American Economic Review 2024, 114(10): 3007–3040
https://doi.org/10.1257/aer.20230016

The Social Costs of Keystone Species Collapse: Evidence from the Decline of Vultures in India[†]

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Scientific evidence has documented we are undergoing a mass extinction of species, caused by human activity. However, allocating conservation resources is difficult due to scarce evidence on damages from losing individual species. This paper studies the collapse of vultures in India, triggered by the expiry of a patent on a painkiller. Our results suggest the functional extinction of vultures—efficient scavengers that removed carcasses from the environment—increased human mortality by over 4 percent because of a large negative shock to sanitation. We quantify damages at $69.4 billion per year. These results suggest high returns to conserving keystone species such as vultures. (*JEL* I12, O13, O15, Q53, Q57, Q58)

> [D]isgusting
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> —Charles Darwin, observing a vulture off the deck of the Beagle, January 1, 1835

We are in the midst of the sixth mass extinction in the history of the planet, likely induced by human activity (Ceballos et al. 2015). Since 1900, 477 vertebrate species have become globally extinct in the wild, at a rate about a hundred times higher than the "background" level estimated between the five previous mass extinctions (Pimm et al. 2014; Jaureguiberry et al. 2022). Local extinctions, where a species disappears from the wild in a part of the world, are even more common (Kuussaari et al. 2009; Wan et al. 2019). Well before local extinction, severely deteriorated

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[†]Go to https://doi.org/10.1257/aer.20230016 to visit the article page for additional materials and author disclosure statement(s).

wildlife populations may no longer be capable of filling their role in the ecosystem, resulting in what ecologists refer to as "functional extinctions" (Valiente-Banuet et al. 2015; Carmona et al. 2021).

These facts set the stage for a thorny policy challenge. Wildlife levels can collapse quite rapidly, with trajectories that are difficult to predict or reverse. Curtailing or regulating economic activity, or investing in conservation initiatives, might protect or restore some species populations. Unfortunately, since it is impossible to prevent every extinction, conservation policy must solve a crucial targeting problem: Which of the many endangered species should we protect or restore? This question is difficult to answer because although biodiversity loss is arguably damaging in general (Cardinale et al. 2012), estimates of the effects of losing *specific* species on human well-being are sparse.[1] Despite this lack of evidence, several policies focus on preventing the extinction of species. In the United States, leading examples are the Endangered Species Act, Marine Mammals Protection Act, Migratory Bird Treaty Act, and the Magnuson–Stevens Fishery Conservation and Management Act, with similar laws passed in other countries, for example, Natura 2000 in the European Union. Globally, nations have committed to the goal of preserving biodiversity by signing the Convention on Biological Diversity and establishing the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.[2] Without species-specific evidence of damages from extinction, policymakers find themselves in the undesirable situation of having to allocate scarce resources toward a few lucky winners, with little sense of the magnitude or even sign of the social benefits of their choices.

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

In this paper, we study the sudden and catastrophic collapse of vulture populations across the Indian subcontinent, making progress on all three fronts. First, we use a local functional extinction to study the costs to society of a catastrophic collapse of vultures in India, caused by the introduction of the painkiller diclofenac to treat cattle. The disappearance of vultures resulted in the loss of sanitation services that these birds had previously provided through scavenging dead livestock.

[1] In contrast, we know much more about the impacts of nonbiological 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 2016; Currie et al. 2015; Ebenstein et al. 2017; Deryugina et al. 2019; Keiser and Shapiro 2019; Marcus 2020) or changes in weather conditions (Schlenker, Hanemann, and Fisher 2006; Deschênes and Greenstone 2007; Deschênes, Greenstone, and Guryan 2009; Schlenker and Roberts 2009; Dell, Jones, and Olken 2014; Costinot, Donaldson, and Smith 2016; Fujiwara, Meng, and Vogl 2016; Hsiang et al. 2017; Proctor et al. 2018; Corno, Hildebrandt, and Voena 2020; Carleton et al. 2022).

