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

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Abstract

Scientific evidence documents an ongoing mass extinction of species, caused by human activity. Allocating conservation resources is difficult due to scarce evidence on the damages from losing specific species. This paper studies the collapse of vultures in India, triggered by the expiry of a patent on a painkiller. Our results suggest the functional extinction of vultures — efficient scavengers who removed carcasses from the environment — increased human mortality by over 4% because of a large negative shock to sanitation. These effects are comparable to estimates of heat deaths from climate change. We quantify damages at $69.4 billion per-year.

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1 Introduction

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

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

These facts set the stage for a thorny policy challenge. Wildlife levels can collapse quite rapidly, with trajectories that are difficult to predict or reverse. Curtailing or regulating economic activity, or investing in conservation initiatives, might protect or restore some species populations. Nevertheless, although biodiversity loss is arguably damaging in general (Cardinale et al. 2012), evidence regarding the effects of losing *specific* species on human well-being is sparse. Unfortunately, without such evidence it is hard to decide where to target conservation efforts. Policymakers are instead left in the undesirable situation of being potentially subject to multiple unpredictable shocks (collapse of a species), with little sense of the sign or magnitude of their costs on society.[1]

Economic theory has long recognized the conceptual and practical difficulties involved in carrying out a forward looking cost-benefit analysis in the presence of uncertainty, irreversibility, and catastrophic tail risks (Arrow and Fisher 1974; Weitzman 2009). The costs of species extinction are hard to estimate for several reasons. First, the effect of a catastrophic collapse cannot in general be recovered by studying the impact of marginal changes. Second, causal evidence is hard to produce because we often possess very little data on species population counts and experimental estimates are unavailable because manipulating ecosystems can be both unethical and infeasible (Frank and Schlenker 2016; Ferraro et al. 2019). Third, the number of potentially endangered species is large forcing us to target both evaluation and conservation efforts.

In this paper, we study the sudden and catastrophic collapse of vulture populations across the Indian subcontinent, making progress on all three fronts. First, we use a local functional extinction to study the costs on society of a catastrophic collapse of vultures in India, because of the loss of the sanitation services that these birds had provided through scavenging dead livestock. We provide evidence of a meaningful increase in human mortality after vul-

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

tures died out and were no longer removing carcasses from the environment. Although this analysis is retrospective, local functional extinctions are more easily reversed than global extinction in the wild, enabling evidence of this type to constructively influence conservation policy in extinction areas, and protection of vultures in parts of the world where they still provide scavenging services.

Second, we overcome the causal inference challenges associated with estimating social costs by drawing upon empirically and theoretically grounded measures of habitat suitability developed by ecologists. Specifically, we use a differences-in-differences approach comparing changes in mortality in areas with habitats that had high vs low vulture-suitability, before and after a near-total decline in bird populations due to an unintentional, unexpected, and rapid poisoning event. Habitat definitions in this setting provide an indicator for regions where the population change is expected to have been large. We find that districts that were highly suitable to vultures saw an average increase in all-cause human death rates of 4.2% in the years following their sudden collapse. This number is measured relative to areas that were never well-suited for vultures and thus unaffected by their decline. Our results hold up to multiple robustness checks and specifications, and to an alternative triple-difference approach that exploits the fact that negative effects are likely to be concentrated in districts which had *both* vultures and large livestock populations. The effect size we obtain implies an average of 104,386 additional deaths a year relative to a population of 430 million people in our main sample. Using an India-specific value of statistical life of $665,000 (Nair et al. 2021), this implies mortality damages of $69.4 billion per-year.

Lastly, the example of vultures suggests that one way to target evaluation, conservation and protection efforts is to focus on what are known as *keystone species* — those that help “hold the [eco]system together.”[2] Keystone species are seen as being crucial to the functioning of an ecosystem, sometimes providing unique services, such that if they are removed, the effects on the ecosystem are potentially large (Paine 1969; Power et al. 1996; Hale and Koprowski 2018).

In India for instance, vultures have provided critical environmental sanitation services. The 2019 livestock census in India reported a population of over 500 million animals, more than any other country in the world. Vultures are extraordinarily efficient scavengers and farmers historically relied on them to quickly remove livestock carcasses (D. L. Ogada et al. 2012). In the course of a few years in the second half of the 1990s, the number of Indian vultures in the wild fell by over 95%. Once too numerous to count, with a population in the tens of millions, this decline is the fastest of a bird species in recorded history and the largest in magnitude since the extinction of the passenger pigeon in the United States. Of the three primary species affected by dicoflenac, just a few thousand birds survive in India today, and they are all critically endangered.

As vultures died out, the scavenging services they provided disappeared too, and carrion were left out in the open for long periods of time. Ecologists have argued that this may have led to an increase in the population

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

of rats and feral dogs, which are a major source of rabies in India. Rotting carcasses can also transmit pathogens and diseases such as anthrax, to other scavengers. In addition, these pathogens can enter water sources either when people dump carcasses in rivers or because of erosion by surface runoff (Vijaikumar et al. 2002; R. T. Watson et al. 2004; Markandya et al. 2008; D. Ogada et al. 2016). We discuss these possible mechanisms further in Section 2.

The cause of vultures' death was initially mysterious. It was only in 2004 that research showed that several vulture species would develop kidney failure and die within weeks of digesting carrion with even small residues of the chemical diclofenac (Oaks et al. 2004). This discovery was a surprise because diclofenac is a common painkiller, harmless to human beings, and has been widely prescribed for people across the world since it was introduced in 1973.

In 1994, farmers in India began using this drug, previously prescribed only to humans, to treat their livestock (Cuthbert et al. 2014). The veterinary use of diclofenac became newly feasible and economically viable because of the entry of cheap generic brands made by Indian companies, and the expiry of a patent long held by the pharmaceutical company Novartis (Subramanian 2015). Unfortunately, treating cattle with diclofenac produced carcasses that were deadly to vultures.

In addition to our headline finding of an increase in human death rates in districts which lost vultures, we also find several additional pieces of evidence that support the hypothesis of increased mortality due to the removal of scavenging services. Carcass dumps in India tend to be on the outskirts of towns.

We find that elevated mortality is largest in these populated areas. We document evidence of higher dog populations in high-vulture suitability districts, and a sharp increase in sales of rabies vaccines following the veterinary use of diclofenac. We also find evidence of worse water quality in districts affected by the disappearance of vultures after their collapse.

Our results suggest that the restoration of vultures could lead to large increases in human welfare in India and suggest a critical need to protect vultures in other settings such as parts of Africa, where the birds still exist and feed partly on livestock carrion. They also point to the importance of evaluating the role of keystone species in different production functions such as public health. Prospectively evaluating the effects of policies such as the introduction of new chemicals on these species might reduce the probability of negative outcomes like those we study here.

Related Literature Our work links to several strands of the literature. We quantify the impact of a catastrophic shock to a keystone species with evidence on mechanisms. Economic theory shows that this type of estimate is essential to a meaningful cost-benefit analysis of conservation policy and ecosystems management (Weitzman 1992; Solow et al. 1993; Weitzman 1993; 1998; Nehring and Puppe 2002; Brock and Xepapadeas 2003).

We also build 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; Hooper et al. 2005; Estes et al. 2011; Martin et al. 2013; Ceballos et al. 2015;

J. E. M. Watson et al. 2016; Luis et al. 2018; Dainese et al. 2019; Schmeller et al. 2020). We join a nascent strand of the economics literature that has provided empirical evidence on the value of biodiversity. Using variations in environmental suitability, Alsan (2015) studied the long-term effects of the tsetse fly on agricultural production and political institutions. More recent papers study how farmers increase their use of insecticides to substitute for the loss of pest control following declines in insect-eating bats (Frank 2021); how air pollution increases after tree die-offs caused by the emerald ash borer (Jones and McDermott 2018); and how reintroducing wolves can change the behavior of deer and reduce deer-vehicle-collisions (Raynor et al. 2021).

Finally, we add to a body of work outside the economics literature on the vulture collapse in the Indian sub-continent. Vibhu Prakash et al. (2012), Cuthbert et al. (2014), and Galligan et al. (2020) document the magnitude and spatial extent of the loss of vultures and investigate whether restrictions on the veterinary use of diclofenac have aided recovery.[3] To the best of our knowledge, the closest paper to our work is Markandya et al. (2008) who use national aggregate statistics to perform a back of the envelope calculation suggesting the socio-economic costs of losing vultures are around 34 billion dollars for the period between 1993 and 2006 when considering only the impacts of increased mortality from rabies, but excluding water pollution and

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

other infectious diseases. In this paper, we collect panel data at the district level to test whether the decline in vultures had a detrimental effect on health outcomes, and leverage baseline variation in vulture suitability to identify the causal effect of their decline.

The remainder of this paper is organized as follows. In Section 2 we describe the role of vultures as scavengers and outline the mechanisms through which their disappearance might impose costs on society. In Section 3 we discuss the cause of the sudden population collapse of vultures in India. In Section 4 we describe the sources of data we use in this paper. In Section 5 we outline the econometric approach we use and present different specifications that we take to the data. In Section 6 we present our estimates of the mortality impacts of losing vultures including an assessment of the costs of replacing their ecosystem services with technology (incinerators). We also present other supporting evidence on the hypothesized mechanisms and a summary of different robustness checks and alternative specifications. We conclude in Section 7.

2 The Role of Vultures as Ecosystem Scavengers in India

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.[4] In comparison, an average human has a pH level of two in their stomach, making it ten to a hundred times less acidic than that of a vulture. This is one of the key adaptations that allows vultures to safely consume carrion, and also results in most bacteria not surviving their digestive system (D. L. Ogada et al. 2012; Roggenbuck et al. 2014).

Vultures are extremely effective at reducing a carcass to its bones, and can consume the carrion of an entire cow within forty minutes (D. L. Ogada et al. 2012). Other scavenging species are not good substitutes from a sanitation point of view because they leave flesh behind. Recent experimental evidence confirms that non-vulture species are not able to compensate and functionally replace vultures in terms of scavenging efficiency (Hill et al. 2018). For this reason, the historic presence of large and stable vulture populations simultaneously reduced pathogen and bacteria concentrations in the environment, as well as crowded out other mammalian scavengers such as dogs and rats that transmit various diseases including rabies (Moleón et al. 2014). In the absence of vultures, the composition of species that feed on carcasses changes towards dogs and rats.[5]

The removal of carrion from the environment by vultures becomes more important in low to middle income countries where these birds have effectively

[4] 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.

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

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

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

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

Muslims will not consume animals not killed according to *halal*.

