or other suitable area. Given the number of livestock

[6] Population estimates for white-backed vultures estimated around thirty million in 1980, with a decline to about eleven thousand by 2010. All three species combined had recent historical population peaks around fifty million, yet populations estimates circa 2010 placed that number around sixty thousand (2011).

animals, this adds high labor costs for livestock operations. Alternatively, livestock carrion can be disposed of using specially designed incinerators, yet they are prohibitively expensive to construct and operate. According to a 2020 report by the Indian 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).

3 Vulture Presence, Health, & Livestock Census Data

In this section, we briefly summarize the data sets for the presence of vulture species, health outcomes, dogs counts, and water quality we use in the analysis. We provide additional details in the Appendix. In Table 1, we provide descriptive statistics for the main variables we used in the analysis, and how they are balanced across the high- and low-vulture-suitability districts at baseline.

3.1 Vulture Habitat Ranges

We obtain maps from BirdLife International (BLI) on the species distribution ranges of all bird species. In our analysis, we extract the range maps for vulture species, and perform two spatial calculations with the 1981 district boundaries: whether the district intersects with the range map, and the area of overlap between the range map and the district. 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. This approach is more flexible, and less dependent on functional form assumptions previously used to relate environmental suitability to outcomes of interest (Alsan 2015).

Vulture species that are negatively affected by diclofenac are mostly concentrated inland. Figure 1 shows the spatial distribution of the classification into high and low suitability categories for diclofenac-affected-vultures. In general, the further you move away from the center of the country, the lower the suitability is for the three affected species.

The data in the species distribution maps provided by BLI is regarded as the most complete source of information regarding the habitat areas of bird species around the world. BLI also assess the conservation status and extinction risk as part of the Red List, produced by the International Union for Conservation of Nature.[7] 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.[8] 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.[9]

3.2 Sales & Product Entry of Pharmaceuticals in India

We use data provided by the Indian Statistical Institute on the sales and the patent dates of drugs across India from 1991 to 2003. The data include information about the main active ingredient, the concentration, usage (topical, oral, or injection), as well as data quantity sold, value sold, and the year when product was launched. We focus on painkillers where diclofenac is the active ingredient, especially the products that are injected as those are more likely to be used in livestock animals. In Figure 2a, we plot the quantity sold and new product entries across India of veterinary-related diclofenac products. While we see an increasing trend from 1991 to 1996, there is a clear jump in sales in 1997 along with the entry of five new products.

[7] 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.

[8] Some records are considered confidential as their release might jeopardize the species if they are actively traded in domestic and international wildlife trade markets.

[9] 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.

3.3 Observation Records of Bird Species

There are no detailed survey data on vulture populations that allow us to compare changes in the presence of vultures at the district level. In the absence of scientifically collected data on the population levels of birds, we instead use citizen science data from eBird on the recorded observations of different bird species at the national level. We count the number of observations of either the diclofenac-affected-vultures, all vulture species, all bird species, and all bird species that have non-zero observations reported each year during 1990 to 2005.

We summarize the share of the diclofenac-affected-vultures relative to each other group. Taking the ratio between two groups allows us to account for growing trends in reporting, as long as those are not changing differentially over time for different bird species. In Figure 2b, we report a decline in the share of diclofenac-affected-vultures relative to each other group of bird species, which coincides with the timing of when reports of large vulture die-offs began to emerge in 1996. It is likely that once it became known that vulture were declining in the wild, bird enthusiasts dedicated more effort to documenting them in the wild, upward biasing our measure of vulture observations. Consequently, this simple national-level comparison recovers a lower bound for the magnitude of the decline of the diclofenac-affected-vultures.

We complement the observational data from eBird by reproducing a previously documented sharp decline of three orders of magnitude in a set of survey results that spanned 1992 to 2007 (Prakash et al. 2007). At five different years, survey teams traveled along the same 70 road transects and counted vulture species. In Figure A1, we plot the data from the same 70 road transects that show a decline from over tens of thousands of recorded vultures to just a few hundred vultures by 2003.

3.4 Health Outcomes

In the paper we use mortality data at the district level from the Vital Statistics of India (VSI), reported as part of the Civil Registration System (CRS). The VSI-CRS data provide us with sub-state-level data, yet are known to suffer from issues of underreporting (we compare the extent of under-reporting in a following subsection). 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 a denser village. An area is officially classified as urban if it has a population above 5,000 people, or if more than 75% of men work in non-agricultural jobs (Burgess et al. 2017).

The VSI-CRS data experienced a shift in 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 population growth rates, and use an exponential growth function to interpolate population during inter-censal 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 Appendix, we provide the results for the full 1981 to 2005 period.

Using the classification into high and low suitability for the diclofenac-affected-vultures, we plot the mean population-weighted all-cause death rate across in Figure 2c. We observe an increase in mortality in the high-vulture-suitability districts following the onset of diclofenac use in 1993. However, no similar change in magnitude or trend is observed in the lowest suitability category. The habitat suitability groups overlap 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.

3.5 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).

3.6 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. Surprisingly, the data in the livestock census also report a count of dogs at the district level.[10]

We collected data from the states on their livestock censuses that were conducted in 1987, 1992, 1997, 2003, and 2012. Our main variable of interest is the number of dogs recorded at the district level in 2012 as it allows us to provide suggestive evidence for two key mechanisms for an adverse health shock following the decline in vulture populations. If dog populations increased more 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. In addition, it provides a proxy for an increase in mammalian scavengers such as dogs and rats. We also use the data to classify districts as high or low livestock districts at baseline, which we use as part of a triple-differences design (see the empirical strategy section for more details).

3.7 Quantifying Under-Reporting in the CRS Data

In the analysis, we rely on data from the Civil Registration System (CRS) as it offers data at the district level. One known limitation of the CRS data is that many vital statistics events

[10] 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. Only since 2012 did the livestock census start to systematically collect data on feral dog populations.

go unrecorded, and as a result, the CRS under-reports the true magnitude of mortality. As our analysis compares changes over time using the CRS data, we are still able to recover the level differences. However, when we interpret those effects relative to a baseline level of mortality, using the mean mortality reported in the CRS data will over-estimate the relative change in mortality.

In order to obtain a nationally representative baseline of mortality, we use the Sample Registration System (SRS) data. The SRS samples less than one percent of the population, but is designed to recover a nationally representative sample (Rao and Gupta 2020). While we do find that at the national level, the CRS underestimates mortality rates by about a factor of two relative to the SRS, when controlling for state and zonal council-by-year fixed effects, both recover similar trends in mortality rates (see Figure C1). Consequently, in the interpretation of the analysis, we interpret the magnitude of the coefficients relative to the mean level from the SRS data, which reflect the national-level death rate.

4 The Collapse of Vultures in India as a Natural Experiment

To estimate the causal effect of the collapse in vulture populations on public health, one would ideally randomly assign vulture densities. Fully randomizing the distribution and abundance of vultures could allow to study their effects through the channels of sanitation provision, and crowding other mammalian scavengers, such as dogs and rats. In practice, historic vulture densities were determined through a combination of environmental conditions, and the collapse in vulture populations started after the simultaneous large-scale adoption of diclofenac by cattle farmers around 1994.

The setting of diclofenac use in India provides plausibly exogenous variation in vulture densities. The combination of vulture populations at baseline, along with the shock to specific species, approximates the ideal experiment discussed above. Our empirical strategy relies on two key assumptions. First, that vulture populations were in equilibrium prior to the onset of diclofenac use. Second, that diclofenac use was not restricted only to the areas with high suitability for diclofenac-affected vultures.

Under the assumptions that the ecosystem was in equilibrium, and that diclofenac use offers as-good-as-random variation in vulture density, then it is likely that public health outcomes were developing along similar trends. Vultures were widely considered abundant with populations in the millions, and were even playing a key role in burial rites for the Parsi, which support that the ecosystem equilibrium assumption holds (Subramanian 2011). Diclofenac use started after the patent held by Novartis expired in 1993 (see section 2 for more details). The connection between the decline in vulture populations and diclofenac use was not made until 2004, which supports the exogeneity assumption.

Our approach treats the decline in vultures as a shock to the functioning of the ecosystem. This shock resulted in potential reductions in the provision of sanitation, and a potential increase in dog and rat abundance, leading to higher rates of animal bites and transmission of infectious diseases. We exploit this sharp change in environmental conditions and compare districts that were highly-suitable for vultures to those less suitable, before and after the 1994 onset of diclofenac use. The key identifying assumption in our difference-in-differences design is that districts would have seen their health outcomes develop along parallel trends in the absence of the collapse in vulture populations.

We explore the similarity in the outcomes of interest along with additional covarites across the high and low vulture suitability groups prior to the collapse of vultures in Table 1. We find that the mean all-cause death rate between 1988 and 1993 was lower by 1.2 deaths per-1,000 people in the high-vulture-suitability districts relative to the low-vulture-suitability districts. We also find no difference in the mean number of livestock animals as recorded in the livestock censuses of 1987 and 1992. Combined, we interpret the two comparisons as that in the early 1990s, districts with high or 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 in the low-vulture-suitability districts.

The difference is suitability in also reflected in differences in weather and precipitation. Districts with high suitability have more warm days, and less precipitation. While we find small differences in the popualtion share by age-group, and a substantial difference in the literacy rate, we do not detect any meaningful differences in baseline water quality or water access. More importantly, we 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. Our identifying assumptions do not require the districts to be fully balanced across the vulture suitability gradient, but this comparison helps to rule out that there are clear differences between districts that might explain a divergence in mortality over time.

4.1 Differences-In-Differences Design

Exploiting the sudden and drastic change in the presence of vultures across the Indian sub-continent, we examine how health changed in the between the high- and low-vulture suitability districts, before and after the decline in vulture populations. 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 estimate the following event-study-like difference-in-differences (DD) regression specification:

$$
\begin{aligned}
y_{daszt} = & \sum_{\substack{\tau \in \{\underline{T}, \ldots, \overline{T}\} \\ \tau \neq 1993}} \beta_\tau (\text{High Vulture Suitability})_d \times \mathbb{1}\{t = \tau\} + \\
& \lambda_{da} + \delta_{zt} + \boldsymbol{X_{daszt}\theta} + \varepsilon_{daszt} \qquad (1)
\end{aligned}
$$

Where high vulture suitability is a dummy variable that equals one for districts that we classify as highly suitable for the three vulture species affected by the exposure to diclofenac.

Explicitly, we define high suitability as either being in the third and second terciles of the overlap between vulture ranges and districts areas (see Figure 1). We interact the suitability dummy that defines that treated group with year dummies. We omit 1993 as the baseline year as that is the year in which the patent on diclofenac expired, allowing the generic pharmaceutical industry to produce the painkiller, making it widely accessible for farmers (see section 2 for more details).

The coefficients on the interaction term, 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 bound of the effects following the collapse in vulture populations. Even the districts we classify as low suitability are likely affected as they overlap to some degree with at least one affected vulture species.[11] This means that our analysis is leveraging differences in the intensity of the collapse experienced in each district. Consequently, the control group of low-suitability districts provides us with an approximation to a counterfactual of lower treatment intensity, but not an absolute of zero treatment.

We are interested in the residual variation that is not explained by time invariant characteristics of districts, or pooled time-trends. To account for district observable and unobservable traits that are constant throughout the sample, we include district-area fixed effects, λ_{da}. These help to control for any baseline differences in sanitation, morbidity, mortality,

[11] In mainland India, there are only two districts that do not overlap with any of the ranges of the diclofenac-affected vulture species.

and healthcare access.[12] We flexibly 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.[13]

To further account for time-varying effects, we include state-linear time trends as well as state-by-year fixed effects. We include these additional time controls as we are mostly concerned with differential reporting at the state level. If the states that we classify as high suitability for diclofenac-affected-vultures are also systematically those that increase their reporting of the outcomes of interest, then we could interpret the spurious correlation in reporting and high suitability as the effect of vulture population collapse. It is important to recognize that by using more granular fixed effects we also absorb more of the treatment effect as vulture-suitability is highly correlated within states.

To further test that any observed results are strictly driven by the interaction of vulture suitability and diclofenac use onset, we also include other control variables, $\boldsymbol{X_{daszt}}$. We include weather variables in the form of flexible degree days in intervals of three-degree Celsius bins, along with precipitation quintiles. Any unobserved variation is captured by the error term, ε_{daszt}. We allow standard errors to be correlated across years and across urban and rural areas within a district. In our baseline results, standard errors are not correlated across districts. In the Appendix, we relax the assumption on no spatial correlation of the standard errors using permutation inference.

To summarize the average treatment effects, we define a post-diclofenac use dummy variable that is equal to one for 1994 onward, and we collapse the post-treatment coefficients

[12] When running regressions that include data from both urban and rural areas, this fixed effect allows urban and rural areas in the district to have separate fixed effects. When we subset the data to urban or rural only, or when we combine the data from urban and rural areas, this collapses to a district fixed effect.