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

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

We provide evidence of a meaningful increase in human mortality after vultures died out and were no longer removing carcasses from the environment. Although this analysis is retrospective, local functional extinctions are more easily reversed than global extinction in the wild, 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 versus low vulture suitability, before and after a near-total decline in bird populations due to an unintentional, unexpected, and rapid poisoning event in which vultures became exposed to the painkiller diclofenac. Habitat definitions in this setting provide an indicator for regions where the population change is expected to have been large.[4] We find that districts that were highly suitable to vultures saw an average increase in all-cause human death rates of 4.7 percent in the years following their sudden collapse. This number is measured relative to areas that were always poorly suited to vultures and thus much less affected. Our results hold up to multiple robustness checks and specifications and to an alternative triple-difference approach that exploits the fact that negative effects are likely to be concentrated in districts that had *both* vultures and large livestock populations. The effect size we obtain implies an average of 104,386 additional deaths a year relative to a population of 430 million people in our main sample. Using an India-specific value of statistical life of $665,000 (Nair et al. 2021), this implies mortality damages of $69.4 billion per year.

Lastly, the example of vultures suggests that one way to target evaluation, conservation, and protection efforts is to focus on what are known as *keystone species*: those that help "hold the [eco]system together."[5] Keystone species are seen as being crucial to the functioning of an ecosystem, sometimes providing unique services, such that if they are removed, the effects on the ecosystem are potentially large (Paine 1969; Power et al. 1996; Hale and Koprowski 2018). 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 (Ogada, Keesing, and Virani 2012). As vultures died out, the scavenging services they provided disappeared too, and carrion were left out in the open for long periods of time, creating a large negative sanitation shock.

Related Literature.—Our work links to several strands of the economics and ecology literature. We build on a theoretical foundation in ecology that explores how declines in species that perform important ecosystem functions can have effects

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

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

beyond their immediate ecosystem (Dirzo et al. 2014; Hooper et al. 2005; Estes et al. 2011; Martin et al. 2013; Ceballos et al. 2015; Watson et al. 2016; Luis, Kuenzi, and Mills 2018; Dainese et al. 2019; Schmeller, Courchamp, and Killeen 2020). We quantify the impact of a catastrophic shock to a keystone species with evidence on mechanisms. Economic theory shows that this type of estimate is essential for a meaningful cost-benefit analysis of conservation policy (Weitzman 1992; Solow, Polasky, and Broadus 1993; Weitzman 1993, 1998; Nehring and Puppe 2002; Brock and Xepapadeas 2003). Our approach offers an alternative to back-of-the-envelope approaches that have valued global ecosystem and natural capital at nearly twice the output of the global economy (Costanza et al. 1997). Such approaches have been criticized as an "Audacious bid to value the planet" (*Nature* 1998). Furthermore, our use of a natural experiment overcomes some of the limitations inherent to contingent valuation methods (Daily et al. 2000; Heal 2000), as discussed in Hanemann (1994) and Carson (2012).

We also join a nascent strand of the economics literature that has provided empirical evidence on the value of biodiversity. Using variations in environmental suitability, Alsan (2015) studied the long-term effects of the tsetse fly on agricultural production and political institutions. More recent papers study how farmers increase their use of insecticides to substitute for the loss of pest control following declines in insect-eating bats (Frank 2024), how air pollution increases after tree die-offs caused by the emerald ash borer (Jones and McDermott 2018), the importance of tree shade to human health (Jones 2019), and how reintroducing wolves can change the behavior of deer and reduce deer–vehicle collisions (Raynor, Grainger, and Parker 2021). Other related work in economics has focused not on the impacts that keystone species have on human well-being but on how technology and trade can play a role in their decline (Taylor 2011), how anticipated scarcity can lead to extinction (Kremer and Morcom 2000), or even actively promote extinction (Mason, Bulte, and Horan 2012).

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

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

The remainder of this paper is organized as follows. In Section I we describe the role of vultures as scavengers and outline the mechanisms through which their disappearance might impose costs on society, followed by the cause of the sudden population collapse of vultures in India. In Section II we describe the sources of data we use in this paper. In Section III we outline the econometric approach we use and present different specifications that we take to the data. In Section IV we present our estimates of the mortality impacts of losing vultures. We also present supporting evidence on the hypothesized mechanisms and a summary of different robustness checks and alternative specifications. In Section V we benchmark the effects of losing vultures against other environmental or sanitation shocks and include an assessment of the costs of replacing their ecosystem services with technology (incinerators). We conclude in Section VI.