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

3 The Sudden Collapse of the Indian Vulture Population

Vultures were once an ubiquitous sight across India with a population that may have exceeded fifty million birds. Today, the three species that made up the bulk of the population are considered critically endangered after declining by more than 95%.[6] Their collapse is attributed to chemical residue of the

[6] 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 have diclofenac residue

pain killer diclofenac in livestock animals, administered by farmers to treat fevers and inflammations. A vulture that feeds on a carcass with diclofenac residue can develop kidney failure within weeks and die.[7] In Figure 2, we plot the classification of districts according to their baseline habitat suitability for the affected vultures (we explain this classification in detail in Section 4.1).

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

Anecdotal accounts place the timing of the patent expiration in the early 1990s (Subramanian 2015). Sales data that we purchased from the company IQVIA shows a dramatic growth in the entry of Indian drug manufacturers around this time (see Figure 3a). In order to more precisely determine the onset of diclofenac use, we draw on additional sources of data. We start with formal records regarding the patent and its expiration. The patent originally belonged to the pharmaceutical company Novartis. Using documents from the Federal Drug Administration regarding drug patents, we are able to trace the first approval for a generic version granted to Novartis in 1993. See the

that caused the collapse in the other species.

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

Appendix for additional details. This is consistent with a survey of veterinary clinics conducted by Cuthbert et al. (2014) which indicates the first veterinary formulations in India became available in 1994. With these sources of information, we classify 1994 as the first year in which diclofenac was widely used to treat livestock, and assign this as the year of treatment onset.

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

At first, several conjectures were made regarding the potential cause. These included the emergence of a new wildlife disease or the effect of pesticide accumulation, as well as deliberate poisoning by western countries (Subramanian 2015). It took about a decade to establish the root cause when Oaks et al. (2004) used both autopsy data, and experimental exposure of vultures to diclofenac, to show that even trace amounts of diclofenac in the carcasses that vultures feed on results 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 document rampant illicit use of diclofenac in livestock including through the diversion of human doses (Galligan et al. 2020).

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

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

4 Vulture Presence, Health & Livestock Census Data

In this section, we briefly summarize data sources that we use in our analysis. We also use the raw data to provide descriptive evidence of the growth of dicoflenac, the decline of vultures, and possible effects on mortality.

4.1 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: (i) whether the district intersects with the range map, and (ii) 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). Figure 2 shows the spatial distribution of the classification into high and low suitability categories for diclofenac-affected vultures.

The data in the species distribution maps provided by BLI is the most complete source of information regarding the habitat areas of bird species around the world. BLI also assess the conservation status and extinction risk as part of the Red List, produced by the International Union for Conservation of Nature.[8] 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.[9] 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.[10] Additional information on this dataset is provided in the Appendix (Section C).

4.2 Sales & Product Entry of Pharmaceuticals in India

We purchased data from IQVIA on the sales 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 on the quantity sold, value sold, and the year when product was launched. Sales of rabies vaccines and of dicoflenac-based painkillers are of particular interest in the context of this paper.

In Figure 3a, we plot both the price and quantity sold of injectable painkillers containing diclofenac. We see that prices dropped dramatically over a short period of time such that by 1996, the mean price begins to stabilize at less

[8] 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.

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

[10] 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.

than half of its level in 1991. Meanwhile diclofenac sales increased by almost ten-fold from 1991 to 2003. Although these data largely correspond to medical sales, the sharp fall in price that we observe helps explain the reported entry of diclofenac into the veterinary market in 1994 (Cuthbert et al. 2014). We plot data on injections as that is the version of the drug that is most commonly used to treat animals.

4.3 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 part this is because the birds went from being too numerous to count, to nearly extinct, in a very short period of time. However, we were able to collect data on the recorded observations of different bird species at the national level. We draw on records from the Global Biodiversity Information Facility (GBIF) database, which aggregates multiple reporting sources of data, from scientific studies and citizen science reports.[11].

In our empirical specifications we use the habitat suitability measures described earlier, but it is nevertheless informative to use GBIF data to compare how the national share of observations for diclofenac-affected vultures evolves relative to all other bird species. To do this, we count the number of observations of the diclofenac-affected vultures, and all bird species that have non-zero observations reported each year during 1990 to 2005. Figure 3b,

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

shows a decline in the share of diclofenac-affected vultures relative to all other bird species, with a trend break that follows the veterinary use of diclofenac in 1994. Unfortunately these data cannot be used to draw conclusions about the rate of decrease of vultures because once it became known that they were growing rare in the wild, bird enthusiasts would have dedicated more effort to documenting them. These considerations make GBIF records unsuitable for our main empirical specifications.

In the Appendix we complement this information by reproducing 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 these surveys — they show a decline by about three orders of magnitude in ten years.

4.4 Health Outcomes

We use mortality data at the district level from the Vital Statistics of India (VSI), reported as part of the Civil Registration System (CRS). 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, and if more than 75% of men work in non-agricultural jobs (Burgess et al. 2017).

Using the classification into high and low suitability for the diclofenac-affected vultures, we plot the mean population-weighted all-cause death rate in Figure 3c. We observe an increase in mortality in the high-vulture-suitability districts following the introduction of veterinary diclofenac. However, no similar change in magnitude or trend is observed in the lowest suitability category. The habitat suitability groups 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. While high-suitability districts exhibit a break from their 1988 to 1993 trend, low-suitability districts maintain the same mean death rate from 1988 to 2005.

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

An important limitation of CRS data in India is that many vital statistics events go unrecorded, and as a result, the CRS under-reports the true magnitude of mortality. We adjust for this when interpreting our empirical results and discuss this further in Section 5.

4.5 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. Notwithstanding the name, the livestock census also reports a count of dogs at the district level, however, these were only systematically collected for feral dogs starting in 2012.[12]

We contacted the relevant government departments of different states to obtain data from their livestock censuses as conducted in 1987, 1992, 1997, 2003, and 2012. We use the data to classify districts as high or low livestock districts at baseline (as above or below the median level), which we use as part of a triple-differences design (see Section 5). We are also interested in the number of dogs recorded at the district level in 2012. If dog populations are higher in the high suitability areas for diclofenac-affected vultures, then that is consistent with the anecdotal evidence regarding the increase in feral dogs, animal bites, and rabies cases.

4.6 Water Quality

India's Central Pollution Control Board operates a network of water quality monitors covering different surface and groundwater sources. Greenstone and Hanna (2014) draw upon this data and use 489 monitors located at different points along 162 rivers to create an unbalanced district-level panel spanning

[12] 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."

1986-2005. We use this dataset for our analysis and more details on its construction are available in the original paper.

4.7 Stable District Boundaries

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

5 The Collapse of Vultures in India as a Natural Experiment

To estimate the causal effect of the collapse in vulture populations on public health, the ideal experiment would randomly assign vultures to different districts across India. Fully randomizing the distribution and abundance of vultures would allow us to study their effects through the channels of sanitation provision, and crowding other mammalian scavengers, such as dogs and rats. This ideal experiment is impossible to conduct.

In practice, the density of vultures is determined by a combination of environmental conditions creating variation in baseline populations in Indian districts. The poisoning of vultures from diclofenac residue in livestock carcasses provides a plausibly exogenous and large shock, affecting those areas where vultures were historically prevalent. The timing of this shock was not based on local factors but rather was determined by the expiry of a long-standing international patent, the consequent approval of a generic formulation in 1993, and the introduction of veterinary formulations in 1994. Nor were the effects on vultures anticipated at the time, indeed the connection of the drug to the demise of specific vulture species was only made a decade later in 2004. Finally, diclofenac itself was neither new to humans nor harmful to people or cattle. To this day it remains one of the most widely used treatments for pain and inflammation across the world (Altman et al. 2015).

5.1 Differences-In-Differences Design

We use a difference-in-differences approach to estimate the impact of vultures on health outcomes. We treat the sudden decline in vultures after 1994 as a shock resulting in the removal of a key ecosystem service, thus resulting in lower sanitation and an increased risk of disease including from rabies, following the mechanisms described in Section 2. We then 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 this type of difference-in-differences design is that both groups of districts would have seen their health outcomes develop along parallel trends in the absence of the collapse in vulture populations.[13]

More precisely, we estimate the following event-study-like regression specification:

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

Our main outcome of interest is the all-cause death rate, y_{daszt}, in district d, rural or urban area a, state s, in zonal council z, and time period t. We denote the treatment variable as HVS, which flags the high-vulture-suitability districts. Explicitly, it is a dummy variable that equals one for districts that we classify as having *high suitability* for the three vulture species affected by the exposure to diclofenac, and zero otherwise. We define high suitability as being in the top and middle terciles of the overlap between vulture ranges and districts areas (see Figure 2). We interact the suitability dummy with year dummies. We omit 1993 as the baseline year as that is the year of approval of the Novartis generic and prior to the introduction of veterinary formulations in 1994 (Cuthbert et al. 2014).

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

[13] This implicitly requires two additional assumptions that we find reasonable. First, that vulture populations were in equilibrium prior to the onset of diclofenac use. Second, that diclofenac was used widely to treat cattle and not only in areas with high suitability for affected vultures.

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. This is because the districts we classify as low suitability may still be affected to some degree since their baseline vulture populations are unlikely to have been zero.[14] 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, and healthcare access.[15]

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

[15] 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 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. 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.[16]

These fixed effect designs also help adjust for under-reporting in death rates from the CRS since our estimates are based on relative changes and not the absolute levels of mortality in the data. In the Appendix, we use an alternative source of more aggregated vital statistics data from India's Sample Registration system to show that although the CRS underestimates mortality rates by about a factor of two relative to the SRS, after controlling for state and zonal council-by-year fixed effects, both sources of data allow us to recover similar trends in mortality rates. When reporting estimates in percentage terms, we use the nationally representative baseline mean of all-cause death rates of 10.7 deaths per-1,000 people between 1988 to 1993, as reported by the United Nations Population Division.

To further test that any observed results are strictly driven by the inter-

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.

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

action 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.

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

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

5.2 Investigating the Role of Livestock in a Triple-Differences Design

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

The mediating role of livestock in the link between vultures and mortality can be tested through a triple-differences approach. Thus in addition to our main difference-in-differences design, we also estimate a triple difference specification. To do this we first construct a measure of baseline livestock for each district. For this, we compute for every district the mean of livestock counts in 1987 and 1992 using data from the corresponding livestock census. Next, we construct a dummy variable, (High Livestock), which takes the value one when the district has above the median level of livestock at baseline. Finally we run a specification as below:

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

Just as high livestock regions might be more affected by the loss of vultures, so might urban areas. Populated regions are more likely to have animal landfills on their outskirts, 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. Thus a similar triple-difference design can also be implemented using an indicator for a district-area being urban.