[13] This sub-national division is similar to census regions in the setting of the United States.

using the following specification:

$$
\begin{aligned}
y_{daszt} =&\beta(\text{High Vulture Suitability})_d \times (\text{Post-Diclofenac Use})_t + \\
&\lambda_{da} + \delta_{zt} + \boldsymbol{X_{daszt}\theta} + \varepsilon_{dszt}
\end{aligned} \tag{2}
$$

4.2 Utilizing Sub-Groups in a Triple-Differences Design

We add to the DD design by leveraging two more sub-groups where a decline in the scavenging function of vultures would have potentially resulted in larger detrimental impacts: urban areas and districts with higher livestock levels at baseline. Urban areas are more likely to have animal landfills, and are denser than rural areas. Consequently, we would expect that urban areas would experience a greater loss of sanitation, and potentially a larger increase in feral dogs and rat populations. In addition, districts that had a larger livestock sector at baseline, prior to the collapse of vultures, would have relied more strongly on the scavenging of vultures to dispose of animal carrion. Specifically, we use a dummy variable to denote sub-district areas a urban, or as above the median level of livestock as measured in the 1987 and 1992 livestock censuses to estimate a triple-difference (DDD) design:

$$
\begin{aligned}
y_{daszt} =&\beta(\text{High Vulture Suitability})_d \times (\text{Post-Diclofenac Use})_t \times (\text{Sub Group})_d + \\
&\lambda_{da} + \delta_{zt} + \boldsymbol{X_{daszt}\theta} + \varepsilon_{dszt}
\end{aligned} \tag{3}
$$

The urban and rural DDD design allows us to better control for time-varying characteristics that are pooled within the district, but could vary across districts. The urban versus rural estimation compares potentially higher to lower treatment intensity levels. However, it is likely that rural areas experienced a decline in sanitation, even if they experienced a smaller decline relative to urban areas. Using the high-livestock at baseline group allows us to test whether any meaningful changes in the outcomes of interest show up where the condition of having a supply of livestock carcasses is met.

5 The Consequences of Vulture Die-offs on Human Health

Here we present the evidence for the main finding of an increase in 0.85 deaths per-1,000 people, reflecting a 9.2% increase. After validating our results are robust to different temporal controls, sample compositions, and definitions of treatment, we present suggestive evidence on the mechanisms that link vulture decline with human health.

5.1 Results for All-Cause Death Rate

Following the onset of diclofenac use in 1994, 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. We estimate that, on average, death rates were higher by 9.2%, and that by 2005 they were almost 15% higher relative to the nationally representative mean level in 1993 of 9.2 deaths per-1,000 people, as reported in the SRS data.[14]

High and low suitability districts did not have systematically different trends with respect to death rates between 1988 and 1992, relative to 1993. Only after farmers widely started using diclofenac in 1994, did death rates increase in areas that overlap with the range of diclofenac-affected-vultures. In Figure 3, we report the event-study estimation results using Equation 1. 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 1.5 deaths per-1,000 people, reflecting an increase of 7.1% and 14.9% relative to the 1993 nationally representative mean level, respectively.

[14] 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 section 3.7: Quantifying Under-Reporting in the CRS Data for additional details.

Farmers rapidly adopted diclofenac leading to a sharp, but not instantaneous decline in observed vultures (see Figures 2b and A1). This means that the decline in the functional capacity of vultures as environmental sanitizers occurred over several years. As a result, we should expect to see a dynamic response where the treatment effect intensifies over time. With vulture populations reaching a new low equilibrium, so does the divergence in death rates. By the early 2000s, vulture populations were described as a shadow of their previous levels, in agreement with the stabilization of the difference between the the high and low suitability districts.

Using the data reported by sub-district division on urban and rural areas, we find that urban areas experienced an earlier increase in death rates relative to rural areas. In Figures 3b and 3c, we estimate how death rates changed in urban and rural areas separately, and find that death rates in both areas increased after 1994. In urban areas, the divergence started before it was evident in rural areas. This is consistent with the location of carrion landfills at the outskirts of areas classified as urban. Urban areas also have higher population densities that can result in higher infection rates following the loss of sanitation functions provided by vultures, and an increase in dog and rat populations.

Estimating the mean change in death rates, we find they increased by close to 10% in the post-diclofenac period of 1994 to 2005. In Table 2, we summarize the magnitude of the effects using different degrees of temporal controls, with or without accounting for local weather variation. In Panel A of Table 2, we report the results from the parsimonious specification, without weather controls. On average, death rates are higher by 0.81 or 0.91 deaths per-1,000 people when including either year or zonal council-by-year fixed effects (Table 2, Panel A, columns 1 and 2). Controlling for local weather variation reduces the average increase in death rates to 0.76 and 0.85 additional deaths per-1,000 people (Table 2, Panel B, columns 1 and 2).

In our sample, ranging from 1988 to 2005, the post-diclofenac period covers 12 years (1994 to 2005). We estimate the average post-diclofenac effect separately for the periods of 1994 to 1999, and 2000 to 2005. Accounting for zonal council-by-year fixed effects, 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 (Table 2, Panel A, column 3). When we add state-linear time trends, or include state-by-year fixed effects, the magnitude of the estimates drops, and is only precisely estimated during the latter 2000 to 2005 time period (Table 2, Panels A and B, columns 4 and 5). Including state-by-year fixed effects likely absorbs some of the treatment effect because the vulture suitability is highly spatially correlated within states.[15]

Our preferred specification includes zonal council-by-year fixed effects, and controls for local weather variation. This allows us to account for regional trends in death rates, as well as adjusting for the effect of weather as it is both an important driver of mortality, and a dimension on which our high and low vulture suitability districts are not fully balanced on. To better interpret the average increase in the all-cause death rate of 0.85 deaths per-1,000 people (Table 2, Panel B, column 2), we estimate the effect of weather on mortality (see the Appendix for a full description of the estimation and the results). We find that an additional hot day, above 34°C, relative to the reference category of 22°C to 24°C, results in close to 0.2 additional deaths per-1,000 deaths. Relative to the nationally representative level of mortality, this reflects an increase of close to 2.2%. Our mortality and weather relationship estimates are in agreement with previous work by Burgess et al. (2017) that spans a longer panel. Estimating the weather-mortality functions helps to anchor the effect of the decline in vultures as comparable to a week-long heatwave in India.

5.2 Sensitivity Analysis & Robustness Checks

In the Appendix, we evaluate the robustness of the main results in several ways. 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 A2). To better account

[15] Because we hold districts fixed at their 1981 borders, this results in aggregating some districts to their state level. As a results, three states are fully absorbed by the state-by-year fixed effects. To verify that the small change in composition does not meaningfully affect the results, we report in Table 2, column 6, the the estimation in column 3, using the sample in column 5.

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 A3).

In the analysis above, we require that districts have non-missing death rate data during 1988 to 2005. As part of the data quality issues with vital statistics data in India, gaps in reporting limit the number of districts that meet the balanced reporting condition to 153 districts, relative to the 339 districts in the sample. We address the potential concern regrading the representativeness of our balanced sample by using a long-differences comparison (Burke and Emerick 2016). Explicitly, we collapse the data to two periods: 1988 to 1993, and 2000 2005. In each period, we calculate the mean death rate, and estimate the DD specification in Equation (2) using the collapsed two period sample. This allows us to use data from up to 285 districts, covering 69% of the population in our sample. In the long-differences comparison, we find a precisely estimated increase of 0.61 deaths per-1,000 people (Table A1, Panel A, column 3).

We explore whether alternative treatment status assignment affect our results by estimating 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. The habitat range maps produced by BLI, which we use to classify districts as having high or low vulture suitability, also rely on a habitat suitability model along with expert knowledge and other unpublished records. 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 the Appendix 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 another 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 A4, and report the average treatment effects in Table A3. 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 Figure A5 and A6. Finally, we preform 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 are in the right tail of the distribution. The permutation inference analysis also allows us to evaluate whether we are underestimating the standard errors by clustering at the district level due to spatial correlation of the error term. This appears to not be a concern as the exact p-values we obtain are well below 1%.

5.3 Differential Impacts in Urban Areas & High Livestock Districts

Two potential sources of treatment heterogeneity are the availability of animal landfills, and the amount of livestock in the district. We use the data reported for sub-district urban and rural areas and re-estimate the average effects in Table A2. We find a larger increase of 0.91 deaths relative to 0.79 deaths in urban relative to rural areas (Table A2, Panels A and B, column 4). However, we cannot reject that the effects are the same. Estimating the DDD specification in Equation (3) using the long-difference sample, we do not find a differential effect when using the fully balanced sample, but do find a large and precisely estimated effect in the expanded sample, which includes districts with at least some years of data in the pre- and post-collapse years (Table A1, Panel B, columns 2 and 4).

We proceed to compare districts that had a larger livestock population at baseline and find that those are the districts that see the largest increase in the death rate. India provides district-level counts of livestock as part of the livestock census that is carried out roughly every five years. We use the data from the census in 1987 and 1992 to calculate the mean level of livestock for each district. We define a district as high livestock at baseline if they are above the median level of livestock in levels, or normalized by district area. In Table 3, we report the DDD results from Equation (3). We find that following the collapse in vulture populations, high-vulture-suitability districts that had a high level of livestock at baseline saw an increase of 0.73 deaths per-1,000 people (Table 3, column 4). This is in strong agreement that the key driver of mortality after the collapse in vulture populations is the availability of a large supply of animal carrion that is not getting disposed of by vultures, and not simply the decline in vultures absent of livestock.

5.4 Suggestive Evidence For Increasing Dogs, Rabies & Degraded Water Quality

India has limited data on the number of feral dogs, the prevalence of rabies, and there are few water quality monitoring stations, especially ones that report consistently over time. We use different data collections on all three outcomes to explore whether they provide supporting evidence for the key mechanisms that link a decline in vulture populations to adverse human health outcomes.

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. Starting in 2012, India began collecting data on feral dogs as part of its livestock census. Using the cross-sectional data on feral dog counts, we find suggestive evidence for an increase in the number of dogs following the decline in vulture populations. In Figure **??**, we plot the correlation between the binned values of feral dogs, in log points, to the mean habitat overlap with the diclofenac-affected-vultures. We observe a strong association between the degree of habitat suitability and feral dog counts. 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.

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). 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).

We use data on water quality outcomes that are most directly linked to a larger presence of carrion when disposal by scavengers declines: dissolved oxygen and fecal coliforms. 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 coliforms.

In Table 4, we find that water quality deteriorates, but only in the urban subsample (columns 2, 3, 5, and 6). Dissolved oxygen drops by 12% in the DDD comparison (column 2), while dropping by 7% in the urban subsample (column 6). Fecal coliforms more than double in water samples in either the DDD or DD comparison (columns 5 and 6). In the Appendix Table A4, 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).

6 Conclusions

Using a panel of health outcomes at the district level, we provide evidence on the public health implications from the decline of vulture species in India. The collapse in their populations was a direct effect of the use of the painkiller diclofenac in livestock. Affected vulture species would die within weeks if they fed on a carrion with diclofenac residue. We find that all-cause death rates increased in the districts that are considered more suitable to the diclofenac-affected vultures to those less suitable before and after the large-scale onset of diclofenac use. Our results inform current vulture recovery efforts in India, and global conservation policies more broadly. Further research is needed on the role that other species play in key outcomes of interest.

The findings have direct implications for evaluating regulatory steps taken in India and elsewhere to restrict the use of diclofenac, as well as providing a basis for funding of recovery programs for vultures in India. The empirical results highlight that disturbances to ecosystem interactions are not necessarily confined to the species who are directly affected. As these interactions can lead to changing environmental conditions, human settlements that are linked to these ecosystems can experience adverse effects as well. Our analysis demonstrates that through the use of quasi-experimental methods, we can uncover the role that different species have in production functions of interest, such as public health. More broadly, the vulture collapse in India provides a particularly stark example of the type of irreversible and unpredictable costs that must be accounted for when evaluating the introduction of new chemicals into fragile and diverse ecosystems.

Beyond the direct effects on ecosystem interactions, and their impacts on public health, the loss of vultures also has ramifications for the leather tanning industry, and burial rituals practiced by a specific ethnic group in India. The tanning industry relies on quick removal of carrion by vultures. In their absence, in some places, people have shifted to burying or burning cattle which reduces the supply of cattle skin for leather manufacturing (Markandya et al. 2008). The Parsi community in India has burial rituals that require vultures to consume the body. Following their decline, practitioners have experienced the discomfort of discovering that the bodies of their relatives are not going through the ritual as intended (2008; Subramanian 2011).

References

Alsan, Marcella. 2015. “The Effect of the TseTse Fly on African Development.” *The American Economic Review* 105 (1): 382–410.

Arrow, Kenneth J, and Anthony C Fisher. 1974. “Environmental Preservation, Uncertainty, and Irreversibility.” *Quarterly Journal of Economics* 88 (2): 312–319.

Banerjee, Abhijit, Esther Duflo, Gilles Postel-Vinay, and Tim Watts. 2010. “Long-Run Health Impacts of Income Shocks: Wine and Phylloxera in Nineteenth-Century France.” *The review of economics and statistics* 92 (4): 714–728.

Barrios, Thomas, Rebecca Diamond, Guido W Imbens, and Michal Kolesár. 2012. “Clustering, Spatial Correlations, and Randomization Inference.” *Journal of the American Statistical Association* 107 (498): 578–591.

Bloome, Deirdre, James Feigenbaum, and Christopher Muller. 2017. “Tenancy, Marriage, and the Boll Weevil Infestation, 1892-1930.” *Demography* 54 (3): 1029–1049.

Booth, Trevor H, Henry A Nix, John R Busby, and Michael E F Hutchinson. 2014. “BIOCLIM: the first species distribution modelling package, its early applications and relevance to most current MaxEnt studies.” Edited by Janet Franklin. *Diversity & distributions* 20 (1): 1–9.

Braczkowski, Alexander R, Christopher J O'Bryan, Martin J Stringer, James E M Watson, Hugh P Possingham, and Hawthorne L Beyer. 2018. "Leopards provide public health benefits in Mumbai, India." *Frontiers in ecology and the environment* 16 (3): 176–182.

Brock, William A, and Anastasios Xepapadeas. 2003. "Valuing Biodiversity from an Economic Perspective: A Unified Economic, Ecological, and Genetic Approach." *The American Economic Review* 93 (5): 1597–1614.