I. Vultures as Ecosystem Sanitizers

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

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

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

Livestock agriculture also becomes a source of water pollution once farmers need to dispose of dead animals themselves (Engel et al. 2004; Kwon et al. 2017).

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

[8] As Dr. Asad Rahmani, Director of the Bombay Natural History Society, put it, "Now there are dogs. They eat anything, live or dead. There are dogs on the ground but the skies are empty" (Subramanian 2011, p. 47).

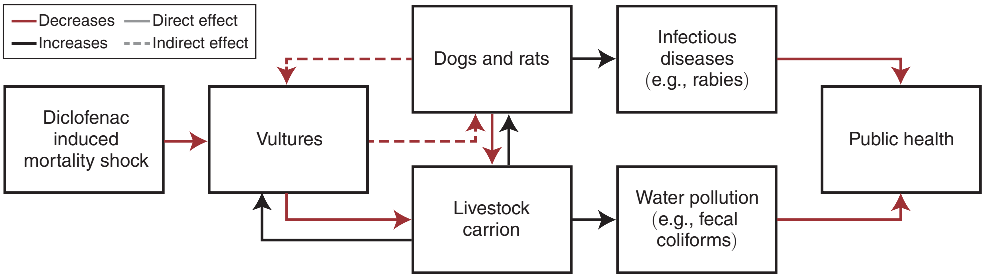

Figure 1. Schematic Relationship of Ecosystem Interactions and Environmental Quality

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

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, p. 42).

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

We summarize the interactions between vultures, mammalian scavengers, environmental quality, and public health in Figure 1. Within the ecosystem interaction group of vultures, mammalian scavengers (dogs and rats), and livestock carrion, the former two are competing for the food source (dead animals). Greater availability of carrion supports larger populations of both scavenger types, efficient (vultures) and inefficient (dogs and rats). Because both types compete for the same food source, each type indirectly limits the population growth of the other type.

In the absence of vultures, livestock farmers and municipalities can utilize either labor-intensive or capital-intensive substitutions. Farmers can exercise deep burial, but given the number of livestock animals, this adds high labor costs. Since these costs are private, while the costs of disposing of animals in carcass dumps or water are socialized, it is not surprising that deep burial remains uncommon. Livestock carrion can be disposed of using specially designed incinerators, yet they are expensive to buy and operate and require a reliable mechanism for making sure that farmers transport dead animals to them. According to a 2020 report by India's Central Pollution Control Board, India has yet to adopt livestock incinerators as a substitution for vultures: "Very few cities have carcass utilization plants and incinerators. One such carcass utilization plant is installed in Delhi and incinerator is under installation in Chandigarh" (Central Pollution Control Board 2020, p. 10).

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

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

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

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

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

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

Reports of vulture declines rapidly followed the veterinary use of diclofenac. Field observations in 1996 found only half of the 353 nesting vulture pairs recorded in 1984 in Keoladeo National Park outside Delhi (Subramanian 2011). After Dr. Vibhu

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

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

Prakash, at the time a PI in the Bombay Natural History Society, communicated his findings, colleagues reported similar patterns they thought were simply idiosyncratic to their study sites. Population declines were so rapid that in 2000, all three species were classified as critically endangered. The Indian government eventually banned the veterinary use of diclofenac in 2006 (Prakash et al. 2012; Ogada, Keesing, and Virani 2012). However, surveys conducted up to 2018 document rampant illicit use of diclofenac in livestock, including by diverting human doses (Galligan et al. 2020). As a result, vulture populations in India have never recovered.