6 Results

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

6.1 Comparison of High and Low Suitability Districts at Baseline

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

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

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

6.2 Results for All-Cause Death Rate

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

Following the onset of diclofenac use after 1993, and the first observed signs of large-scale decline of vultures in 1996, we find that death rates from all causes increased in the high-vulture suitability districts. In 1996, the first year in which the decline in vulture populations gained widespread recognition, the all-cause death rate was higher in the high-suitability districts by 0.65 deaths per-1,000 people. By the end of the sample, in 2005, death rates were higher by about 1.4 deaths per-1,000 people. These reflect an increase of 6.1% and 13.1% relative to the nationally representative mean level of 10.7 deaths per-1,000 in the pre-treatment period, respectively.

Farmers gradually increased diclofenac use after the expiry of the patent. This should have caused the vulture population to decrease over the next few years. This is consistent with both GBIF and transect data (see Figures 3b and A1). Once vulture populations reach a low equilibrium (functionally extinct in the wild) any further changes in diclofenac use will have no effect on the sanitation services provided by the vultures in the ecosystem. These dynamics would suggest that death rates in high-vulture suitability regions should first diverge from the low-suitability control over a few years and then flatten out. This is precisely what we see in Figure 4 where an equilibrium treatment effect is reached around 2000, by which time vulture populations were a shadow of their previous levels and designated as critically endangered by the IUCN Red List. Importantly, these patterns would hold only if no compensating adaptive investments were made to replace vultures. This appears to be true — the alternative means of disposal is the use of incinerators and government reports as late as 2020 document their near total absence (Central Pollution Control Board 2020).

We turn next to our aggregate specifications in Equation 2. Table 2 contains these results both with and without temperature and rainfall controls.

The model in Panel A, Column 1 aggregates over the year-by-year coefficients in the event study by using a single post-dummy for years after 1993. On average, death rates are higher by 0.91 deaths per-1,000 people. Column 2 breaks this down into averages for the 1994 to 1999 period and the equilibrium period (2000 to 2005) as in Equation 2. We estimate precise increases in the all-cause death rate by 0.52 and 1.26 deaths per-1,000 people in the two periods (Panel A, column 2). These models control for zonal-council-by-year fixed effects capturing regional factors that might change death rates including regional and national macro-economic factors.

One concern we may have is the possibility of differential reporting of death rates in high vs low suitability districts that may not be fully captured by zonal trends. To control for this, in Panel A Column 3, we use a specification that includes linear time trends for each state, which is the level at which the civil registry reporting system is administered. These controls soak up some of our variation, in particular in the period where treatment effects are also growing over time. However our finding for equilibrium outcomes remains qualitatively similar, a fairly precise point effect of 0.48 additional deaths per-1,000 people. This reflects a 4.2% increase relative to the nationally representative mean level between 1988 and 1993 of 10.7 deaths per-1,000 people (UN 2022), as reported in the SRS data.[17]. We regard this as our preferred specification for estimating equilibrium elevated death rates due to the disappearance of vultures. Finally we report results using state-by-year fixed effects in Column

[17] 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. We use the values as reported by the UN Population Division. See section 4.4: Quantifying Under-Reporting in the CRS Data for additional details.

4. This absorbs more variation but our results remain broadly similar.[18]

The fact that carcass dumps tend to be on the outskirts of population centres suggest that urban areas (as defined by the census) might see larger effects than the combined sample. 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.[19] Using the urban-rural breakdown of reported district death rates, we re-estimate all models for urban areas only and report results in Columns 5-8 of Table 2. Across all specifications we find that urban areas experienced a larger increase in death rates relative to the combined sample. For our preferred specification including state-linear trends (Columns 3 and 7), urban death rates increase by 0.68 per 1000 people after reaching equilibrium (2000-2005). This compares with an estimate of 0.48 in the combined sample.

6.3 Long-Difference Models

In the main analysis described above, we balance our panel to require that each district in the panel reports death rates every year from 1988 to 2005. This limits the number of districts in our sample. Many districts are missing at least a year of data, and in the case of the state of Uttar Pradesh, we are

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

[19] Areas classified as urban are not necessarily similar to a city, and might be closer to a dense and large village. An area is officially classified as urban by the Census if it has a population above 5,000 people, or if more than 75% of men work in non-agricultural jobs.

missing data for all districts from 1996 to 1999.[20] After we hold districts in their 1981 geographic borders, there are 340 districts in our sample. Of these, 153 districts have fully balanced data in the combined urban and rural sample.

We are able to use a larger sample of districts by estimating a long differences model (Burke and Emerick 2016). Using long differences allows us to overcome issues with missing data in the middle of the panel, and allows us to take averages during pre- and post-treatment periods to address uneven reporting in those periods. The important modification is that we limit the sample to a pre-treatment period of 1990 to 1995, and a single post-treatment period of 2000 to 2005. With a relaxed requirement that districts only have non-missing data in these two periods, we are able to include as many as 324 districts (relative to 153) in the combined urban and rural sample, and as many as 279 districts (relative to 156) in the urban sample.

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

Estimating state-level trends poses more of a challenge once we relax the requirement for the panel to be balanced as some districts enter and exit the sample. For our preferred specification with state-linear trends, as well as when including state-by-year fixed effects, we recover smaller and impre-

[20] We went through considerable efforts to fill in any missing years of data. See the Data Appendix for a full documentation.

cise estimates when using data from both urban and rural areas (Table 3, Columns 3 and 4, Panel A). However, as before, in the urban area subsample the magnitude of the estimated effect remains meaningful and precise when including either state-linear trends or state-by-year fixed effect (Table 3, Panel B, columns 4-7). This is not surprising as the Indian practice of instituting carcass dumps on the outskirts of populated areas provides an ex-ante reason to expect the impacts of reduced scavenging might be concentrated in these areas.

6.4 Benchmarking the Effect of Vultures on Mortality

Failing to quickly dispose of rotting carcasses might intuitively seem like a significant public health concern. An effect size of 0.48 deaths per-1,000 people (Table 2, Panel B, column 3) implies an average of 104,386 additional deaths a year relative to a population of 430 million people in the main sample. Using an India-specific mortality risk reduction value (or value of statistical life) of \$665,000 implies mortality damages of \$69.4 billion per year.

These effect sizes are substantial but well-bounded by studies on the improvement of sanitation in other contexts. Geruso and Spears (2018) estimate a reduction in infant mortality rate in India by 28% when open defecation drops from 66% to 33%. In the context of privatizing water provision to improve sanitation and quality, Galiani et al. (2005) find that child mortality drops by 8%, on average, and as much as 26% in the poorest regions. Cutler and Miller (2005) estimate a even larger drop, of 48%, in infant mortality rates from the improvements to water quality in US cities around 1900. In Mexico, where water chlorination went up from 58% to 90%, Bhalotra et al. (2021) find that child mortality dropped by 45% to 67%. Finally, Kesztenbaum and Rosenthal (2014), find that when sanitation improved in Paris during 1880 to 1914 by one standard deviation, life expectancy increased by about two years.

An alternative way to contextualize these effects are to contrast them to another frequently studied environmental risk factor, namely high temperatures. T. Carleton et al. (2022) study the mortality effects of future temperature changes due to climate change. One of the countries projected to be most negatively affected by high temperatures is India. The estimates in this study suggest an increase in death rates by 0.60 per 1000 in 2099 under an RCP 8.5 warming scenario (a relatively pessimistic 'business as usual' projection of future emissions and warming). Climate change damages are comparable in magnitude to our estimate of a 0.48 increase in deaths from losing the sanitation services provided by vultures.

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

In 2019, India's livestock population was over 500 million, with about 300 million of those being cattle (20th Livestock Census). Although it is illegal to slaughter cows in India, they do not survive long after their productive life as milch animals because farmers may set them free, effectively denying them access to sufficient food or medicines. Assuming an average life span of about 10 years suggests an annual burden of about 30 million carcasses. Using the estimates from Ishwar et al. (2016), this suggests annual costs of operating a nation-wide network of carcass rendering machines of about USD 768 million (2014-15 dollars), solely for cows. This calculation also ignores air pollution damages from the incinerators.

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

6.5 Investigating the Role of Livestock

We turn next to the role of livestock in increasing the value of the sanitation services provided by vultures. In Table 4, we report results from the triple-difference specification in Equation 3. We find that following the collapse in vulture populations, high-vulture suitability districts that also had a high level of livestock at baseline showed a significantly higher increase in death rates, relative to districts with below median livestock populations. This gap widens further when restricting the sample to urban areas (Table 4, columns 3 and 4). These results are consistent with the hypothesis that the main driver of mortality after the collapse in vulture populations is the presence of a large supply of animal carrion that is not effectively scavenged, rather than simply the decline in vultures themselves.[21]

Another way to undertake the comparison embedded in the triple difference is to separately evaluate two diff-in-diff components. In Table 5, we split the sample into high and low vulture suitability districts. We then carry out a diff-in-diff exercise comparing districts with high vs low levels of baseline livestock, before and after the veterinary use of diclofenac.

This decomposition reveals that it is only in high-vulture suitability districts that baseline levels of livestock are associated with elevated death rates in the post period (compare Table 5, Columns 1 and 2 in Panels A and B). In areas where vultures were expected to be absent or less abundant, there is no differential change in mortality between low and high livestock districts. This is true even when restricting attention to urban portions of districts — here the difference is even more marked (Table 5, Columns 3 and 4).

This decomposition analysis also alleviates concerns regarding possible spatial clustering of the vulture suitability as it offers an alternative research design. The diff-in-diff results in Table 5, Panel A, only rely on districts in the more central parts of India, where environmental conditions are similar in terms of vulture habitats. The comparison of high to low livestock demonstrates that even without relying on districts located in coastal or mountainous

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

areas in India, we can recover similar magnitudes for the effect of the post-1994 collapse.

6.6 Evidence Supporting a Sanitation Channel: Dogs, Rabies, Water Quality

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

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

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

We also purchased national level data on the sales of rabies vaccines from IQVIA. These vaccines are administered as the live-saving treatment after an animal bite, although there are sadly many people in India who still die from rabies because they delay reporting to hospitals.[22] In Figure 5a, we observe a sharp increase after 1996 in the quantity of rabies vaccines sold.

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

We use data on the water quality outcomes that are most directly linked to a larger presence of carrion when disposal by scavengers declines: namely dissolved oxygen and fecal coliforms.[23] In Table 6, we report results from a triple-difference specification using water quality as an outcome variable and separately examining urban vs rural outcomes. We find that water quality deteriorates in the urban subsample (columns 2, 3, 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

[22] 36% of global deaths from rabies still occur in India (Chatterjee 2009).

[23] 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.

water samples using either the DDD or DD comparison (columns 5 and 6). In Appendix Table A3, 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).