Brundage, Adrienne. 2021. "Carrion Ecology." In *Wildlife Biodiversity Conservation: Multidisciplinary and Forensic Approaches,* edited by Susan C Underkoffler and Hayley R Adams, 193–210. Cham: Springer International Publishing.

Burgess, Robin, Olivier Deschenes, Dave Donaldson, and Michael Greenstone. 2017. "Weather, Climate Change and Death in India." *Working Paper.*

Burke, Marshall, and Kyle Emerick. 2016. "Adaptation to Climate Change: Evidence from US Agriculture." *American Economic Journal: Economic Policy* 8 (3): 106–140.

Cardinale, Bradley J, J Emmett Duffy, Andrew Gonzalez, David U Hooper, Charles Perrings, Patrick Venail, Anita Narwani, et al. 2012a. "Biodiversity loss and its impact on humanity." *Nature* 486 (7401): 59–67.

———. 2012b. "Biodiversity loss and its impact on humanity." *Nature* 486 (7401): 59–67.

Ceballos, Gerardo, Paul R Ehrlich, Anthony D Barnosky, Andrés Garcı a, Robert M Pringle, and Todd M Palmer. 2015. "Accelerated modern human–induced species losses: Entering the sixth mass extinction." *Science Advances* 1 (5): 1–5.

Central Pollution Control Board. 2020. *Guidelines for Carcass Disposal.*

Cuthbert, Richard J, Mark A Taggart, Vibhu Prakash, Soumya S Chakraborty, Parag Deori, Toby Galligan, Mandar Kulkarni, et al. 2014. "Avian scavengers and the threat from veterinary pharmaceuticals." *Philosophical transactions of the Royal Society of London. Series B, Biological sciences* 369 (1656).

Dainese, Matteo, Emily A Martin, Marcelo A Aizen, Matthias Albrecht, Ignasi Bartomeus, Riccardo Bommarco, Luisa G Carvalheiro, et al. 2019. "A global synthesis reveals biodiversity-mediated benefits for crop production." *Science advances* 5 (10): eaax0121.

Dasgupta, Partha, and Geoffrey Heal. 1974. "The Optimal Depletion of Exhaustible Resources." *The Review of economic studies* 41:3–28.

Dirzo, Rodolfo, Hillary S Young, Mauro Galetti, Gerardo Ceballos, Nick J B Isaac, and Ben Collen. 2014. "Defaunation in the Anthropocene." *Science* 345 (6195): 401–406.

Engel, Bernard A, Kyoung Jae Lim, Jin-Yong Choi, and Larry Theller. 2004. "Evaluating Environmental Impacts." In *Carcass disposal: A comprehensive review.* {National Agricultural Biosecurity Center Consortium USDA APHIS Cooperative Agreement Project}.

Estes, James A, John Terborgh, Justin S Brashares, Mary E Power, Joel Berger, William J Bond, Stephen R Carpenter, et al. 2011. "Trophic downgrading of planet Earth." *Science* 333 (6040): 301–306.

Ferraro, Paul J, James N Sanchirico, and Martin D Smith. 2019. "Causal inference in coupled human and natural systems." *Proceedings of the National Academy of Sciences of the United States of America* 116 (12): 5311–5318.

Fisher, R A. 1966. *The Design of Experiments.* 8th. 866 Third Avenue, New York, NY 10022: Hafner Publishing Company.

Frank, Eyal. 2021. "The Economic Impacts of Ecosystem Disruptions: Private and Social Costs From Substituting Biological Pest Control." *Working Paper.*

Frank, Eyal G, and Wolfram Schlenker. 2016. "Balancing economic and ecological goals." *Science* 353 (6300): 651–652.

Galligan, Toby H, John W Mallord, Vibhu M Prakash, Krishna P Bhusal, A B Sarowar, Fergus M Anthony, Ruchi Dave, et al. 2020. "Trends in the availability of the vulture-toxic drug, diclofenac, and other NSAIDs in South Asia, as revealed by covert pharmacy surveys." *Bird Conservation International,* 1–17.

Gorman, James. 2004. "A Drug Used for Cattle Is Said to Be Killing Vultures." *The New York Times.*

Greenstone, Michael, and Rema Hanna. 2014. "Environmental Regulations, Air and Water Pollution, and Infant Mortality in India." *The American economic review* 104 (10): 3038–3072.

Heal, Geoffrey. 2000. "Valuing Ecosystem Services." *Ecosystems* 3 (1): 24–30.

Hill, Jacob E, Travis L DeVault, James C Beasley, Olin E Rhodes Jr, and Jerrold L Belant. 2018. "Effects of vulture exclusion on carrion consumption by facultative scavengers." *Ecology and Evolution* 8 (5): 2518–2526.

Hooper, D U, F S Chapin, J J Ewel, A Hector, P Inchausti, S Lavorel, J H Lawton, et al. 2005. "Effects of Biodiversity on Ecosytem Functioning: A Consensus of Current Knowledge." *Ecological monographs* 75 (1): 3–35.

Hooper, David U, E Carol Adair, Bradley J Cardinale, Jarrett E K Byrnes, Bruce A Hungate, Kristin L Matulich, Andrew Gonzalez, J Emmett Duffy, Lars Gamfeldt, and Mary I O'Connor. 2012. "A global synthesis reveals biodiversity loss as a major driver of ecosystem change." *Nature* 486 (7401): 105–108.

Hugo. 2021. *Vultures - small things and big consequences.* https://www.p3mountains.org/post/vultures-small-things-and-big-consequences. Accessed: 2022-3-10.

Jones, Benjamin A, and Shana M McDermott. 2018. "Health Impacts of Invasive Species Through an Altered Natural Environment: Assessing Air Pollution Sinks as a Causal Pathway." *Environmental & Resource Economics* 71 (1): 22–43.

Kwon, Man Jae, Seong-Taek Yun, Baknoon Ham, Jeong-Ho Lee, Jun-Seop Oh, and Weon-Wha Jheong. 2017. “Impacts of leachates from livestock carcass burial and manure heap sites on groundwater geochemistry and microbial community structure.” *PloS one* 12 (8): e0182579.

Lange, Fabian, Alan L Olmstead, and Paul W Rhode. 2009. “The Impact of the Boll Weevil, 1892–1932.” *The journal of economic history* 69 (3): 685–718.

Luis, Angela D, Amy J Kuenzi, and James N Mills. 2018. “Species diversity concurrently dilutes and amplifies transmission in a zoonotic host-pathogen system through competing mechanisms.” *Proceedings of the National Academy of Sciences of the United States of America.*

Markandya, Anil, Tim Taylor, Alberto Longo, M N Murty, S Murty, and K Dhavala. 2008. “Counting the cost of vulture decline—An appraisal of the human health and other benefits of vultures in India.” *Ecological Economics* 67 (2): 194–204.

Martin, Emily A, Björn Reineking, Bumsuk Seo, and Ingolf Steffan-Dewenter. 2013. “Natural enemy interactions constrain pest control in complex agricultural landscapes.” *Proceedings of the National Academy of Sciences of the United States of America* 110 (14): 5534–5539.

Metrick, Andrew, and Martin L Weitzman. 1998. “Conflicts and Choices in Biodiversity Preservation.” *The journal of economic perspectives: a journal of the American Economic Association* 12 (3): 21–34.

Moleón, Marcos, José A Sánchez-Zapata, Antoni Margalida, Martina Carrete, Norman Owen-Smith, and José A Donázar. 2014. “Humans and Scavengers: The Evolution of Interactions and Ecosystem Services.” *Bioscience* 64 (5): 394–403.

Naeem, Shahid, Daniel E Bunker, Andy Hector, Michel Loreau, and Charles Perrings. 2009. *Biodiversity, Ecosystem Functioning, and Human Wellbeing: An Ecological and Economic Perspective.* OUP Oxford.

Nehring, Klaus, and Clemens Puppe. 2002. “A Theory of Diversity.” *Econometrica: journal of the Econometric Society* 70 (3): 1155–1198.

Oaks, J Lindsay, Martin Gilbert, Munir Z Virani, Richard T Watson, Carol U Meteyer, Bruce A Rideout, H L Shivaprasad, et al. 2004. “Diclofenac residues as the cause of vulture population decline in Pakistan.” *Nature* 427 (6975): 630–633.

Ogada, Darcy, Phil Shaw, Rene L Beyers, Ralph Buij, Campbell Murn, Jean Marc Thiollay, Colin M Beale, et al. 2016. “Another Continental Vulture Crisis: Africa’s Vultures Collapsing toward Extinction: African vultures collapsing toward extinction.” *Conservation Letters* 9 (2): 89–97.

Ogada, Darcy L, Felicia Keesing, and Munir Z Virani. 2012. “Dropping dead: causes and consequences of vulture population declines worldwide.” *Annals of the New York Academy of Sciences* 1249:57–71.

Polasky, Stephen, and Andrew R Solow. 1995. “On the value of a collection of species.” *Journal of environmental economics and management* 29 (3): 298–303.

Prakash, V, R E Green, D J Pain, S P Ranade, S Saravanan, N Prakash, R Venkitachalam, R Cuthbert, A R Rahmani, and A A Cunningham. 2007. “Recent changes in populations of resident Gyps vultures in India.” *Journal of the Bombay Natural History Society* 104 (2): 129–135.

Prakash, Vibhu, Mohan Chandra Bishwakarma, Anand Chaudhary, Richard Cuthbert, Ruchi Dave, Mandar Kulkarni, Sashi Kumar, et al. 2012. “The population decline of Gyps vultures in India and Nepal has slowed since veterinary use of diclofenac was banned.” *PloS one* 7 (11): e49118.

Ramesh, Vijay, Trisha Gopalakrishna, Sahas Barve, and Don J Melnick. 2017. “IUCN greatly underestimates threat levels of endemic birds in the Western Ghats.” *Biological conservation* 210:205–221.

Rao, Chalapati, and Mamta Gupta. 2020. “The civil registration system is a potentially viable data source for reliable subnational mortality measurement in India.” *BMJ Global Health* 5 (8).

Raynor, Jennifer L, Corbett A Grainger, and Dominic P Parker. 2021. “Wolves make roadways safer, generating large economic returns to predator conservation.” *Proceedings of the National Academy of Sciences of the United States of America* 118 (22).

Roggenbuck, Michael, Ida Bærholm Schnell, Nikolaj Blom, Jacob Bælum, Mads Frost Bertelsen, Thomas Sicheritz-Pontén, Søren Johannes Sørensen, M Thomas P Gilbert, Gary R Graves, and Lars H Hansen. 2014. “The microbiome of New World vultures.” *Nature communications* 5:5498.

Sahajwalla, C G, A D Bhatt, S C Bhatia, R Bakshi, K J Doshi, M M Banavalikar, E D Bharucha, K V Marthak, N N Shah, and K C Gupta. 1991. “Comparative bioavailability of slow release diclofenac (Voveran SR) with enteric coated tablet and internationally used Voltaren Retard.” *The Journal of the Association of Physicians of India* 39 (7): 546–548.

Santori, Claudia, Ricky-John Spencer, Michael B Thompson, Camilla M Whittington, Thomas H Burd, Samantha B Currie, Timothy J Finter, and James U Van Dyke. 2020. “Scavenging by threatened turtles regulates freshwater ecosystem health during fish kills.” *Scientific Reports* 10 (1): 14383.

Schmeller, Dirk S, Franck Courchamp, and Gerry Killeen. 2020. “Biodiversity loss, emerging pathogens and human health risks.” *Biodiversity and conservation.*

Seddon, Philip J, Doug P Armstrong, and Richard F Maloney. 2007. “Developing the science of reintroduction biology.” *Conservation Biology* 21 (2): 303–312.

Sharma, Rajiv J, and Alok J Singh. 2016. *Writ Petition (PIL) No.140 of 2015.*

Solow, Andrew, Stephen Polasky, and James Broadus. 1993. “On the Measurement of Biological Diversity.” *Journal of environmental economics and management* 24 (1): 60–68.

Subramanian, Meera. 2011. “Can the world’s fastest growing nation restore its prime scavenger before there are untold human consequences?” *VQR.*

Subramanian, Meera. 2015. *A River Runs Again: India's Natural World in Crisis, from the Barren Cliffs of Rajasthan to the Farmlands of Karnataka.* Hachette UK.

Swift, Michael John, O W Heal, Jonathan Michael Anderson, and J M Anderson. 1979. *Decomposition in Terrestrial Ecosystems.* University of California Press.

Taylor, Charles. 2021. "Cicadian Rhythm: Insecticides, Infant Health and Long-term Outcomes." *CEEP WP 9.*

Tschirhart, John. 2009. "Integrated Ecological-Economic Models." *Annual Review of Resource Economics* 1 (1): 381–407.

Vijaikumar, M, Devinder M Thappa, and K Karthikeyan. 2002. "Cutaneous anthrax: an endemic outbreak in south India." *Journal of tropical pediatrics* 48 (4): 225–226.

Watson, James E M, Danielle F Shanahan, Moreno Di Marco, James Allan, William F Laurance, Eric W Sanderson, Brendan Mackey, and Oscar Venter. 2016. "Catastrophic Declines in Wilderness Areas Undermine Global Environment Targets." *Current Biology* 26 (21): 2929–2934.

Watson, Richard T, Martin Gilbert, J Lindsay Oaks, and Munir Virani. 2004. "The collapse of vulture populations in South Asia." *Biodiversity* 5 (3): 3–7.

Weitzman, Martin L. 1992. "On diversity." *The quarterly journal of economics,* 363–405.

———. 1993. "What to Preserve? An Application of Diversity Theory to Crane Conservation." *The quarterly journal of economics* 108 (1): 157–183.