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

II. Data

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

A. *Vulture Habitat Ranges*

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

Therefore, to determine where vultures used to exist, we obtain maps from BirdLife International (BLI) on the species distribution ranges of all bird species (BirdLife International and Handbook of the Birds of the World 2018). We extract the range maps for vulture species and perform two spatial calculations with the 1981 district boundaries (GADM 2018): (i) whether the district intersects with the range map and (ii) the area of overlap between the range map and the district (see online Appendix Figure A3 for a summary of the distribution of these values). We use the area of overlap to calculate the share of area for each vulture species in each district. Our approach assigns each district a suitability category for diclofenac-affected vultures by dividing the mean overlap of species ranges into terciles. This provides us with a proxy for the abundance of vultures and their prevalence across the district. Figure 2 shows the spatial distribution of the classification into high- and low-suitability categories for diclofenac-affected vultures. In online Appendix Section A.2, we provide an extensive review of the ecology literature as

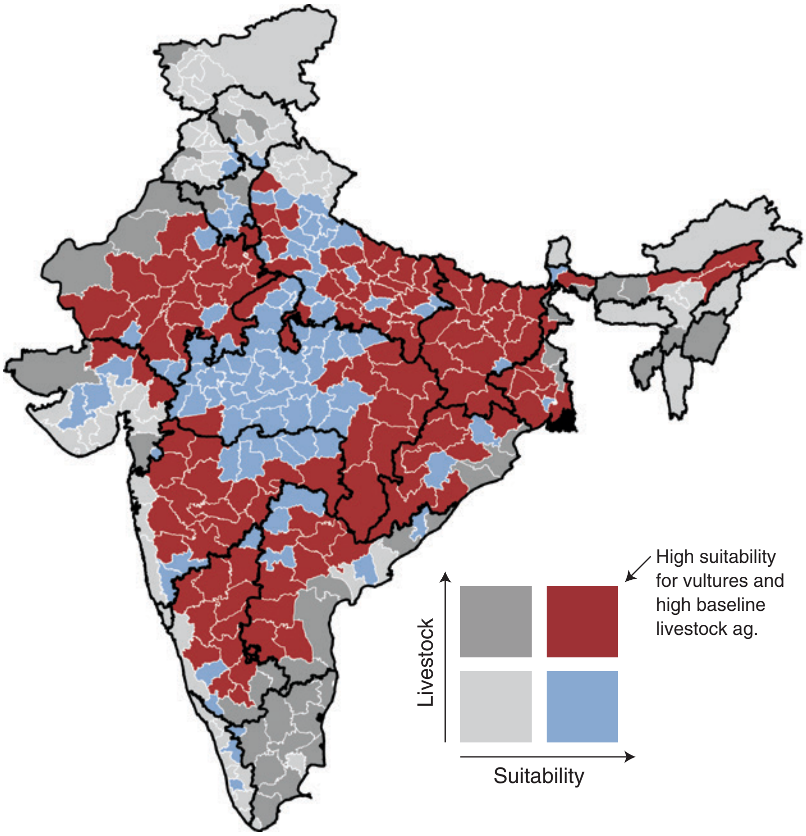

Figure 2. Spatial Distribution of Diclofenac-Affected Vulture Ranges and Livestock Agriculture

Note: 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 II for more details).

well as a set of original validation exercises used to confirm the quality of this proxy. Briefly, we collect data on over 400 bird species in North America for which *both* population counts and habitat range maps are available. We recalculate our habitat overlap measures for each of these species and find a tight relationship between habitat overlap and population counts. An additional benefit of this approach is that it is less dependent on functional form assumptions previously used in the economics literature to relate environmental suitability to outcomes of interest (Alsan 2015).

Finally, although we use habitat suitability scores for our empirical specifications, it is possible to gain some sense of how vulture populations changed by relying on citizen science reports. The Global Biodiversity Information Facility (GBIF) database (GBIF 2024) aggregates multiple reporting sources of data, including some scientific studies and citizen science reports.[11] We calculate the share of reports of diclofenac-affected vultures relative to other bird species that have nonzero observations each year from 1990 to 2005. Figure 3, panel B shows a decline in this

[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).