6.7 Sensitivity Analysis & Robustness Checks

We evaluate the robustness of the main results in several ways and report outcomes in the Appendix. First, we further examine the presence of pre-trends in the data by extending the sample to cover 1981 to 2005, and verify that we recover similar estimates (Figure A2). To better account for other factors that could be changing over time at the state level we confirm that including state-linear time trends or state-by-year fixed effects produces qualitatively similar findings to those in the event study results (Figure A3). We also use census data to test for differences in per-capita hospitals and health centers, as well as doctors and health workers, between the two groups of districts before and after the collapse (Table A4). We are unable reject the hypothesis that there are no differences. We also run a battery of placebo tests using a variety of different outcomes and fail to detect meaningful differences. (Figure A5).

We also explore whether an alternative method of identifying treatment status affects our results by using a habitat suitability model. Habitat suitability models use data on the presence of the species of interest along with environmental conditions to generate predictions regarding the suitability of a habitat for the specific species. In short, the model first links geographic data on the presence of species to environmental conditions, and then uses the inferred relationship to classify the suitability of other geographic areas.[24] 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 A2. 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

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

effects in Figure A6 and A7. Lastly, 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.[25]

7 Conclusions

We live in an era of mass extinctions, only the sixth in the history of the planet and the first to be induced by human activity. Policies intended to preserve biodiversity have therefore never been more important. Yet the paucity of evidence on the costs of losing specific species has made it difficult to target conservation or recovery efforts. Focusing on keystone species is one way to narrow down what would otherwise be a large set of claimants for policy dollars. Nevertheless this still leaves us with the challenge of quantifying the costs of a catastrophic event like extinction.

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

[25] The permutation inference analysis also allows us to evaluate whether we are underestimating standard errors by clustering at the district level due to spatial correlation. This appears to not be a concern as the exact p-values we obtain are well below 1%.

at least 4.2%, averaged over 2000 to 2005.

Our results inform current vulture recovery efforts in India, and conservation efforts elsewhere. Vultures are important scavengers in parts of Africa as well as Spain but their populations are falling and diclofenac is still commonly used in many parts of the world. More broadly, this paper shows how local extinction events can be used to learn about anthropocentric benefits from biodiversity, potentially allowing us to make better policy before a species goes extinct everywhere in the wild. In addition, the vulture collapse in India provides a particularly stark example of the type of hard-to-reverse and unpredictable costs that must be accounted for when evaluating the introduction of new chemicals into fragile and diverse ecosystems. Although it is easy to be wise after the fact, it is plausible that a counterfactual policy regime in India that tested chemicals for their toxicity to *at least* keystone species might have avoided the collapse of vultures.

Beyond their impacts on public health, vultures also provide other important services. India's tanning industry once relied 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 (Subramanian 2011; Markandya et al. 2008).

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

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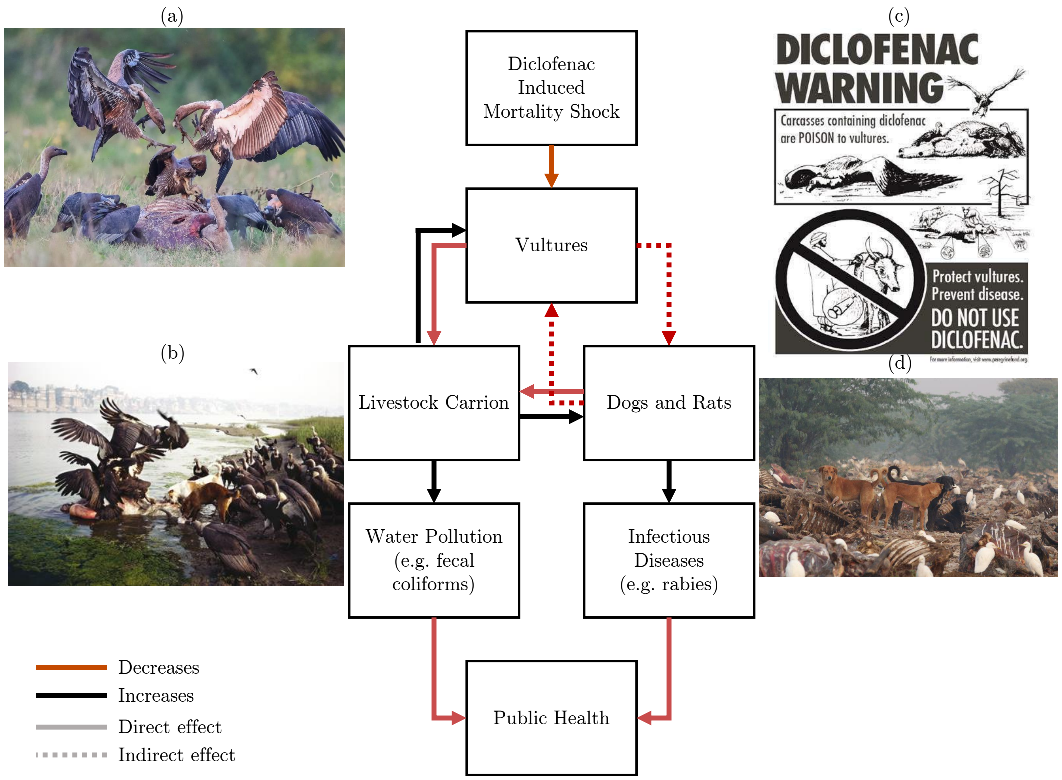
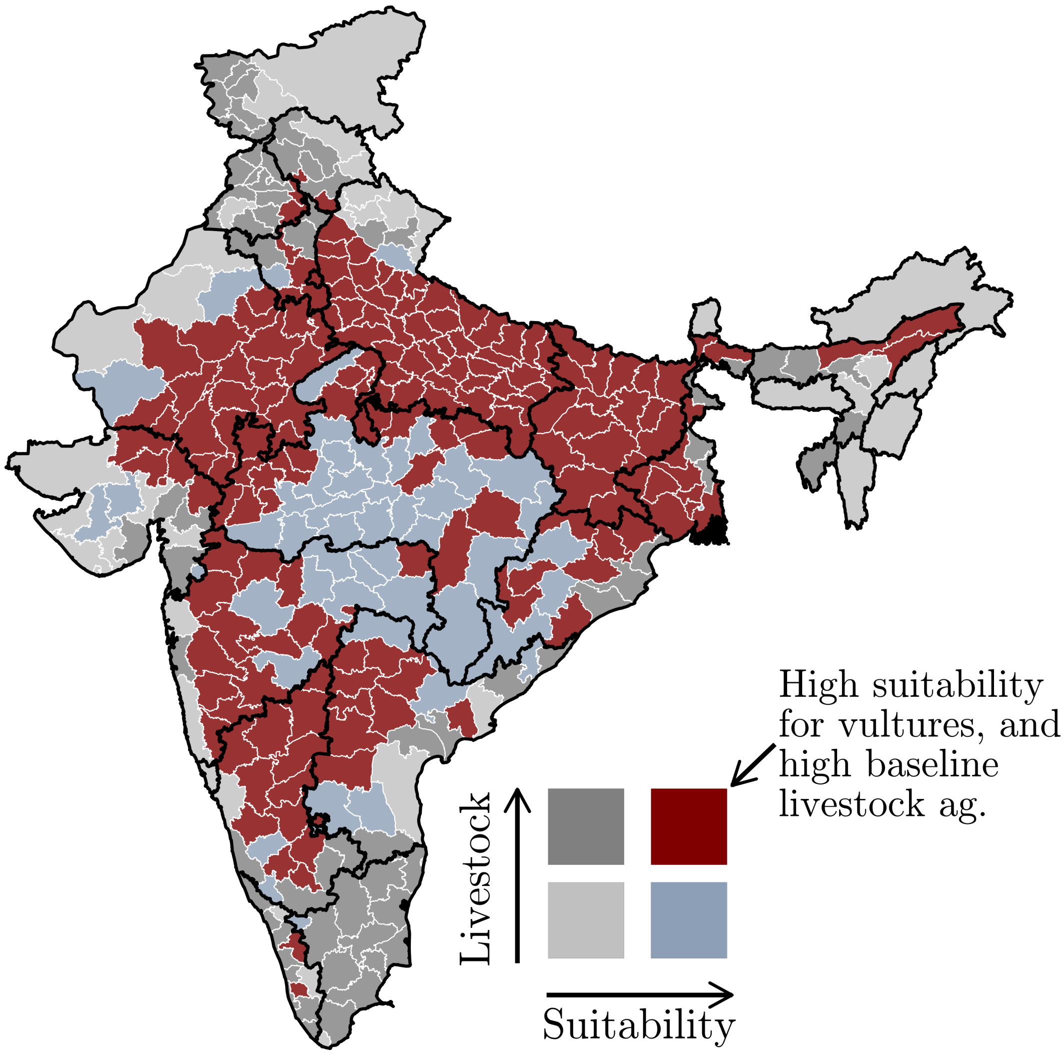
Figure 1: Schematic Relationship of Ecosystem Interactions & Environmental Quality



Notes: The figure summarizes the key components of the coupled natural-human system: (i) ecosystem interactions between vultures, dogs and rats, and livestock carrion; (ii) the impacts that mammalian scavengers and carrion have on environmental quality and public health. Red lines denote a decreasing effect, while black lines denote an increasing effect. Solid lines reflect a direct effect, while dashed lines reflect an indirect (reduced from) effect. Images to the left and right of the schematic model capture how (a) vulture packs descend on carrion; (b) how vultures compete with feral dogs over carcasses, which could end up in water bodies; (c) how warnings are issued to farmers against the use of diclofenac because of the negative impact on vultures; and (d) how packs of feral dogs roam animal landfills. Photo Sources: (a) Sagar Giri via National Trust For Nature Conservation. (b) Tom Stoddart via Getty Images. (c) The Peregrine Fund. (d) Anoop Kumar.