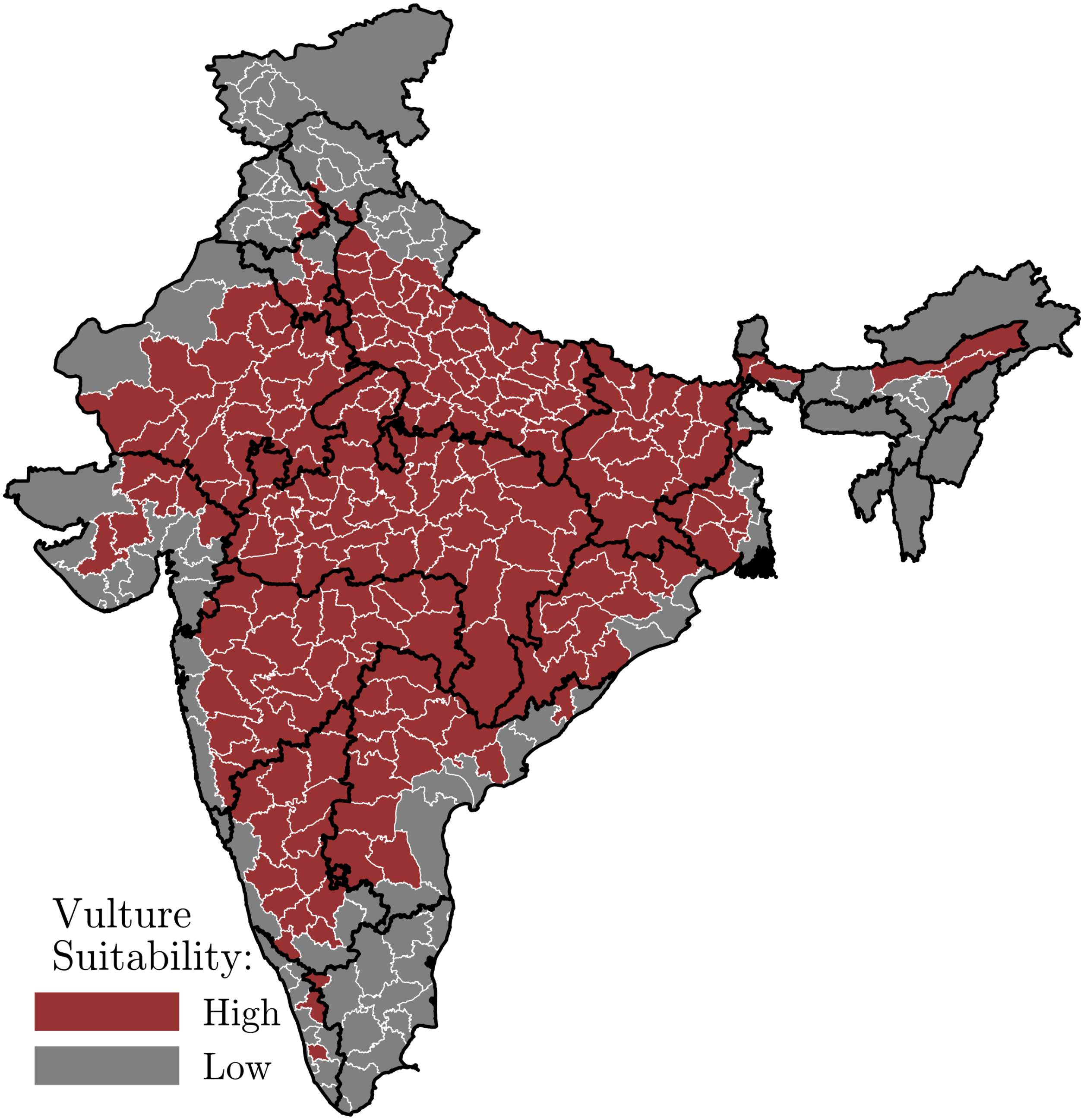
———. 1998. "The Noah's ark problem." *Econometrica: journal of the Econometric Society,* 1279–1298.

———. 2009. "On Modeling and Interpreting the Economics of Catastrophic Climate Change." *The Review of Economics and Statistics* 91 (1): 1–19.

Winfree, Rachael, Neal M Williams, Jonathan Dushoff, and Claire Kremen. 2007. "Native bees provide insurance against ongoing honey bee losses." *Ecology letters* 10 (11): 1105–1113.

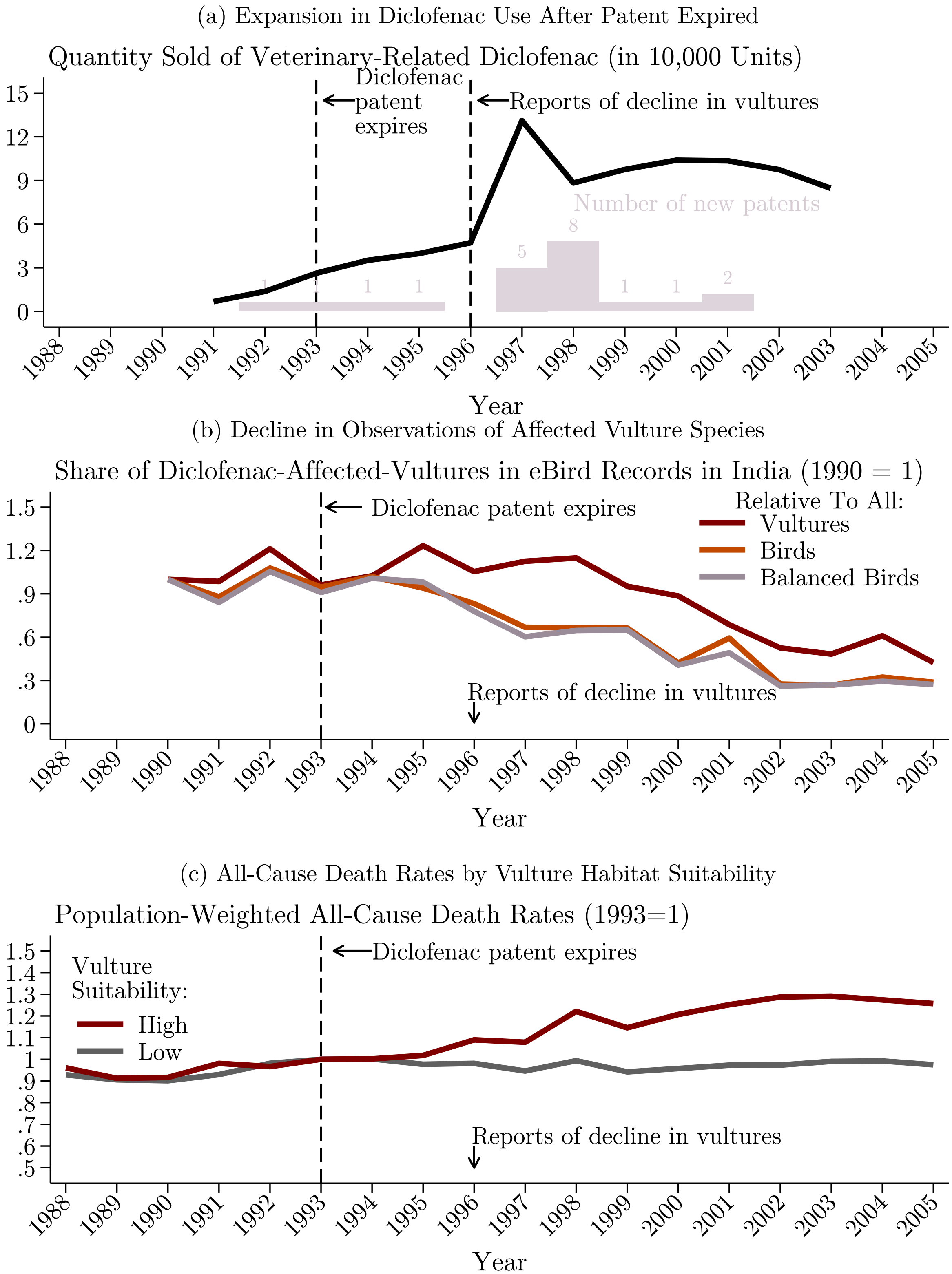
Young, Alwyn. 2019. "Channeling Fisher: Randomization Tests and the Statistical Insignificance of Seemingly Significant Experimental Results." *The Quarterly Journal of Economics* 134 (2): 557–598.

Figure 1: Spatial Distribution of Diclofenac-Affected-Vulture Ranges



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

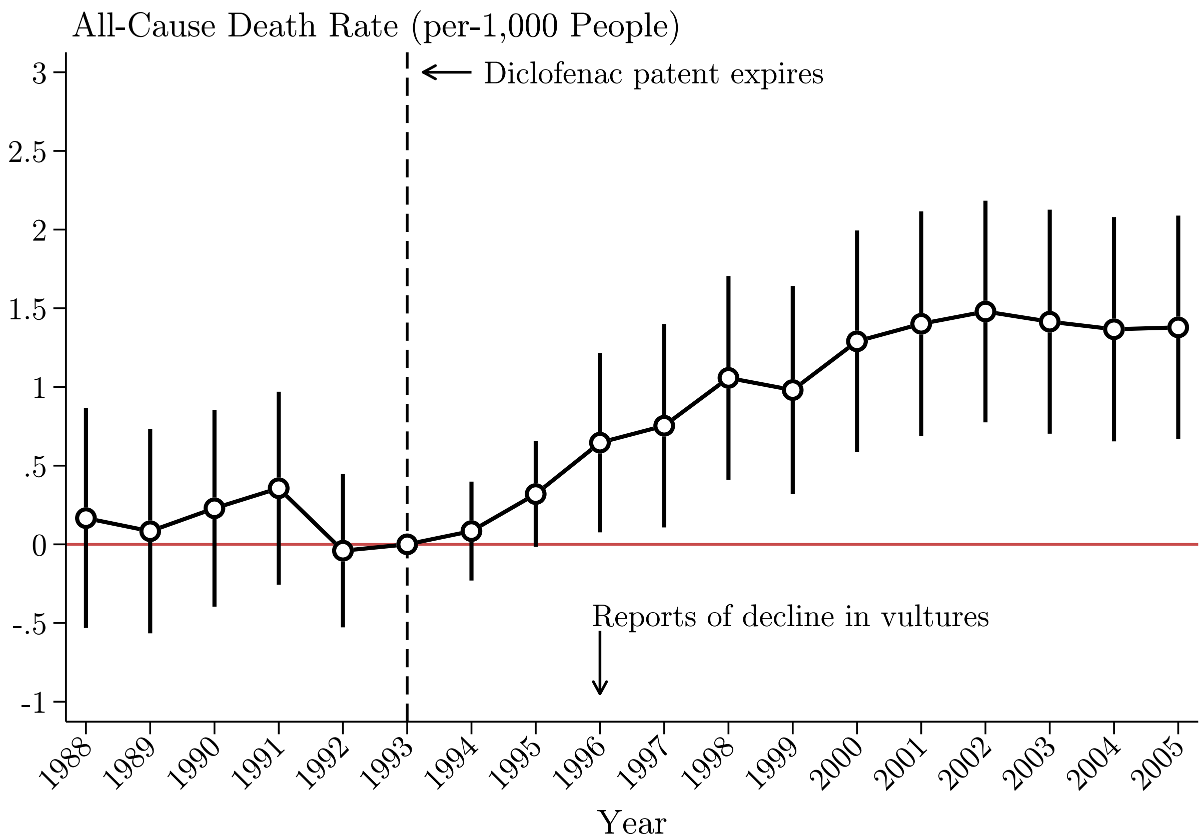
Figure 2: National Trends in Diclofenac Use, Vulture Observations & Death Rates



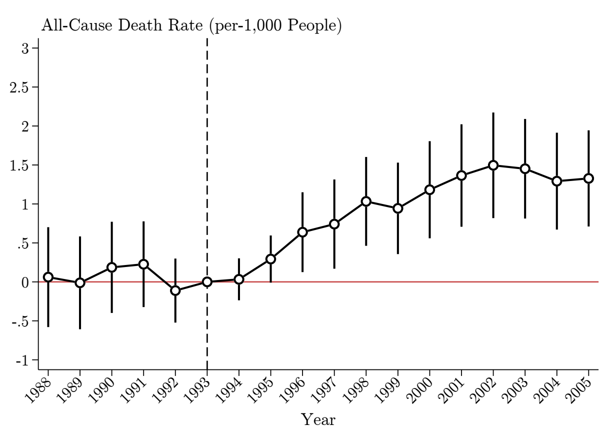
Notes: (a) Veterinary-related diclofenac sales, and the number of new product entries. (b) The share of diclofenac-affected-vultures relative to all other vultures species, all bird species, and all bird species that are consistently reported every year. (c) Mean all-cause death rates (balanced and not residualized) by vulture suitability classification for diclofenac-affected-vultures.