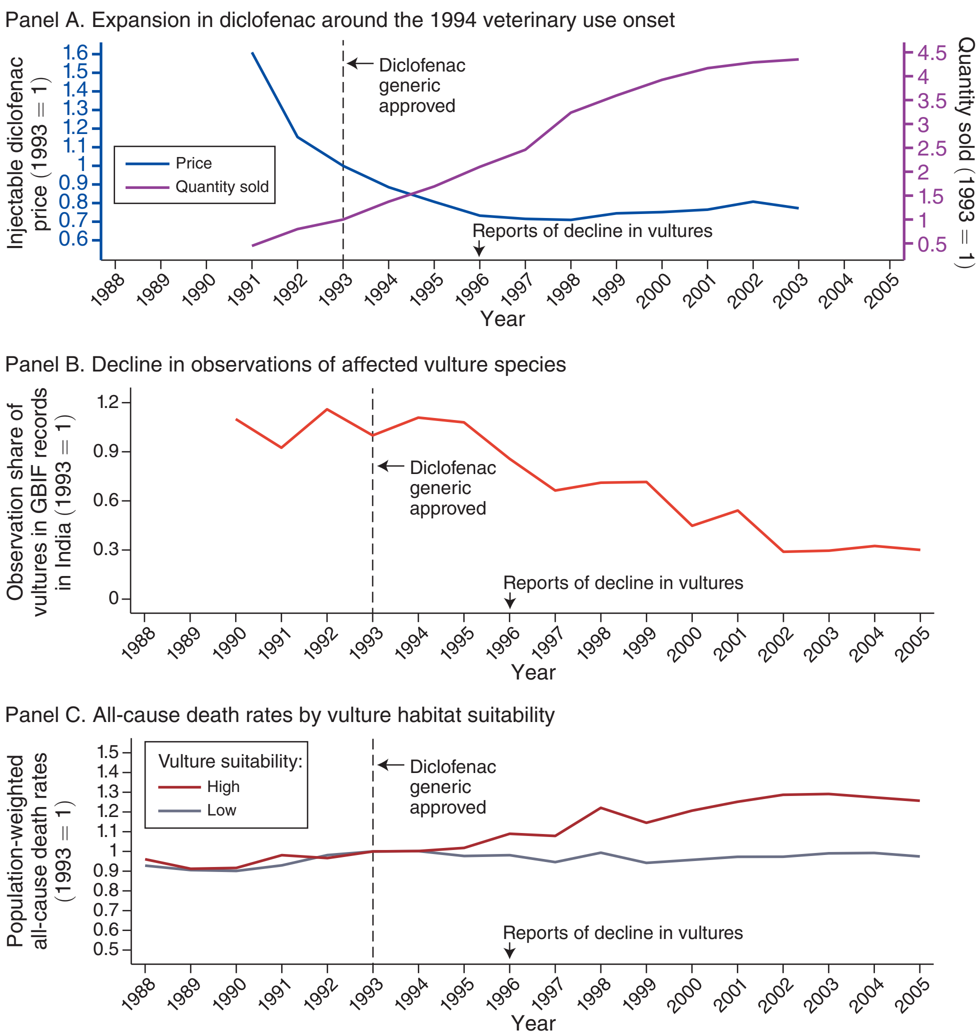

Figure 3. National Trends in Diclofenac Use, Vulture Observations, and Death Rates

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

share, with a trend break that follows the veterinary use of diclofenac in 1994. Unfortunately, these data cannot be used for reliable empirical estimates of the rate of decrease of vultures because once it became known that they were growing rare in the wild, bird enthusiasts would have dedicated more effort to documenting residual birds. In the online Appendix, we add a second piece of indicative evidence of the decline of vultures by reproducing a set of survey results that counted vultures along 70 road transects 5 times between 1992 and 2007 (Prakash et al. 2007). In online Appendix Figure A1, we plot the data from these surveys; they show a decline by about three orders of magnitude over this period.

B. *Sales and Product Entry of Pharmaceuticals in India*

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

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

C. *Health Outcomes*

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

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

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

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

their 1988 to 1993 trend, low-suitability districts maintain the same mean death rate from 1988 to 2005.[13]

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

D. *Livestock Census*

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

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.[14] If dog populations are higher in the high-suitability areas for diclofenac-affected vultures, then that is consistent with the anecdotal evidence regarding the increase in feral dogs, animal bites, and rabies cases.

E. *Water Quality*

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

F. *Additional Environmental and Demographic Data*

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

III. The Collapse of Vultures in India as a Natural Experiment

We turn now to our empirical approach. To estimate the causal effect of the collapse in vulture populations on public health, the ideal experiment would randomly assign vultures to different districts across India. This ideal experiment

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

[14] As Markandya et al. (2008, p. 198) 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."