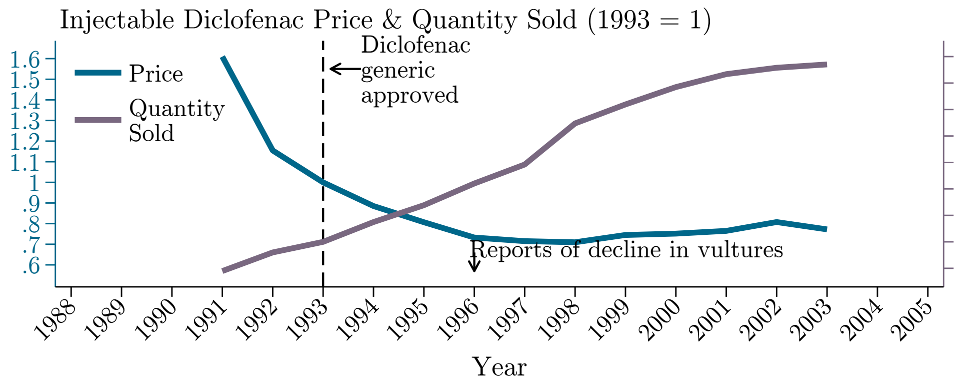
Figure 2: Spatial Distribution of Diclofenac-Affected-Vulture Ranges & Livestock Agriculture



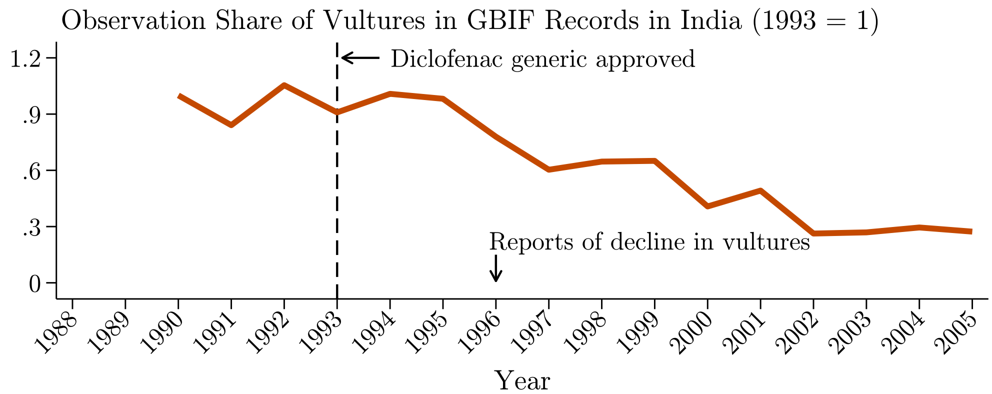
Notes: Districts in India, at their stable 1981 geographic borders, classified as high or low exposure to diclofenac-vulture-collapse, and as high or low baseline livestock agriculture (see Section 4 for more details).

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

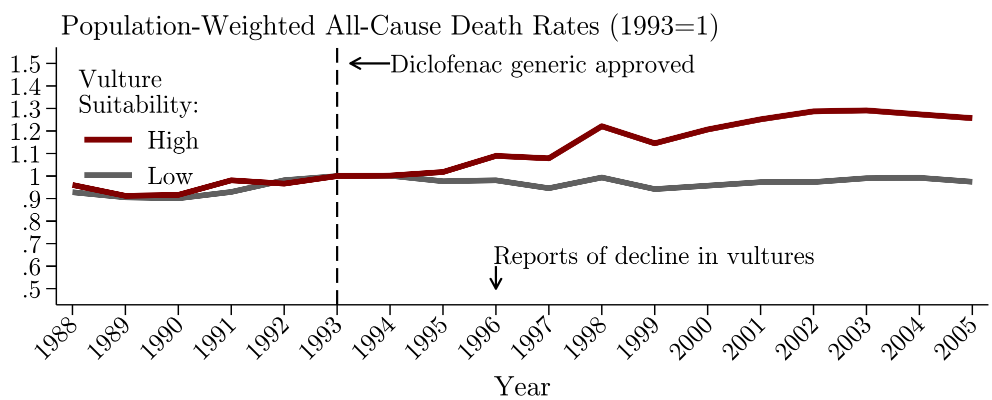
(a) Expansion in Diclofenac Around the 1994 Veterinary Use Onset



(b) Decline in Observations of Affected Vulture Species

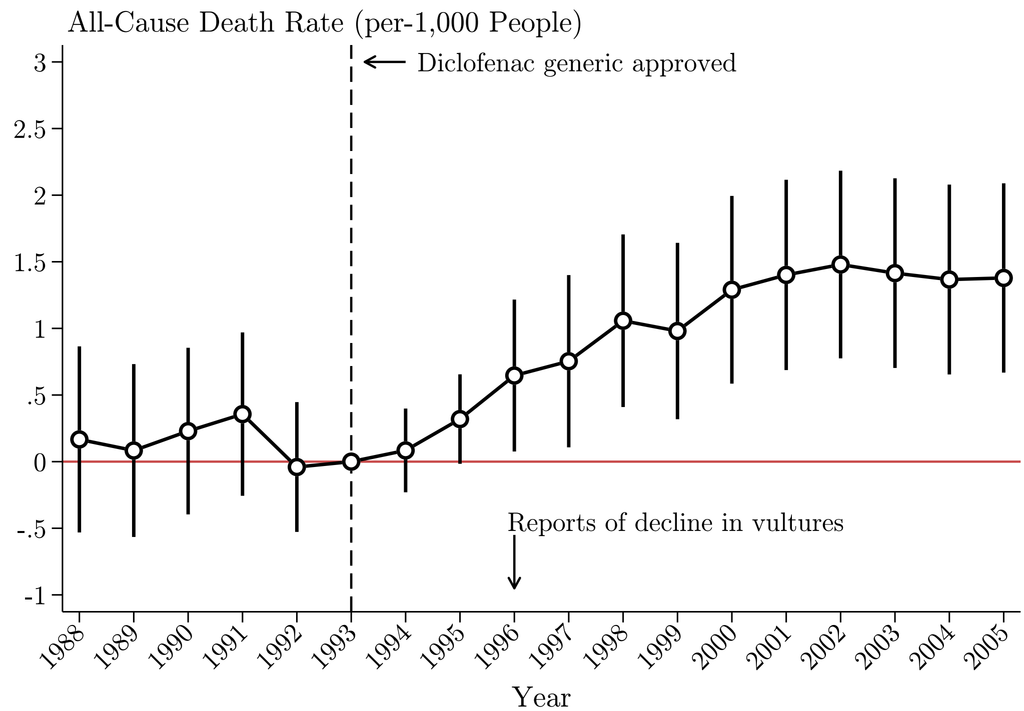


(c) All-Cause Death Rates by Vulture Habitat Suitability



Notes: (a) Injectable forms of diclofenac price and sales. (b) The share of vulture reports relative to 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.

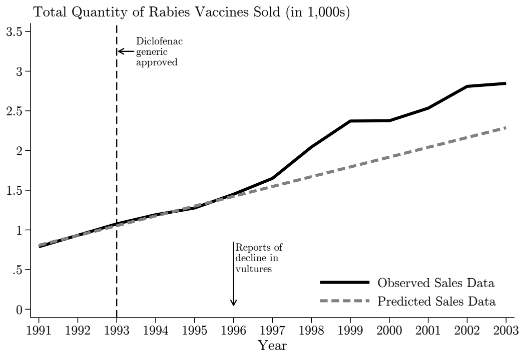
Figure 4: All-Cause Death Rates DD Estimation Results



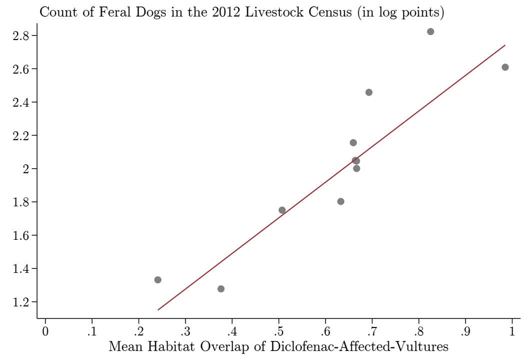
Notes: Estimation results from Equation (1). Comparing the high to low suitability vulture districts. Sample includes all districts (combining urban and rural areas) with balanced data from 1988 to 2005. Regression includes district and zonal council-by-year fixed effects. Observations are population-weighted. Standard errors are clustered at the district level.

Figure 5: Suggestive Evidence for Feral Dog Mechanism

(a) Sales of Rabies Vaccines



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



Notes: (a) National level data on all rabies vaccines sold from 1991 to 2003. The solid black line shows the total sold quantity, and the dashed gray line shows a linear trend using the data from 1991 to 1995. (b) District-level data on feral dogs 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 ($\overline{Y} = 10.7$)

Panel A. Without Weather Controls

	Combined Sample				Census-Urban Sample			
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
HVS×Post-1994	0.91				1.04			
	(0.16)				(0.25)			
HVS×[1994, 1999]		0.52	0.13	0.21		0.68	0.35	0.34
		(0.16)	(0.18)	(0.17)		(0.25)	(0.26)	(0.33)
HVS×[2000, 2005]		1.26	0.48	0.40		1.34	0.68	0.63
		(0.22)	(0.20)	(0.21)		(0.29)	(0.34)	(0.37)
R^2	0.74	0.75	0.77	0.80	0.67	0.68	0.70	0.76
N	2,754	2,754	2,754	2,700	2,808	2,808	2,808	2,754
Clusters	153	153	153	150	156	156	156	153

Panel B. With Weather Controls

	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
HVS×Post-1994	0.85				1.04			
	(0.16)				(0.24)			
HVS×[1994, 1999]		0.51	0.18	0.19		0.72	0.40	0.32
		(0.16)	(0.18)	(0.17)		(0.25)	(0.26)	(0.31)
HVS×[2000, 2005]		1.17	0.45	0.38		1.32	0.67	0.64
		(0.21)	(0.20)	(0.21)		(0.28)	(0.34)	(0.35)
R^2	0.75	0.75	0.78	0.81	0.68	0.69	0.71	0.76
N	2,754	2,754	2,754	2,700	2,808	2,808	2,808	2,754
Clusters	153	153	153	150	156	156	156	153
Zonal Council-by-Year FE	X	X	X		X	X	X	
State-Linear Trends			X				X	
State-by-Year FE				X				X

Notes: Estimation results for the specification in Equation (2). The estimation is comapring high-vulture-suitability (HVS) to low-vulture-suitability districts, after the collapse of the affected vulture populations following the onset of diclofenac use (post-1994), relative to years prior to the expiration of the patent. Results in columns 2 to 4, and 6 to 8 split the post-1994 period to two periods: 1994 to 1999, and 2000 to 2005. When we include state-by-year fixed effects (columns 4 and 8), three states get dropped as they have no district-level data. Reported mean of 10.7 deaths per-1,000 people is for the pre-treatment period of 1988 to 1993, obtained from the UN Population Division. 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.
All-Cause Death Rate Long-Differences Estimation Results ($\overline{Y} = 10.7$)

Panel A. Combined Sample

	(1)	(2)	(3)	(4)	(5)	(6)	(7)
HVS×Post-2000	1.23	0.73	0.68	0.26	0.16	0.18	0.16
	(0.25)	(0.25)	(0.24)	(0.16)	(0.15)	(0.15)	(0.15)
R^2	0.72	0.73	0.85	0.77	0.90	0.79	0.90
N	1,836	3,696	648	3,696	648	3,589	628
Clusters	153	324	324	324	324	314	314

Panel B. Census-Urban Sample

	(1)	(2)	(3)	(4)	(5)	(6)	(7)
HVS×Post-2000	1.23	1.07	1.01	0.66	0.62	0.62	0.62
	(0.25)	(0.27)	(0.29)	(0.22)	(0.23)	(0.23)	(0.23)
R^2	0.64	0.65	0.84	0.69	0.90	0.75	0.90
N	1,872	3,193	558	3,193	558	3,087	538
Clusters	156	279	279	279	279	269	269
Balanced	X						
Zonal Council-by-Year FE	X	X	X	X	X		
State-Linear Trends				X	X		
State-by-Year FE						X	X
Collapsed Sample			X		X		X

Notes: Estimation results for the specifications in Equation (2). The regressions compare the high to the low vulture suitability districts in the post-vulture collapse period (2000 to 2005) to the pre-vulture collapse period (1990 to 1995). Column 1 reports the results from the balanced sample throuhgout 1988 to 2005. Columns 2 to 7 use districts with unbalanced data, as long as the district has non-missing data in both the pre- and post-periods. Columns 1, 2, 4, and 6 maintain the district-year panel structure, and columns 3, 5, and 7 collapse the data to pre- and post-periods using population weights to obtain a weighted mean of the all-cause death rate in each period. All regressions include district fixed effects. Observations are population-weighted. Standard errors are clustered at the district level.