Figure 3: All-Cause Death Rates DD Estimation Results

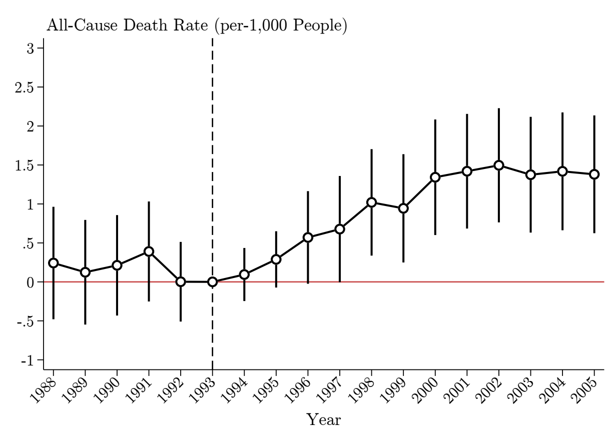
(a) Combined Urban & Rural Areas



(b) Urban Areas

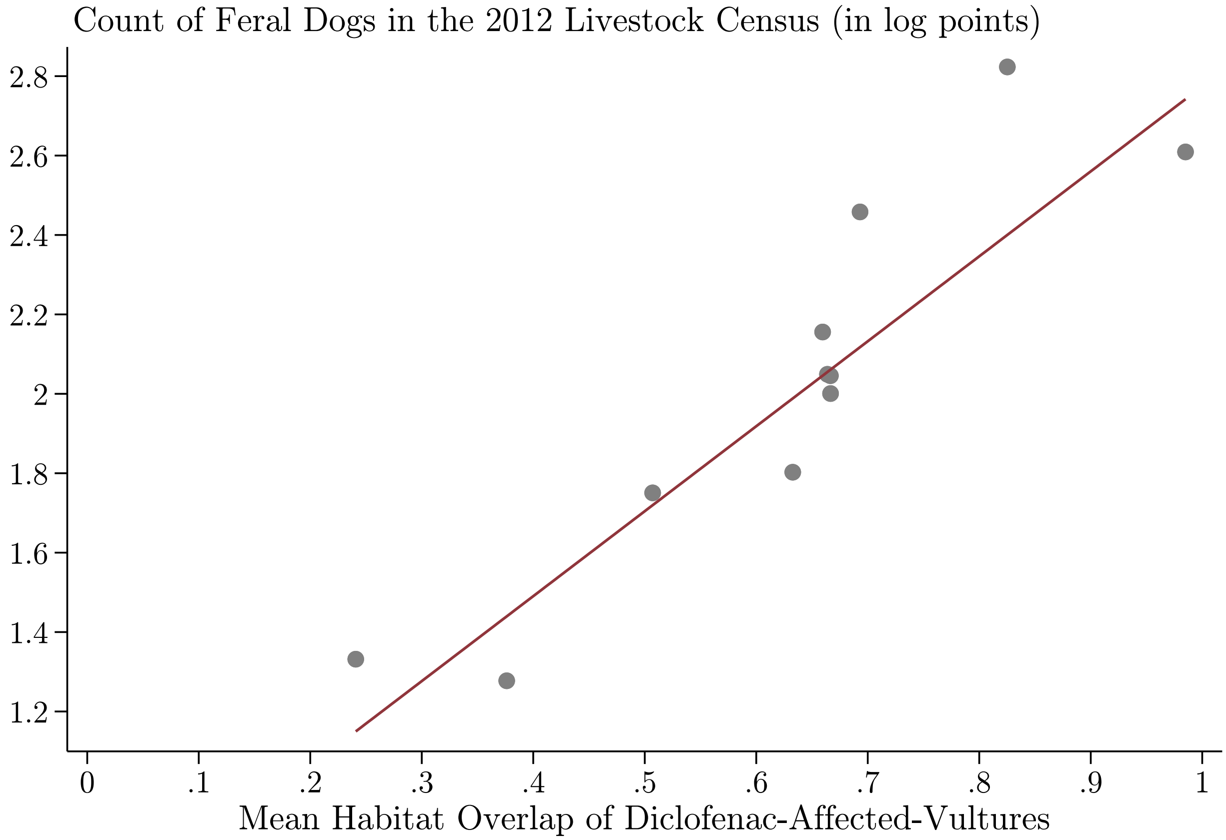


(c) Rural Areas



Notes: Estimation results from Equation (1). Comparing the third and second terciles to the first tercile of vulture habitat overlap. All regression include district and zonal council-by-year fixed effects. Observations are population-weighted. Standard errors are clustered at the district level.

Figure 4: Feral Dogs (2012) VS. Vulture Suitability



Notes: (a) National level data on all rabies vaccines sold from 1991 to 1997.(b) District-level data on feral dogs as counted for the first time during the 2012 livestock census.

Table 1.
Differences in Observables Prior to Vulture Collapse

	(1)	(2)	(3)	(4)
	Group Means		Difference	N
	Low Vulture Suitability	High Vulture Suitability		
Death Rate, per-1,000 people (1988-1993)	5.3	4.2	-1.2	153
	(1.8)	(1.8)	(.32)	
Degree Days Above 30°C (1988-1993)	54	66	12	153
	(43)	(35)	(6.8)	
Precipitation in mm·km^{-2} (1988-1993)	.25	.12	-.12	153
	(.42)	(.18)	(.044)	
Number of Livestock (1987, 1992)	1,632	1,615	-17	138
	(874)	(731)	(158)	
Log(Dissolved Oxygen) (1988-1993)	1.9	1.9	.0045	95
	(.18)	(.27)	(.047)	
Log(Fecal Coliform) (1988-1993)	7.2	7.4	.25	76
	(2.2)	(1.7)	(.48)	
Pop. Share [1, 24] (1991)	.43	.51	.086	142
	(.14)	(.08)	(.023)	
Pop. Share [25. 54] (1991)	.3	.33	.028	142
	(.095)	(.058)	(.016)	
Pop. share [55, 100] (1991)	.085	.088	.0034	142
	(.029)	(.018)	(.0048)	
Share Literate (1991)	.55	.41	-.14	140
	(.13)	(.12)	(.022)	
Water Taps per-100,000 People (1991)	12	13	.84	141
	(28)	(21)	(2.8)	
Water Wells per-100,000 People (1991)	24	57	33	141
	(25)	(42)	(6.1)	
Hospitals & Health Centers per-100,000 People (1991)	1.7	2.4	.66	141
	(1.7)	(2.5)	(.35)	
Doctors & Health Workers per-100,000 People (1991)	8.6	9.8	1.2	141
	(7.6)	(8.6)	(1.6)	

Notes: Mean baseline levels of observable characteristics of districts by vulture suitability classification. Column 3 reports the difference between the high (treatment) and low (control) vulture suitability districts. Sample consists of districts with balanced all-cause death rate data for 1988 to 2005. Observations are population-weighted. Robust standard errors are reported in paranthesis.

Table 2.
All-Cause Death Rate, per-1,000 People ($\overline{Y}_{1993} = 9.2$)

Panel A. Without Weather Controls

	(1)	(2)	(3)	(4)	(5)	(6)
HVS×Post-1994	0.81	0.91				
	(0.17)	(0.16)				
HVS×[1994, 1999]			0.52	0.13	0.21	0.52
			(0.16)	(0.18)	(0.17)	(0.16)
HVS×[2000, 2005]			1.26	0.48	0.40	1.26
			(0.22)	(0.20)	(0.21)	(0.22)
R^2	0.715	0.740	0.746	0.773	0.804	0.738
N	2,754	2,754	2,754	2,754	2,700	2,700
Clusters	153	153	153	153	150	150

Panel B. With Weather Controls

	(1)	(2)	(3)	(4)	(5)	(6)
HVS×Post-1994	0.76	0.85				
	(0.17)	(0.16)				
HVS×[1994, 1999]			0.51	0.18	0.19	0.51
			(0.16)	(0.18)	(0.17)	(0.16)
HVS×[2000, 2005]			1.17	0.45	0.38	1.17
			(0.21)	(0.20)	(0.21)	(0.21)
R^2	0.725	0.747	0.751	0.778	0.807	0.743
N	2,754	2,754	2,754	2,754	2,700	2,700
Clusters	153	153	153	153	150	150
Year FE	X					
Zonal Council-by-Year FE		X	X	X		X
State-Linear Trends				X		
State-by-Year FE					X	

Notes: Estimation results for the specification in Equation (2). The estimation is comapring high-vulture-suitability (HVS) to low-vulture-suitability, after the onset of diclofenac use (post-1994), relative to years prior to the patent expiration. Results in columns 3 to 7 split the post-1994 period to two periods: 1994 to 1999, and 2000 to 2005. When we include state-by-year fixed effects (column 5), three states get dropped as they have no district-level data. In column 6, we repeat the estimation with zonal council-by-year fixed effects for the sample in column 5. The reported mean death rate of 9.2 for 1993 is the nationally representative mean we obtain from the SRS data. Sample includes balanced district-level data from 1988 to 2005. All regressions include district fixed effects. Observations are population-weighted. Standard errors are clustered at the district level.

Table 3
Livestock DDD Results for All-Cause Death Rate ($\overline{Y}_{1993} = 9.2$)

	DD	Above Median Livestock in Levels			Above Median Livestock Density		
	(1)	(2)	(3)	(4)	(5)	(6)	(7)
HVS×Livestock×Post-1994		0.56	0.77	0.73	0.48	0.82	0.80
		(0.37)	(0.31)	(0.31)	(0.40)	(0.33)	(0.33)
HVS×Diclofenac	0.75	0.37	0.31	0.31	0.45	0.28	0.27
	(0.18)	(0.30)	(0.25)	(0.25)	(0.33)	(0.27)	(0.27)
Livestock×Post-1994		0.05	-0.10	-0.08	-0.33	-0.63	-0.59
		(0.27)	(0.24)	(0.24)	(0.31)	(0.26)	(0.26)
Year FE	X	X			X		
Zonal Council-by-Year FE			X	X		X	X
Weather Controls				X			X
R^2	0.73	0.73	0.77	0.77	0.73	0.77	0.77
N	2,466	2,466	2,466	2,466	2,466	2,466	2,466
Clusters	137	137	137	137	137	137	137

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. We classify districts as high livestock at baseline using either the total number of livestock animals (columns 2 to 4), or the normalizing it by the area of the district (columns 5 to 7). Because we have data for 137 of the 153 districts, we repeat the DD estimation for the sample with non-missing livestock data (column 1). Sample includes balanced district data, combining urban and rural areas, from 1988 to 2005. The reported mean death rate of 9.2 is the nationally representative mean we obtain from the SRS data. All regressions include district fixed effects. Observations are population-weighted. Standard errors are clustered at the district level.

Table 4
District Water Quality DD & DDD Estimates

	Log(Dissolved Oxygen)			Log(Fecal Coliforms)		
	U&R		U	U&R		U
	(1)	(2)	(3)	(4)	(5)	(6)
HVS×Urban×Diclofenac		-0.121			1.210	
		(0.051)			(0.479)	
HVS×Diclofenac	0.004	0.046	-0.074	0.288	-0.110	1.142
	(0.026)	(0.031)	(0.041)	(0.351)	(0.452)	(0.341)
Urban×Diclofenac		0.063			-0.422	
		(0.047)			(0.361)	
$\overline{\text{Dep. Var}}_{\leq 1993}$	1.92	1.92	1.89	6.86	6.86	6.85
R^2	0.706	0.707	0.743	0.779	0.780	0.828
N	4,349	4,349	2,073	3,344	3,344	1,578
Clusters	220	220	139	200	200	120

Notes: Estimation results for the specification in Equation (2). Comapring the third and second tercile of diclofence affected vultures to first tercile, before and after the onset of diclofenac use. 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, etc.). In addition, each regression includes year fixed effects. Sample consists of district-level data for urban (U) and rural (R) areas, from 1988 to 2004. Observations are population-weighted. Standard errors are clustered at the district level.

Appendix

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 to the eBird 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 roads 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.

A.2 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 is an abrupt shift in the reporting regime in 1988 where the vital statistics start reporting vital event counts instead of rates. As this allows to fully control the conversion to rates, we prefer to use data reported under the same regime. Second, the number districts that are fully balanced from 1988 to 2005 are 153, while there are only 101 balanced districts for the 1981 to 2005 period. When extending the panel to the full 1981 to 2005 period, and losing about 35% percent of the districts, we recover similar results to those in the main text (Figure A2). Specifically, we do not observe a differential time trend in the years leading 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.3 Accounting for State-Level Temporal Trends

To account for potential differential trends in reporting of vital statistics data that systematically change by state, we repeat the estimation in Equation (1) and include either 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 A3 to those in Figure 3. 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 are 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.

A.4 Addressing Sporadic Missing Vital Statistics Data Using Long-Differences

In the main analysis, we use a fully balanced sample of 153 districts that report their all-cause death rate every year during 1988 to 2005. Because reporting of vital statistics data is often incomplete, requiring full reporting results in excluding many districts from the sample even if they have data during both pre- and post-treatment periods. At times, some states have missing data for all their districts during the middle of the sample, while in other cases the last year or two are missing, or there is a year or two missing earlier in the sample.

In order to increase the representativeness of the sample, by including more districts, we calculate the mean all-cause death rate during two periods: 1988 to 1993 (pre-treatment), and 2000 to 2005 (post-treatment). We use the collapsed data as part of a long differences analysis where we are comparing the two time periods across the high and low vulture suitability districts (Burke and Emerick 2016). This allows us to go from 153 fully balanced districts to 285 districts that have at least a few years of non-missing mortality data in both time periods.

Including more districts in the analysis recovers a qualitatively similar estimate, albeit smaller in magnitude. In Table A1, we report the results from the long-differences estimation for both the DD and DDD designs. In Panel A, we report the results for the the sample that combines the data from urban and rural areas and uses district-level observations. In the DD estimation, for the districts that have fully balanced death rate data from 1988 to 2005, we find a larger effect using long-differences than when using the annual panel. Specifically, when using the annual panel, we estimate an increase, on average, of 0.91 deaths per-1,000 people (Table 2, Panel A, column 3), but when collapsing the data to pre- and post-vulture collapse, we estimate an increase of 1.24 additional deaths per-1,000 people (Table A1, Panel A, column 1). We estimate a smaller increase in death rates of 0.61 when expanding the sample to include even non-fully balanced districts, going from 153 to 285 districts (Table A1, Panel A, column 3). Even though the estimated effect when using the larger sample is half the size of the coefficient when using the fully balanced sample, it is still precisely estimated and reflects a meaningful increase in mortality relative to the nationally representative mean of 9.2 deaths per-1,000 people.