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

A. *Difference-in-Differences Design*

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

Mortality Effects over Time.—We estimate the following event-study-like regression specification:

$$y_{daszt} = \sum_{\substack{\tau \in \{\underline{T}, \ldots, \bar{T}\} \\ \tau \neq 1993}} \beta_\tau (HVS)_d \times \mathbf{1}\{t = \tau\} + \lambda_{da} + \delta_{zt} + \mathbf{X}_{daszt}\boldsymbol{\theta} + \varepsilon_{daszt}. \tag{1}$$

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 is a dummy variable that equals one for districts that we classify as having a high precollapse presence for the three vulture species affected by the exposure to diclofenac, and zero otherwise. We define high presence as being in the top and middle terciles of our habitat suitability index, constructed using the overlap between vulture ranges and district areas and described in more detail in Section II (see Figure 2). We interact the treatment variable with year dummies, with 1993 as the baseline (omitted) year since that is when the use of veterinary formulations began (see Section I for details).

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

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

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

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

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

These fixed effect designs also help adjust for known underreporting 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 online Appendix, we use an alternative source of more aggregated vital statistics data from India's Sample Registration System (SRS) to show that although the CRS underestimates mortality rates by about a factor of two relative to the SRS, after controlling for state and zonal council-by-year fixed effects, both sources of data allow us to recover similar trends in mortality rates. When reporting estimates in percentage terms, we use the nationally representative baseline

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

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

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

mean of all-cause death rates in deaths per 1,000 people between 1988 and 1992 of 10.2 for the entire country, and 7.2 for the census urban area (see online Appendix Section C.5 for additional details).

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

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

$$(2)\quad y_{daszt} = \beta(HVS)_d \times \mathbf{1}\{t \in [1994, 1999]\}_t + \beta(HVS)_d \times \mathbf{1}\{t \in [2000, 2005]\}_t + \lambda_{da} + \delta_{zt} + \mathbf{X}_{daszt}\boldsymbol{\theta} + \varepsilon_{dszt}.$$

B. *Heterogeneity in Effect of Vulture Loss*

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

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

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

$$(3)\quad y_{daszt} = \beta(HVS)_d \times \mathbf{1}\{t \geq 1994\}_t \times (High\ Livestock)_d + \lambda_{da} + \delta_{zt} + \mathbf{X}_{daszt}\boldsymbol{\theta} + \varepsilon_{dszt}.$$

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

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

IV. Results

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

A. *Comparing High- and Low-Suitability Districts*

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

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

On other covariates, we should expect districts with high versus 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 health care as measured

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

Vulture suitability	Group means *Low* (1)	*High* (2)	Δ:(2)–(1) (3)	Observations (4)
All-cause death rate[a,b]	5.3	4.2	−1.2	153
	(1.8)	(1.8)	(0.32)	
Degree days above 30°C[a]	54	66	12	153
	(43)	(35)	(6.8)	
Precipitation (mm·km^{-2})[a]	0.25	0.12	−0.12	153
	(0.42)	(0.18)	(0.044)	
Baseline livestock[c]	1.6	1.6	0.028	153
	(0.87)	(0.73)	(0.15)	
log(*dissolved oxygen*)[a]	1.9	1.9	0.0045	95
	(0.18)	(0.27)	(0.047)	
log(*fecal coliform*)[a]	7.2	7.4	0.25	76
	(2.2)	(1.7)	(0.48)	
Pop. share [1, 24][d]	0.42	0.51	0.097	145
	(0.14)	(0.08)	(0.023)	
Pop. share [25, 54][d]	0.29	0.33	0.035	145
	(0.098)	(0.058)	(0.016)	
Pop. share [55, 100][d]	0.083	0.088	0.0056	145
	(0.029)	(0.018)	(0.0048)	
Share literate[d]	0.55	0.41	−0.14	143
	(0.13)	(0.12)	(0.021)	
Water taps[d,e]	11	13	1.1	145
	(27)	(21)	(2.7)	
Water wells[d,e]	23	57	34	145
	(25)	(42)	(6)	
Hospitals and health centers[d,e]	1.7	2.4	0.73	145
	(1.7)	(2.5)	(0.34)	
Doctors and health workers[d,e]	8.1	9.8	1.7	145
	(7.6)	(8.6)	(1.5)	

Notes: Districts with balanced death rates, 1988–2005. Observations are population weighted. Robust standard errors are in parentheses.
[a] Averaged between 1988 and 1993.
[b] Per 1,000 people.
[c] Values, in millions, for 1987 and/or 1992.
[d] Value for 1991.
[e] Per 100,000 people.

by the number of hospitals and health centers as well as doctors and health workers. This comparison helps to rule out the possibility of preexisting differences in water or health care 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.