Table 4.
DDD Results for All-Cause Death Rate ($\overline{Y} = 10.7$)

Sample:	Combined		Census-Urban	
	(1)	(2)	(3)	(4)
HVS×Livestock×Post-1994	0.60	0.56	1.17	1.19
	(0.31)	(0.31)	(0.45)	(0.46)
HVS×Post-1994	0.49	0.46	0.29	0.32
	(0.26)	(0.26)	(0.34)	(0.33)
Livestock×Post-1994	0.05	0.06	-0.15	-0.15
	(0.24)	(0.24)	(0.36)	(0.37)
Weather Controls		X		X
R^2	0.74	0.75	0.66	0.67
N	2,754	2,754	2,790	2,790
Clusters	153	153	155	155

Notes: Estimation Results for the specification in Equation (3). The DDD estimation compares the districts that are high-vulture-suitability (HVS), and utilizes the additional sub-group of high-livestock at baseline. Using all livestock animals, we define the high-livestock dummy as being above the median at baseline, using the mean of the 1987 and the 1992 livestock censuses. Sample includes balanced district data, combining urban and rural areas (columns 1 and 2), or only urban areas in the districts (columns 3 and 4), from 1988 to 2005. All regressions include district and zonal council-by-year fixed effects. Reported mean of 10.7 deaths per-1,000 people is for the pre-treatment period of 1988 to 1993, obtained from the UN Population Division. Observations are population-weighted. Standard errors are clustered at the district level.

Table 5.
Decomposing the DDD Results by Districts' Vulture Suitability

Panel A. High-Vulture Suitability Subsample

Sample:	Combined		Census-Urban	
	(1)	(2)	(3)	(4)
Livestock×Post-1994	0.62	0.57	1.07	1.05
	(0.23)	(0.23)	(0.31)	(0.31)
R^2	0.828	0.831	0.765	0.771
N	1,350	1,350	1,386	1,386
Clusters	75	75	77	77

Panel B. Low-Vulture Suitability Subsample

	(1)	(2)	(3)	(4)
Livestock×Post-1994	0.08	0.11	-0.07	-0.12
	(0.25)	(0.25)	(0.38)	(0.40)
R^2	0.68	0.69	0.60	0.61
N	1,404	1,404	1,404	1,404
Clusters	78	78	78	78
Weather Controls		X		X

Notes: Estimation results for a specification similar to Equation (2). The estimation is comparing districts with high to low livestock agriculture at baseline, after the collapse of the affected vulture populations. We repeat the analysis in two subsamples: The districts with high vulture suitability where we expect high baseline livestock to affect health (Panel A), and low vulture suitability where do not expect baseline livestock to affect health (Panel B). Sample includes balanced district-level data from 1988 to 2005. All regressions include district and zoncal council-by-year fixed effects. Observations are population-weighted. Standard errors are clustered at the district level.

Table 6.
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		-0.123			1.244	
		(0.052)			(0.475)	
HVS×Diclofenac	0.003	0.046	-0.076	0.304	-0.114	1.212
	(0.026)	(0.031)	(0.044)	(0.353)	(0.452)	(0.332)
Urban×Diclofenac		0.067			-0.475	
		(0.049)			(0.361)	
$\overline{Y}_{\leq 1993}$	1.92	1.92	1.89	6.86	6.86	6.85
R^2	0.71	0.71	0.74	0.78	0.78	0.83
N	4,351	4,351	2,073	3,346	3,346	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 census-urban (U) and rural (R) areas, from 1988 to 2005. Observations are population-weighted. Standard errors are clustered at the district level.

For Online Publication: 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 in the Global Biodiversity Information Facility (GBIF) database are likely upward biased as there was likely more attention given to documenting and reporting vultures after it became public knowledge that their populations were in decline. Unfortunately, there are no large-scale repeating surveys of vulture populations as they were always seen as too numerous to count. One exception is a repeating population survey that took place along 70 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. We prefer to use data reported under the same regime, as this allows to fully control the conversion to rates. Second, the number districts that are fully balanced from 1988 to 2005 are 156, while there are only 104 balanced districts for the 1981 to 2005 period. When extending the panel to the full 1981 to 2005 period, and losing 33% percent of the districts, we recover similar results to those in the main text (Figure 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 4. The divergence in death rates only starts after the vulture populations collapse, yet the magnitude of the effect is smaller. By 2000, all-cause death rates 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 Examining Heterogeneity Between Urban & Rural District Areas

In Table A1, 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 have larger populations, are denser, and more likely to have an animal landfill site at their outskirts, we expect that a larger portion of the average effect is driven by the 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.5 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 in the United States.

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

We use observation records from eBird and from the Global Biodiversity Information Facility (GBIF) to construct the BIOCLIM suitability scores. We then take the mean level of the suitability scores across all three affected species, and use it to define high and low suitability dummy variables. We either split the suitability score into terciles, defining high suitability as the third and second terciles, or we define the high suitability dummy as being above the median suitability 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 A2. 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.6 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 6), as well as increasing levels of BOD and COD. Because BOD only captures biological uses of oxygen, it will be below the COD level which captures 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, turbidity declines as well. This is because scavengers tend to increase turbidity through the act of tearing carrion flesh. As shown in other aquatic environments, the absence of scavengers reduces turbidity (Santori et al. 2020).

In Table A3, 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 6). 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.7 Evaluating Changes to Healthcare Access

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

Here we use data from the 2001 and 2011 census to test whether those healthcare access metrics changed after 1994 in the high- relative to low-vulture-suitability districts. In Table A4, we report estimates that show no difference between the two groups of districts. This finding holds when we use the same set of districts as in the main analysis, or if we use the full set of districts that appear in the census. This result alleviates concerns that our main finding is capturing changes to the healthcare infrastructure that are somehow correlated with the location and timing of the vulture collapse.

A.8 Evaluating Changes to District Characteristics

We expand on the previous analysis on healthcare access and add several other placebo outcomes that should not be affected by the collapse in vulture populations. For each outcome, we have at least one year of data before, and one year of data after the collapse. We summarize the results in Figure A5, where we do not find that alternative explanations in the form of diverging employment or district infrastructure are consistent with the data. The overall differences are often very small relative to the mean of each outcome, and even when they are precisely estimated they move in the direction that would suggest improving health conditions in the treatment group.

A.9 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 A6 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.

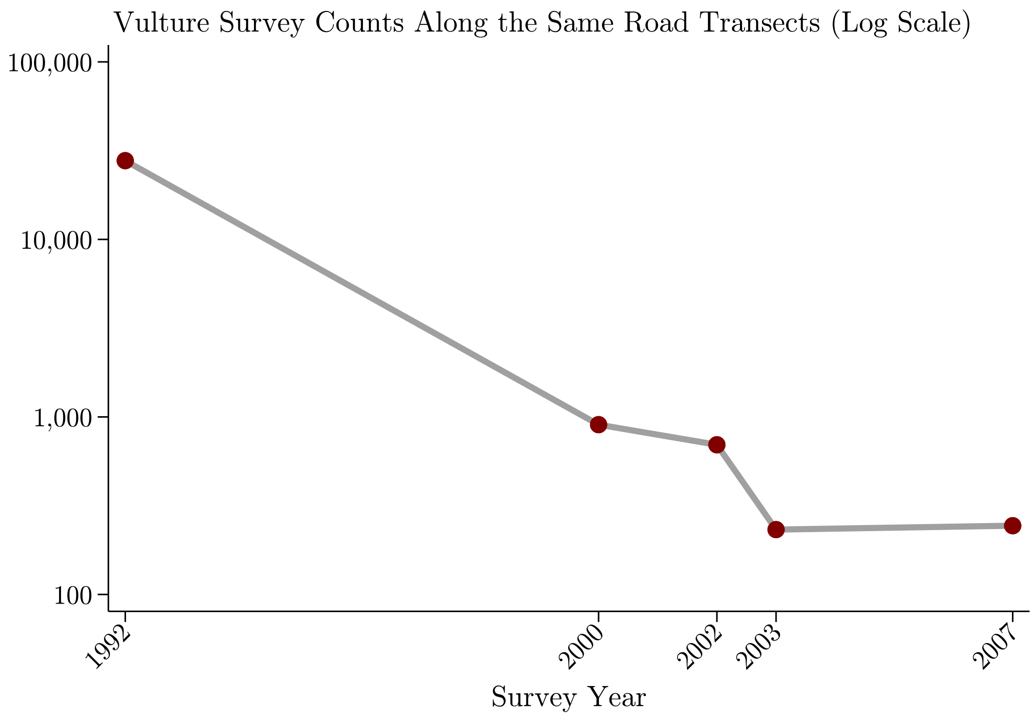
We also conduct the leave-one-out exercise by excluding one state at a time. This allows us to evaluate whether any potential changes in the reporting of vital statistics might be driving the estimated effect in a manner that is not already captured by the inclusion of state-level trends in Figure A3. We recover a narrow distribution of the coefficients with mostly overlapping 95% confidence intervals.

A.10 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.