We also repeat the DDD estimation where we examine whether high livestock at baseline results in a higher increase in death rates. Using the collapsed data, we find a differential increase in death rates of 0.88 deaths per-1,000 people in the high livestock at baseline districts (Table A1, Panel A, column 2), similar to the increase of 0.82 deaths per-1,000 people when using the annual sample (Table 3, column 6). Expanding the sample increases the number of districts from 137 to 235, but recovers an imprecisely estimated increase of 0.41 deaths per-1,000 people in the high livestock at baseline group (Table A1, Panel A, column 4).

Finally, we use the data as reported by urban and rural areas. As in the main analysis, for the balanced sample, we require that each district has non-missing data for both the urban and rural areas for every year during 1988 to 2005. For the expanded sample, we relax that requirement, allowing districts to be either missing for both their urban and rural areas for a few years, or be missing for one of the areas, as long as they report data during both the pre- and post-vulture collapse period. For the DD analysis, we find a similar pattern, where the long-difference estimation using the balanced sample recovers an estimate double the size relative to the the expanded sample (Table A1, Panel B, columns 1 and 3). When comparing the urban to rural areas in the fully balanced sample, we find an even smaller differential of 0.12 additional deaths per-1,000 people (Table A1, Panel A, column 2). However, in the expanded sample, the urban areas have a higher death rate, on average, of 1.15 deaths per-1,000 people. This suggests that in the fully balanced sample, the difference between urban and rural areas is much smaller than the difference across those areas in the districts that get added to the sample.

A.5 Examining Heterogeneity Between Urban & Rural District Areas

In Table A2, we explore the degree to which death rates respond differently to the collapse in vulture populations in either urban or rural areas. Because urban areas are denser, and more likely to have an animal dumping site at their outskirts, we expect that a larger portion of the average effect we estimate in Table 2 is driven by the urban areas. When we use the district-level data reported by urban or rural area, we find a higher average treatment effect in urban relative to rural areas, but the effects are not statistically different from each other.

A.6 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 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 the flat plains of the midwest.

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 consturct 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 scores.

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 A4, and the average treatment effects in Table A3. Across the two alternative treatment classification schemes, we recover similar magnitudes for the change in 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.7 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 4), 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 capture both organic and inorganic uses of oxygen. We should expect to see both BOD and COD levels increase with a 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, turbidty goes up. 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 A4, 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 urban district (columns 2, 3, 5, and 6), similar to how the decline in dissolved oxygen and increase in fecal coliforms was as well (see Table 4). Turbidity declines in water bodies monitored in urban districts (columns 8 and 9), which is consistent with previous findings on declines in scavenger populations.

A.8 Sensitivity Analysis Using Jackknifing

In our analysis, 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 A5 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 A3.

A.9 Permutation Inference Analysis

As an additional robustness test we also run a permutation inference analysis. Using permutation inference analysis allows us to evaluate whether we are underestimating the standard errors of the coefficients by clustering at district level (e.g. due to spatial clustering of the standard errors), as well as ruling 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 A7, where each one of the distributions is centered around zero. More importantly, the estimated effect from the non-permutation sample is in far right tail of each distribution, resulting in an exact p-value well below 1%.

A.10 Estimating the Effect of Weather on Mortality

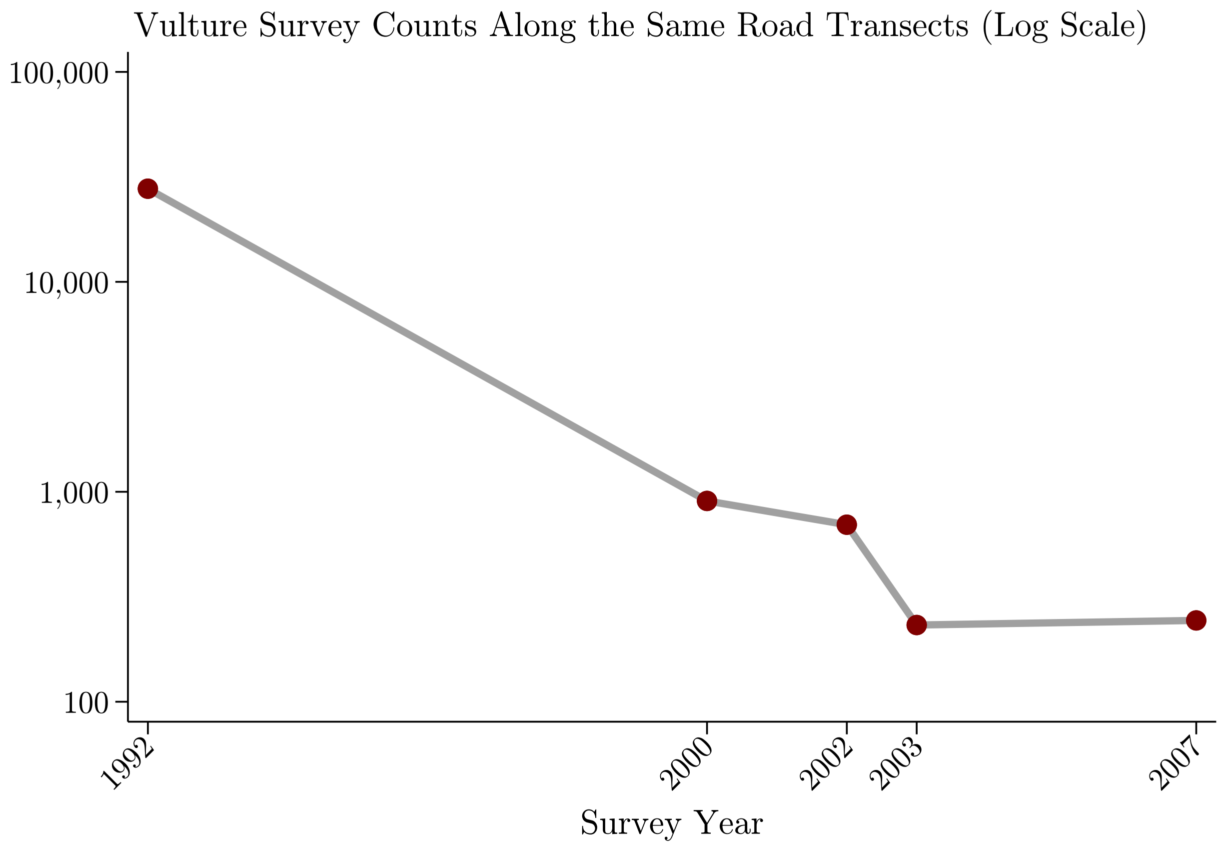
To better interpret the results from our main analysis, we estimate a damage function for weather on mortality. These results allow us to anchor the magnitudes of our estimates to other drivers of mortality, and to compare them with previous results in the literature on environmental quality and health. We follow a similar estimation strategy to that in Burgess et al. (2017), and use degree-day bins along with quantiles for precipitation. To focus on the long-term effect of weather on mortality, net of adaptation, we use the same long-differences approach as discussed earlier in the Appendix (Burke and Emerick 2016). Explicitly, we estimate the following specification:

$$y_{dzt} = \sum_{i=1}^{13} \theta_i (\text{Degree Day Bin})_{idt} + \sum_{j=2}^{5} \omega_j \mathbb{1}\{(\text{Precipitation in Quantile j})_{dt}\} + \lambda_d + \delta_{zt} + \varepsilon_{dzt} \tag{4}$$

Where y_{dzt} is the combined urban and rural death rate, averaged over the two periods: 1988 to 1993, and 2000 to 2005. We include 13 degree day bins, from below 10 degrees Celsius, 10 to 12 degrees, 12 to 14, all the way up to 32 to 34 degrees, and above 34 degrees Celsius. We use the bin of of 22 to 24 as the reference category. We divide the total precipitation into quintiles, and use the first one as the reference category. We include district and zonal council-by-period fixed effects, and cluster the standard errors at the district level.

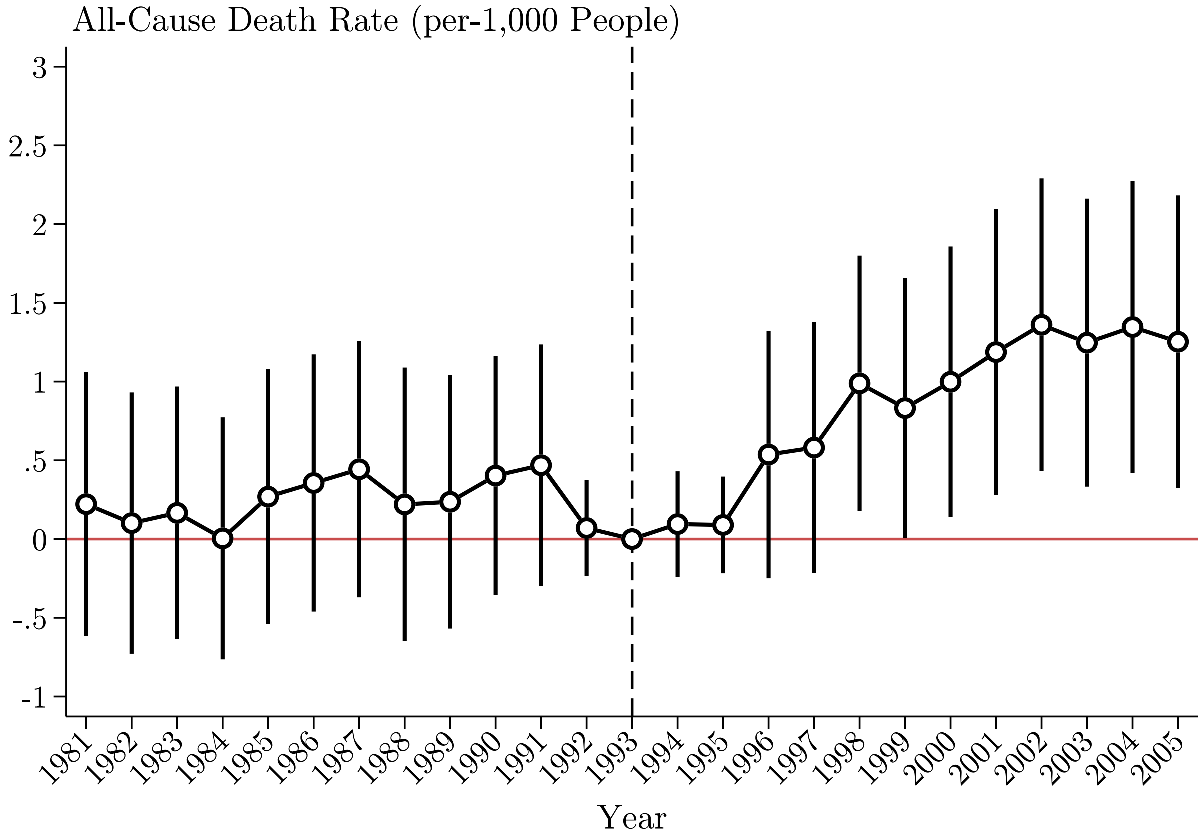
In Figure A8, we report results for the fully balanced sample used in the main analysis, as well as the full sample with all districts that report data at least for one year during the pre- and post-periods. For the fully balanced sample, one additional day above 34°C, relative to the reference category of 22-24°C, results in 0.23 additional deaths per-1,000 people. This means that about three to four additional hot days a year will lead to an increase in mortality similar to that of the average increase in the death rate following the collapse of the vulture populations of 0.85 deaths per-1,000 people (Table 2, Panel B, column 3). If we expand the sample to include more than just the fully balanced districts, we estimate that an additional day above 34°C, relative to the reference category of 22-24°C, results in 0.15 additional deaths per-1,000 people. Even with this lower estimate, a year with six more hot days will lead to an increase of 0.9 deaths per-1,000 people. In short, the increase in mortality we find following the collapse of vulture populations is on par with a week-long heatwave in India.

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).

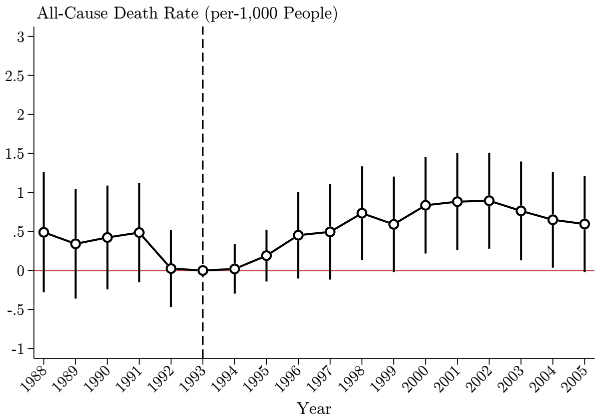
Figure A2: All-Cause Death Rates DD Estimation Results With Earlier Years



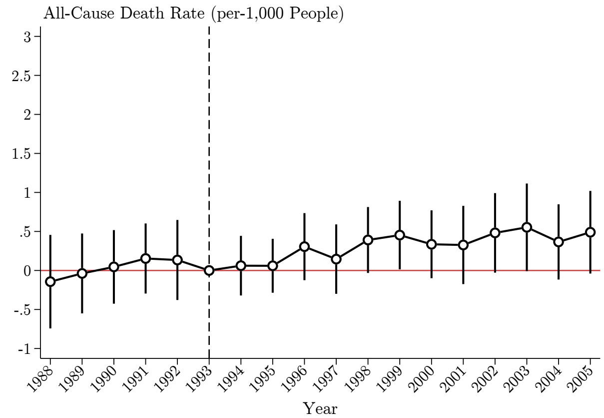
Notes: Estimation results from Equation (1). Comparing the third and second terciles to the first tercile of vulture habitat overlap. The regression includes district and zonal council-by-year fixed effects. Observations are population-weighted. Standard errors are clustered at the district level.