B. *Results for All-Cause Death Rate*

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

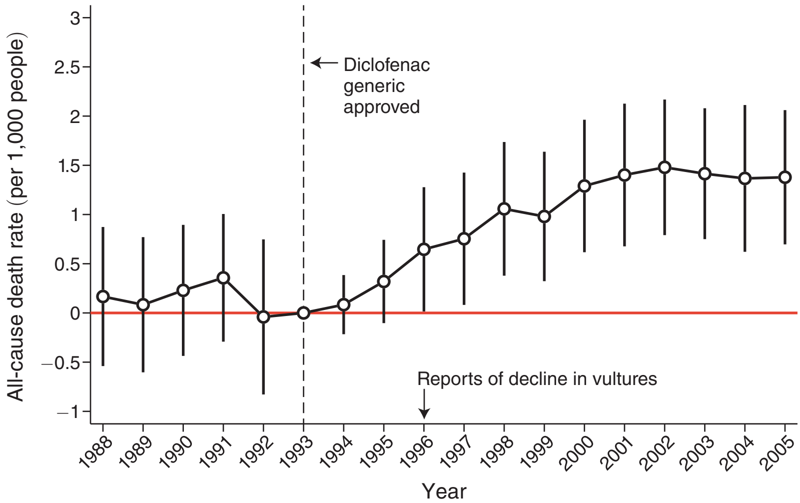

Figure 4. All-Cause Death Rates DD Estimation Results

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

Following the onset of diclofenac use after 1993 and the first observed signs of large-scale decline of vultures in 1996, we find that death rates from all causes increased in the high-vulture-suitability districts. In 1996, the first year in which the decline in vulture populations gained widespread recognition, the all-cause death rate was higher in the high-suitability districts by 0.65 deaths per 1,000 people. By the end of the sample, in 2005, death rates were higher by about 1.4 deaths per 1,000 people. These reflect an increase of 6.4 percent and 13.7 percent relative to the nationally representative mean level of 10.2 deaths per 1,000 in the pretreatment 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 Figure 3, panel B and online Appendix Figure 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

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

	Combined sample ($\bar{Y}$ = 10.2)				Census urban sample ($\bar{Y}$ = 7.2)			
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Panel A. Without weather controls								
HVS × Post-1994	0.91				1.04			
	(0.14)				(0.27)			
HVS × [1994, 1999]		0.52	0.13	0.21		0.68	0.35	0.34
		(0.15)	(0.15)	(0.14)		(0.30)	(0.26)	(0.22)
HVS × [2000, 2005]		1.26	0.48	0.40		1.34	0.68	0.63
		(0.19)	(0.16)	(0.16)		(0.30)	(0.23)	(0.24)
R^2	0.74	0.75	0.77	0.80	0.67	0.68	0.70	0.76
Observations	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								
HVS × Post-1994	0.85				1.04			
	(0.15)				(0.25)			
HVS × [1994, 1999]		0.51	0.18	0.19		0.72	0.40	0.32
		(0.15)	(0.15)	(0.15)		(0.29)	(0.26)	(0.22)
HVS × [2000, 2005]		1.17	0.45	0.38		1.32	0.67	0.64
		(0.19)	(0.17)	(0.17)		(0.25)	(0.22)	(0.25)
R^2	0.75	0.75	0.78	0.81	0.68	0.69	0.71	0.76
Observations	2,754	2,754	2,754	2,700	2,808	2,808	2,808	2,754
Clusters	153	153	153	150	156	156	156	153
Zonal council-by-year FE	X	X	X		X	X	X	
State-linear trends			X				X	
State-by-year FE				X				X

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

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 macroeconomic factors.