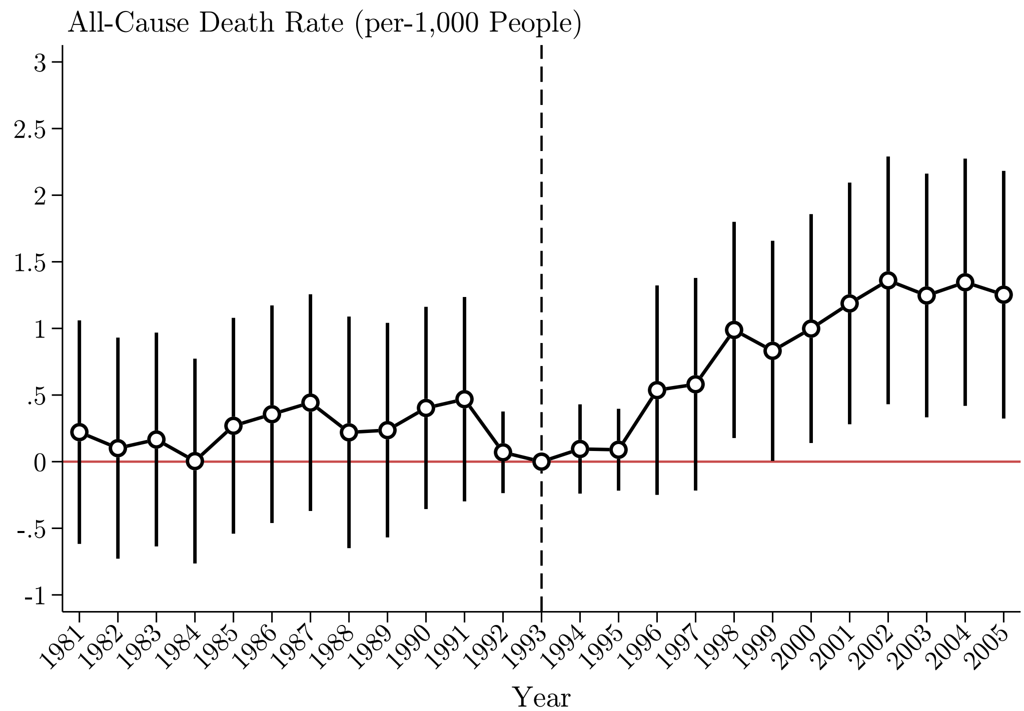
We randomly re-assign the treatment across the districts and re-estimate the effect using the specification in Equation (2), repeating the process 1,000 times. We either fully randomize the treatment dummy across districts and years (full), maintain the same temporal structure but randomly assign districts 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 A8, 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%.

Figure A1: Vulture Counts From Repeated Surveys Along Road Transects



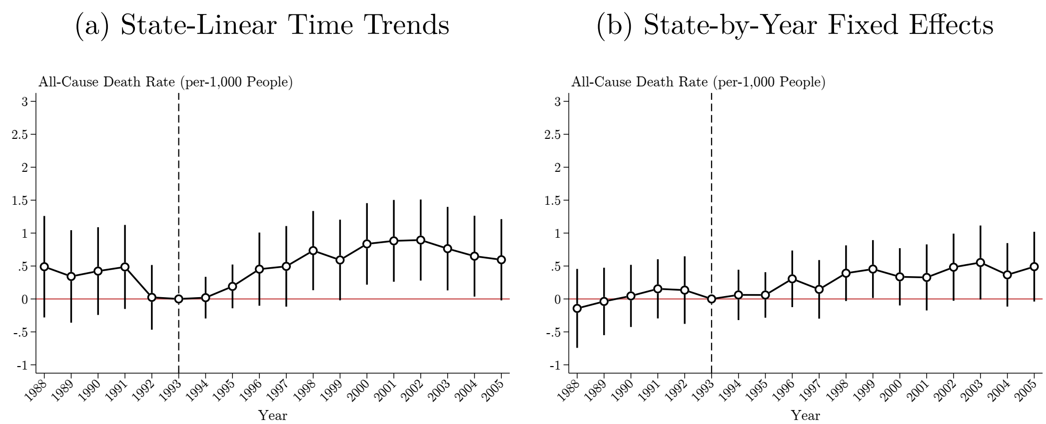
Notes: Each dot is the sum of surveyed vultures, in log scale, along the same 70 road transects for the three diclofenac-affected-species. Data are reproduced from Prakash et al. (2007).

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. Expanding the sample to 1981, while still using a balanced sample, lowers the number of districts 156 to 104. 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

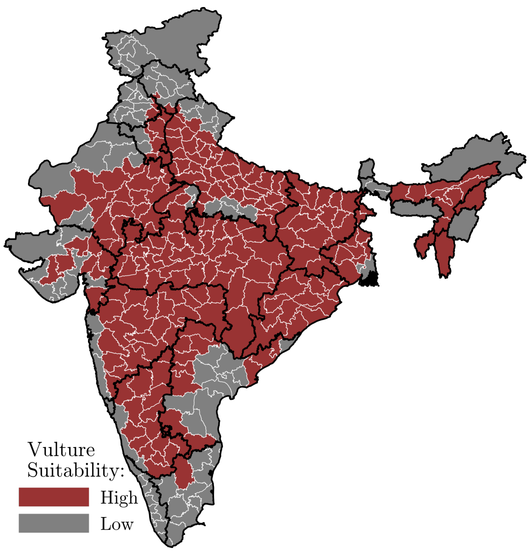


Notes: Estimation results from Equation (1). Comparing the third and second terciles to the first tercile of vulture habitat overlap. All regression include district fixed effects. The regression in (a) includes zonal council-by-year fixed effects and state-level linear time trends, and the regression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

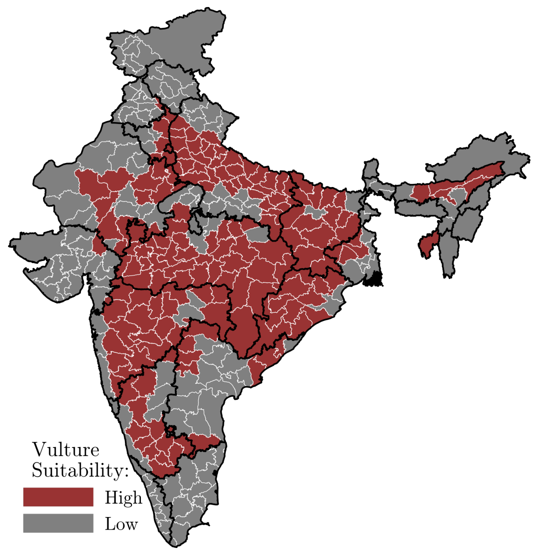
Suitability Score Terciles III & II

(a)



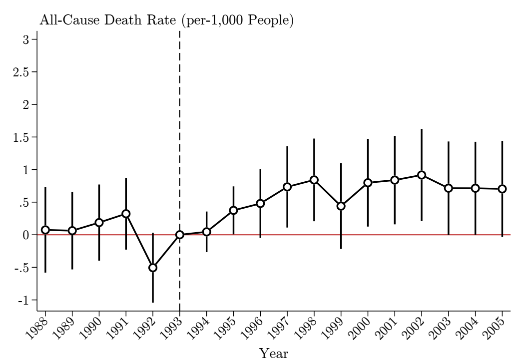
Suitability Score Above Median

(b)



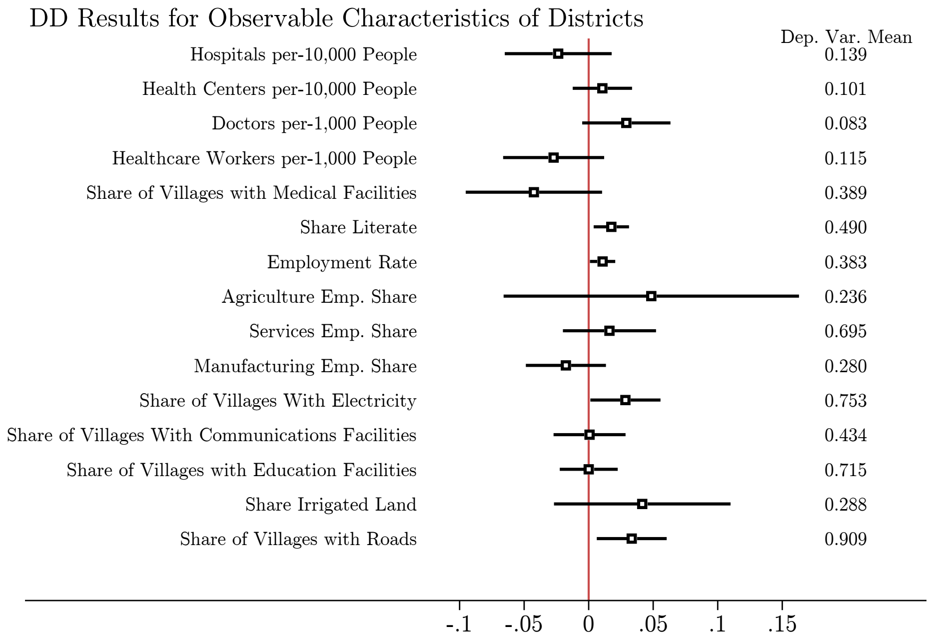
(c)

(d)



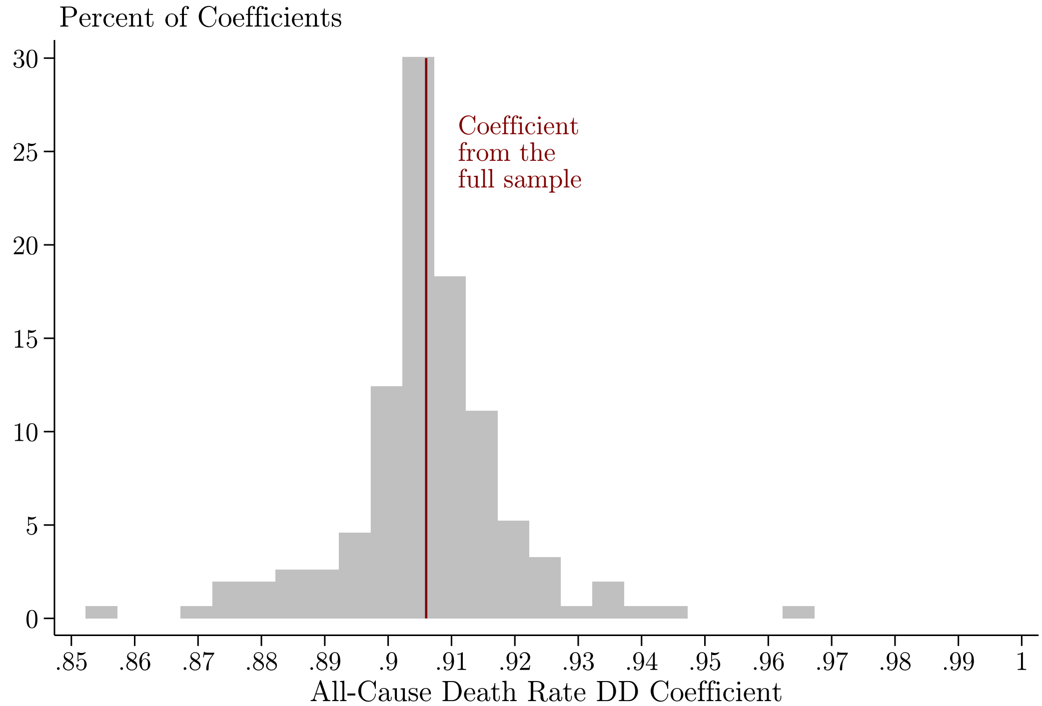
Notes: The treatment classification uses predicted suitability scores for the diclofenac-affected-vultures from the BIOCLIM habitat suitability model. We either split the suitability score into terciles and define treated districts as the third and second terciles (a and c), or split districts as above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