Figure A3: All-Cause Death Rates DD Estimation Results With State-Level Trends

(a) State-Linear Time Trends



(b) State-by-Year Fixed Effects

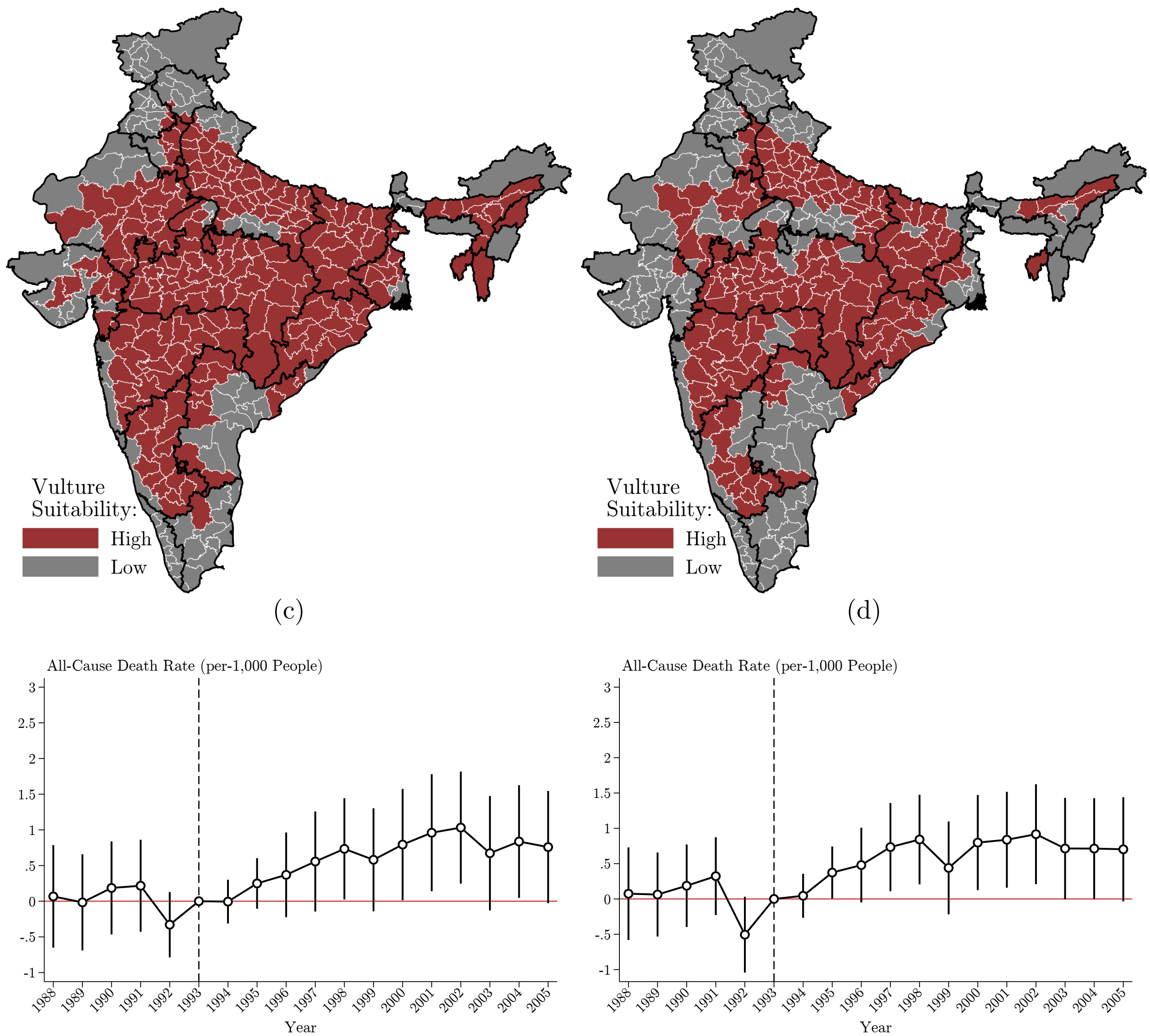


Notes: Estimation results from Equation (1). Comparing the third and second terciles to the first tercile of vulture habitat overlap. All regression include district fixed effects. The regression in (a) includes zonal council-by-year fixed effects and state-level linear time trends, and the regression in (b) includes state-by-year fixed effects. Observations are population-weighted. Standard errors are clustered at the district level.

Figure A4: Classifying Treated Districts Using the BIOCLIM Habitat Suitability Model

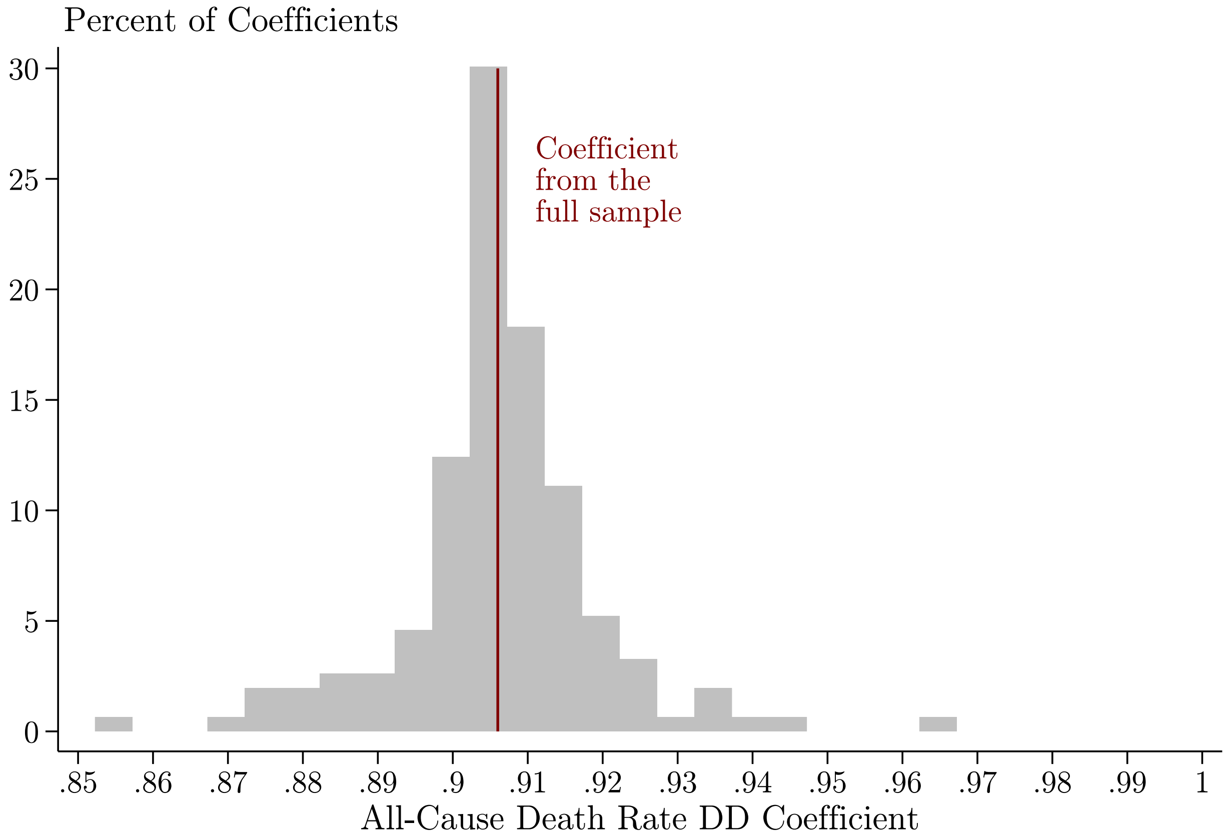
Suitability Score Terciles III & II (a)

Suitability Score Above Median (b)



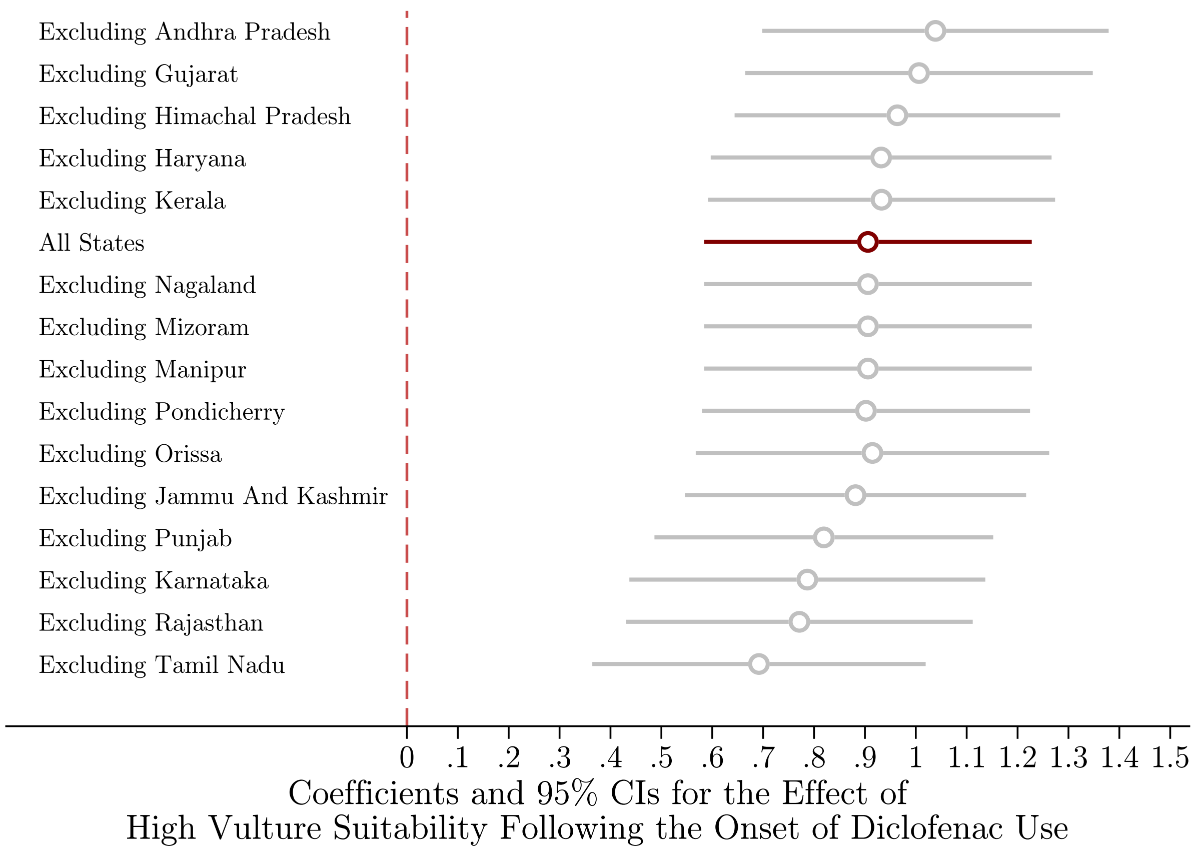
Notes: The treatment classification uses predicted suitability scores for the diclofenac-affected-vultures from the BIOCLIM habitat suitability model. We either split the suitability score into terciles and define treated districts as the third and second terciles (a and c), or split districts as above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

Figure A5: 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.

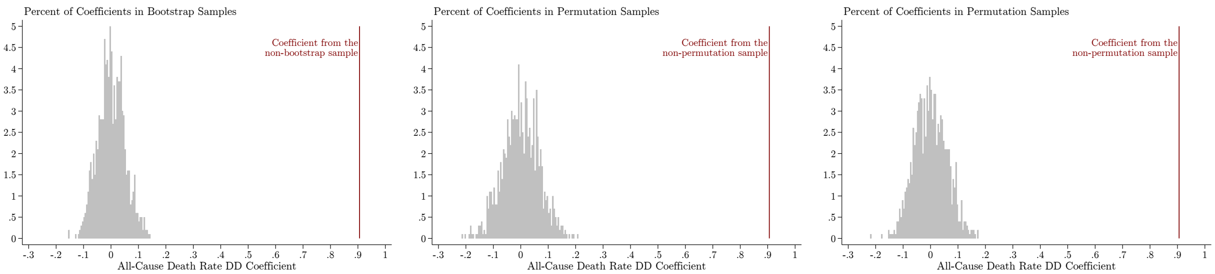
Figure A6: Distribution of Leave-One-State Out DD Estimation Results



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 A7: Permutation Inference DD Estimation Results

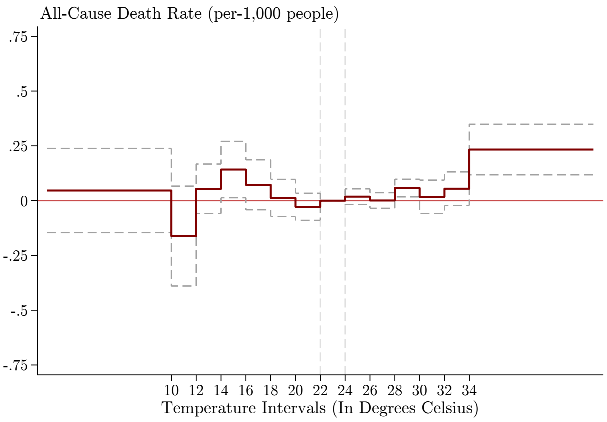
(a) Full (b) Block (c) Within



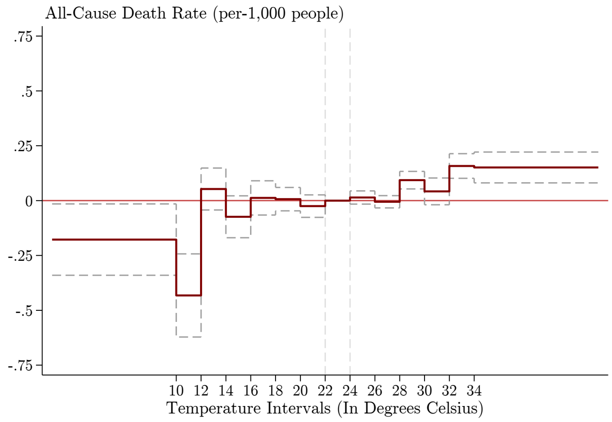
Notes: Distribution of coefficients from permutation samples where the treatment is randomly assigned. The vertical line shows the estimated coefficient from the non-permutation sample.

Figure A8: Weather & Mortality Damage Functions

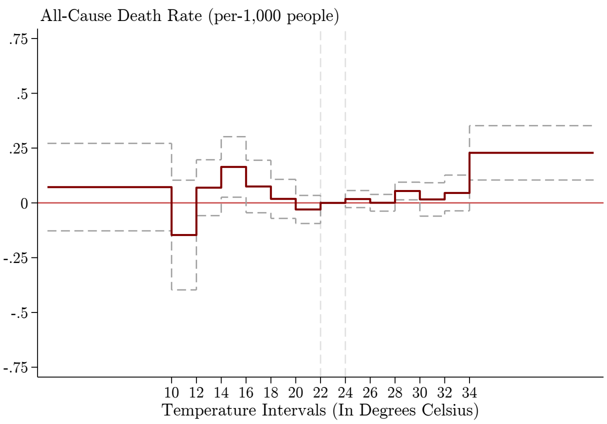
Fully Balanced Sample
(a) Without Precipitation Controls



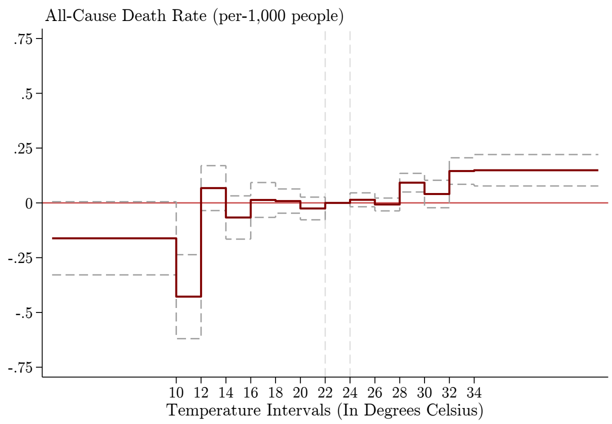
Expanded Sample
(b) Without Precipitation Controls



(c) With Precipitation Controls



(d) With Precipitation Controls



Notes: Estimation results from Equation (4). Observations are population-weighted. Standard errors are clustered at the district level. See Table A1 for additional details.

Table A1.
All-Cause Death Rate Long-Differences Estimation Results

	Fully Balanced Data Sample		Reported Data in Pre/Post	
Panel A. Combined Urban & Rural Areas				
	(1)	(2)	(3)	(4)
HVS×Livestock×Diclofenac		0.88		0.41
		(0.43)		(0.35)
HVS×Diclofenac	1.24	0.51	0.61	0.65
	(0.22)	(0.34)	(0.23)	(0.29)
Livestock×Diclofenac		-0.04		-0.10
		(0.33)		(0.30)
R^2	0.866	0.874	0.881	0.904
N	306	274	570	470
Clusters	153	137	285	235
Panel B. District-by-Area				
	(1)	(2)	(3)	(4)
HVS×Urban×Diclofenac		0.12		1.15
		(0.38)		(0.44)
HVS×Diclofenac	1.29	1.24	0.65	0.29
	(0.27)	(0.32)	(0.22)	(0.24)
Urban×Diclofenac		-0.31		-0.67
		(0.29)		(0.25)
R^2	0.862	0.862	0.866	0.869
N	456	456	1,108	1,108
Clusters	114	114	282	282

Notes: Estimation results for the specifications in Equations (2) and (3). 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 (1998 to 1993). We calculate the mean all-cause death rate in each period. Columns 1 and 2 report the results for the fully balanced districts, which we use in the main analysis. Columns 3 and 4 extend the sample by including all districts that report data for at least one year in both the pre- and post-periods. Panel A uses the data at the district-level, combining the urban and rural areas. Panel B use the data at the district-area level. All regresssions include zonal council-by-year fixed effects. Each regression in Panel A includes district fixed effects, while each regression in Panel B includes district-by-area fixed effects. Observations are population-weighted. Standard errors are clustered at the district level.

Table A2.
All-Cause Death Rate, per-1,000 People ($\overline{Y}_{1993} = 9.2$)

Panel A. District Urban Areas				
	(1)	(2)	(3)	(4)
HVS×Post-1994	0.88	0.84	0.95	0.91
	(0.19)	(0.18)	(0.17)	(0.17)
R^2	0.703	0.712	0.728	0.734
N	5,562	5,562	5,562	5,562
Clusters	156	156	156	156
Panel B. District Rural Areas				
	(1)	(2)	(3)	(4)
HVS×Post-1994	0.76	0.71	0.86	0.79
	(0.17)	(0.17)	(0.16)	(0.16)
R^2	0.715	0.723	0.735	0.742
N	5,670	5,670	5,670	5,670
Clusters	162	162	162	162
Year FE	X	X		
Zonal Council-by-Year FE			X	X
Weather Controls		X		X

Notes: Estimation results for the specification in Equation (2). The estimation is comapring high-vulture-suitability (HVS) to low-vulture-suitability, after the onset of diclofenac use (post-1994), relative to years prior to the patent expiration. The reported mean death rate of 9.2 for 1993 is the nationally representative mean we obtain from the SRS data. Sample includes balanced district level data from 1988 to 2005. All regressions include district fixed effects. Observations are population-weighted. Standard errors are clustered at the district level.

Table A3

Results for All-Cause Death Rate Using BIOCLIM Classifications ($\overline{Y}_{1993} = 9.2$)

Panel A. High & Medium Suitability Score Terciles	(1)	(2)	(3)	(4)
HVS×Livestock×Post-1994			0.830	0.751
			(0.343)	(0.336)
HVS×Diclofenac	0.622	0.539	-0.008	0.001
	(0.189)	(0.183)	(0.251)	(0.252)
Livestock×Post-1994			-0.208	-0.173
			(0.266)	(0.258)
R^2	0.734	0.741	0.761	0.767
N	2,754	2,754	2,466	2,466
Clusters	153	153	137	137

Panel B. Above Median Suitability Score	(1)	(2)	(3)	(4)
HVS×Livestock×Post-1994			0.908	0.885
			(0.335)	(0.329)
HVS×Diclofenac	0.622	0.553	-0.129	-0.143
	(0.184)	(0.178)	(0.258)	(0.260)
Livestock×Post-1994			-0.034	-0.035
			(0.220)	(0.214)
R^2	0.733	0.741	0.761	0.768
N	2,754	2,754	2,466	2,466
Clusters	153	153	137	137
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. The reported mean death rate of 9.2 is the nationally representative mean we obtain from the SRS data. All regressions include district and zonal council-by-year fixed effects. Observations are population-weighted. Standard errors are clustered at the district level.

Table A4
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×Diclofenac		1.4			10.9			-5.8	
		(1.1)			(7.2)			(6.7)	
HVS×Diclofenac	0.6	0.2	1.6	1.6	-2.2	9.0	-0.1	1.4	-5.0
	(0.5)	(0.5)	(1.1)	(3.1)	(2.1)	(7.1)	(3.9)	(4.4)	(6.5)
Urban×Diclofenac		-0.5			-6.3			-1.1	
		(0.7)			(6.3)			(4.2)	
$\overline{\text{Dep. Var}}_{\leq 1993}$	4.01	4.01	5.03	25.32	25.32	28.61	36.44	36.44	40.30
R^2	0.738	0.739	0.753	0.709	0.710	0.752	0.788	0.788	0.782
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 the specification in Equation (2). Comapring the third and second tercile of diclofence affected vultures to first tercile, before and after the onset of diclofenac use. 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, etc.). In addition, each regression includes year fixed effects. Sample consists of district-level data for urban (U) and rural (R) areas, from 1988 to 2004. Observations are population-weighted. Standard errors are clustered at the district level.

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 survey 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. Summary of the responses reported a median onset year for diclofenac of 1994.

Figure B1: FDA Documents Regarding Diclofenac & Generic Drug Approval

	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

(a) Change to Novartis' Diclofenac Patent in 1993

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.

(b) Change Code CRLD

FDA

Guidance Purpose and Goals

- To help applicants submitting an abbreviated new drug application (ANDA) to seek approval of a generic drug to identify:
  - A reference listed drug (RLD), i.e., a previously approved drug product for which an applicant seeks approval of a generic drug;
  - a reference standard, i.e., the previously approved drug selected by FDA that an applicant must use in conducting any in vivo bioequivalence testing required to support approval of its ANDA; and
  - the basis of submission for the ANDA.

www.fda.gov 2

(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.[16] 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.[17]

C.2 Examining the Reporting Accuracy of the CRS Data

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 (dashed lines) between the all-cause death rate in the CRS relative to the SRS data. The SRS death rate is nearly double that the CRS reported death rate. However, when residualizing the death rates on a set of unit and time fixed effects (reported in the solid lines), the two death rates follow similar trends.[18]

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

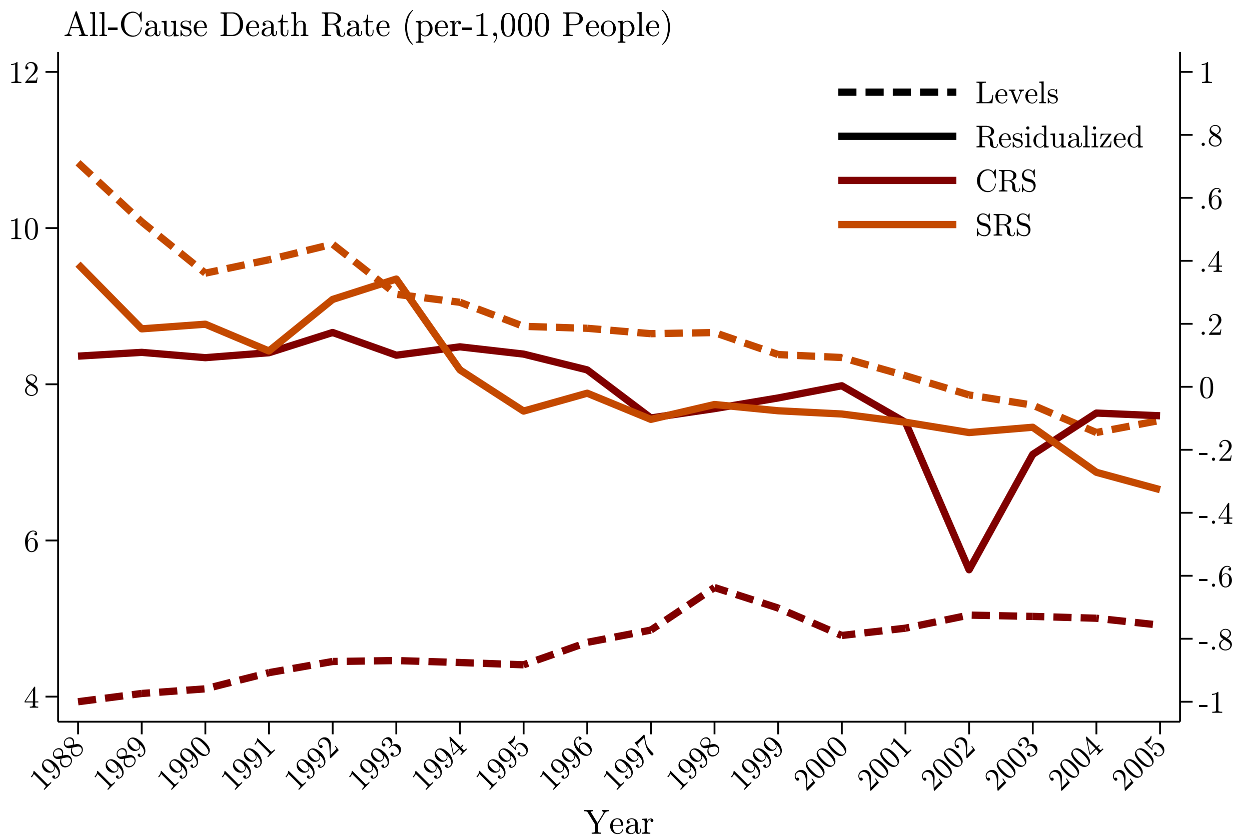
[16] Application can be filled out at: http://datazone.birdlife.org/species/requestdis

[17] BLI provides a summary of these categories here: http://datazone.birdlife.org/species/spcdistPOS

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

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. Throughout the analysis, we present the mean mortality level from the CRS, to reflect the data in the estimation sample, but interpret the magnitude of the coefficients relative to the mean level from the SRS data, which reflect the national-level death rate.

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.