Figure A5: Summary of Placebo Results



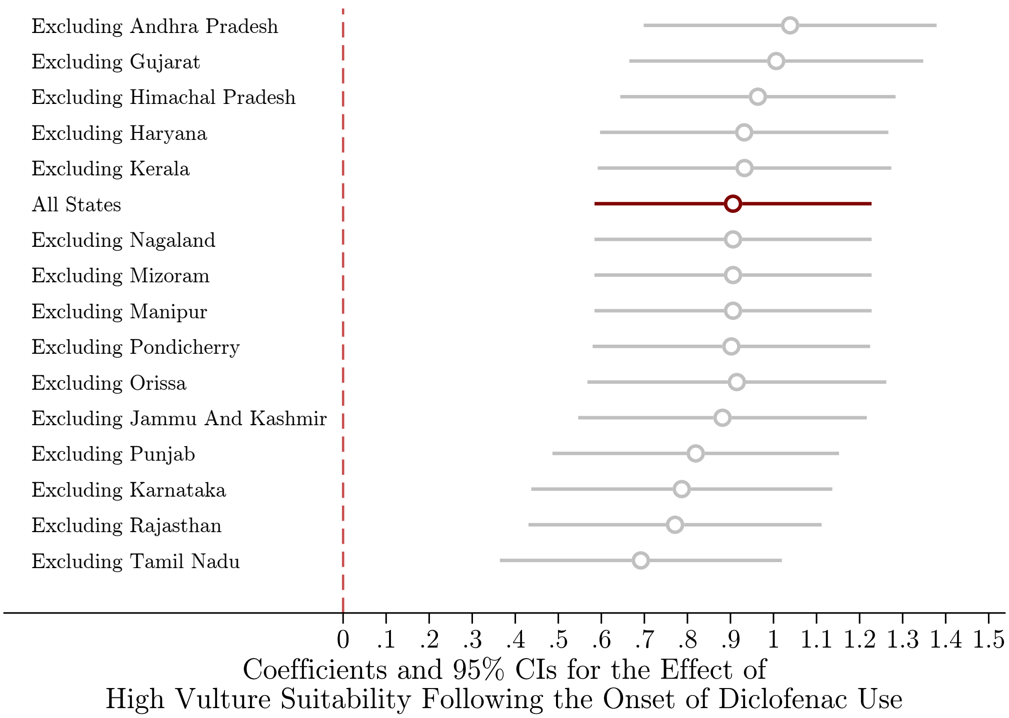
Notes: Estimation results for the specification in Equation (2). Each regression includes district and zonal council-by-year fixed effects. The sample includes all the districts in the balanced sample reported in the main analysis. Observations are population-weighted. Standard errors are clustered at the district level.

Figure A6: 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.

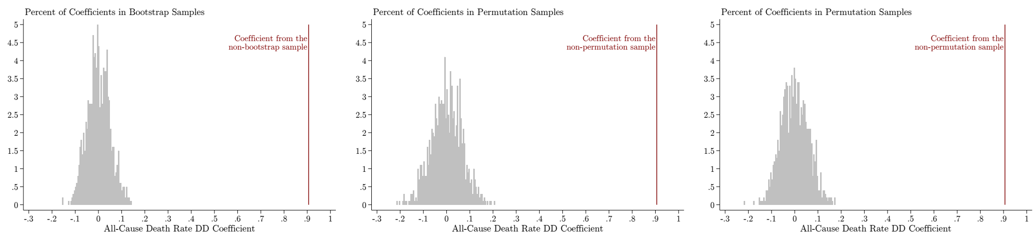
Figure A7: Distribution of Leave-One-State Out DD Estimation Results



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 A8: 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.

Table A1.
All-Cause Death Rate, per-1,000 People ($\overline{Y} = 10.7$)

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. Reported mean of 10.7 deaths per-1,000 people is for the pre-treatment period of 1988 to 1993, obtained from the UN Population Division 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 A2.
Results for All-Cause Death Rate Using BIOCLIM Classifications
($\overline{Y} = 10.7$)

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. Reported mean of 10.7 deaths per-1,000 people is for the pre-treatment period of 1988 to 1993, obtained from the UN Population Division. All regressions include district and zonal council-by-year fixed effects. Observations are population-weighted. Standard errors are clustered at the district level.

Table A3.
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		1.5			11.4			-7.2	
		(1.1)			(7.5)			(6.4)	
HVS×Diclofenac	0.7	0.2	1.8	1.8	-2.2	9.6	-0.7	1.4	-6.0
	(0.5)	(0.5)	(1.1)	(3.1)	(2.1)	(7.3)	(4.0)	(4.4)	(6.3)
Urban×Diclofenac		-0.6			-6.9			-0.2	
		(0.7)			(6.6)			(4.3)	
$\overline{Y}_{\leq 1993}$	4.01	4.01	5.03	25.32	25.32	28.61	36.44	36.44	40.30
R^2	0.74	0.74	0.75	0.71	0.71	0.75	0.79	0.79	0.78
N	4,339	4,339	2,062	4,146	4,146	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 census-urban (U) and rural (R) areas, from 1988 to 2005. Observations are population-weighted. Standard errors are clustered at the district level.

Table A4.
Estimation Results for Healthcare Access

	Main Sample		Census Sample	
	Per-Capita Hospitals & Health Centers	Per-Capita Doctors & Health Workers	Per-Capita Hospitals & Health Centers	Per-Capita Doctors & Health Workers
	(1)	(2)	(3)	(4)
HVS×Post-1994	0.07	1.83	-0.34	2.73
	(0.22)	(2.29)	(0.21)	(2.34)
$\overline{Y}$	1.79	18.03	1.80	21.37
R^2	0.772	0.728	0.702	0.589
N	445	445	964	964
Clusters	153	153	337	337

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

(a) Change to Novartis' Diclofenac Patent in 1993

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        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
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(b) Change Code CRLD

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

(c) Documentation Regarding RLD Changes

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

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.[26] 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.[27]

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

C.2 Examining the Reporting Accuracy of the CRS Data

One known limitation of CRS data in India is that many vital statistics events go unrecorded, and as a result, the CRS under-reports the true magnitude of

[26] Application can be filled out at: http://datazone.birdlife.org/species/requestdis
[27] BLI provides a summary of these categories here: http://datazone.birdlife.org/species/spcdistPOS

mortality. Although there is no alternative to the CRS as far as district-level data is concerned, at the national level a commonly used source of information is the Sample Registration System, which samples less than one percent of the population, but is designed to recover a nationally representative sample (Rao and Gupta 2020).

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

There is a clear difference in levels (Figure C1, dashed lines) between the all-cause death rate in the CRS relative to the SRS data. The SRS death rate is nearly double than the CRS reported death rate. However, when residualizing the death rates on a set of unit and time fixed effects (Figure C1, reported in the solid lines), the two death rates follow similar trends.[28]

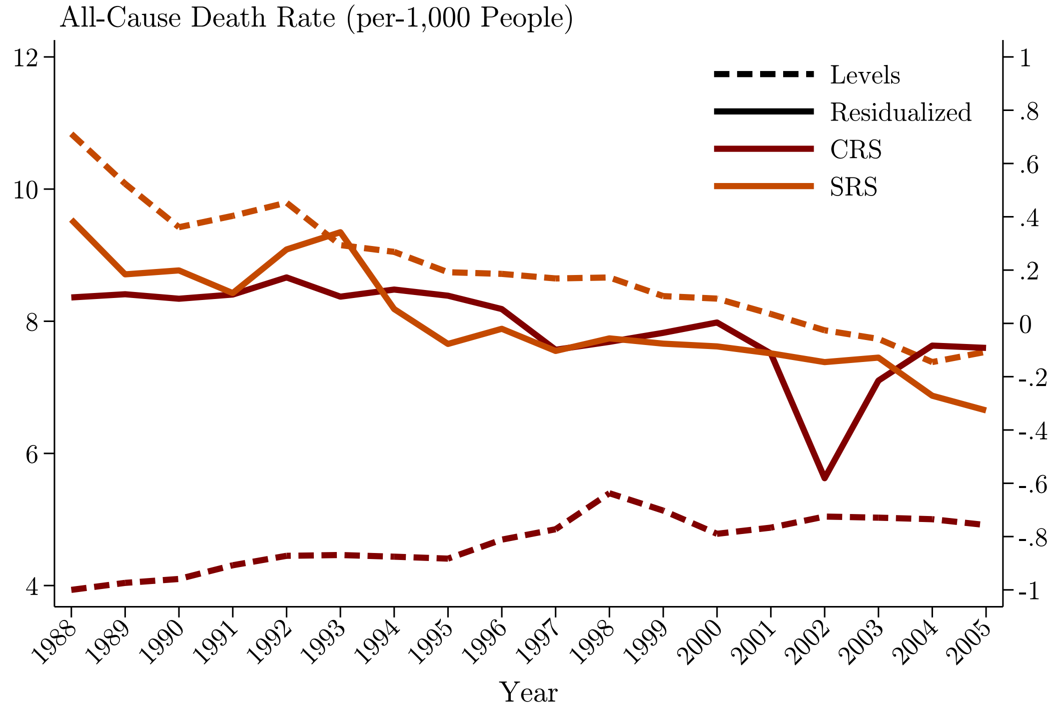
We interpret the agreement between the residualized levels in Figure C1

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

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

The fixed effect specifications we describe in Section 5 compare changes over time and are robust to several forms of under-reporting. This allows us to recover the level differences in mortality. Interpreting our level estimates *relative* to a baseline level of mortality, using the mean mortality reported in the CRS data is undesirable because it would over-estimate the size of relative changes. Consequently, in the interpretation of the analysis, we interpret the magnitude of the coefficients relative to the mean level from the SRS data as reported by the UN Population Division, which reflects 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.