One concern we may have is the possibility of differential reporting of death rates in high- versus low-suitability districts beginning after 1994 that may not be fully captured by zonal trends. To control for this, in Table 2, panel A, column 3, we use a specification that includes linear time trends for each state, which is the level at which the civil registry reporting system is administered. These controls soak up some of our variation, in particular in the period where treatment effects are also growing over time. However, our finding for equilibrium outcomes remains qualitatively similar, with a fairly precise point effect of 0.48 additional deaths per 1,000 people. This reflects a 4.7 percent increase relative to the nationally representative mean level between 1988 and 1992 of 10.2 deaths per 1,000 people, as reported in the SRS data.[19] We regard this as our preferred specification for estimating equilibrium elevated death rates due to the disappearance of vultures. Finally, we report results using state-by-year fixed effects in column 4. This absorbs more variation, but our results remain broadly similar.[20]

As we discuss in Section IIIB, urban areas might have faced a larger sanitation shock due to their proximity to carcass dumps, significantly higher population density, and network infrastructure such as drains allowing pathogens and waste to spread rapidly. Using the urban-rural breakdown of reported district death rates, we reestimate all models for urban areas only and report results in Table 2, columns 5–8. Across all specifications, we find that urban areas experienced a larger increase in death rates relative to the combined sample.[21] For our preferred specification including state-linear trends (columns 3 and 7), urban death rates increase by 0.68 per 1,000 people after reaching equilibrium (2000–2005). This compares with an estimate of 0.48 in the combined sample.

C. *Long-Difference Models*

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

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 posttreatment periods to address uneven reporting in those periods. The important modification is that we limit the sample to a pretreatment period of 1990 to 1995 and a single posttreatment period of 2000 to 2005. With a relaxed requirement that districts only have nonmissing data in these two periods, we are able to include as many as 324 districts (relative to 153) in combined urban and rural specifications and as many as 279 districts (relative to 156) when separating urban areas.

The results remain similar to those from the fully balanced panel. In Table 3, we report the results from estimating the long differences model, similar to the specification in equation (2). Across the larger sample that uses data from almost

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

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

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

[22] Many districts are missing at least a year of data, and in the case of the state of Uttar Pradesh, we are missing data for all districts from 1996 to 1999. We went through considerable efforts to fill in any missing years of data. See the online Appendix for full documentation.

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

	(1)	(2)	(3)	(4)	(5)	(6)	(7)
Panel A. Combined sample							
($\bar{Y}$ = 10.2)							
HVS × Post-2000	1.23	0.72	0.68	0.26	0.16	0.17	0.16
	(0.22)	(0.19)	(0.23)	(0.15)	(0.14)	(0.14)	(0.14)
R^2	0.72	0.73	0.85	0.77	0.90	0.79	0.90
Observations	1,836	3,696	648	3,696	648	3,589	628
Clusters	153	324	324	324	324	314	314
Panel B. Census urban sample							
($\bar{Y}$ = 7.2)							
HVS × Post-2000	1.23	1.04	1.01	0.65	0.62	0.61	0.62
	(0.27)	(0.25)	(0.37)	(0.17)	(0.19)	(0.17)	(0.19)
R^2	0.64	0.65	0.84	0.69	0.90	0.75	0.90
Observations	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 fixed effects	X	X	X	X	X		
State-linear trends				X	X		
State-by-year fixed effects						X	X
Collapsed sample			X		X		X

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

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, panel A, column 3).

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

D. *Investigating the Role of Livestock*

We turn next to the role of livestock in increasing the value of the sanitation services provided by vultures. In Table 4, we report results from the triple-differences specification in equation (3). We find that following the collapse in vulture populations, high-vulture-suitability districts that also had a high level of livestock at

Table 4—DDD Results for All-Cause Death Rate

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

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

baseline showed a significantly higher increase in death rates, relative to districts with below-median livestock populations.[23] This gap widens further when restricting the sample to urban areas (Table 4, columns 3 and 4). These results are consistent with the hypothesis that the main driver of mortality after the collapse in vulture populations is the presence of a large supply of animal carrion that is not effectively scavenged, rather than simply the decline in vultures themselves.[24] We present results from decomposing the triple-differences into two DD comparisons in online Appendix Table A3, showing that the interaction of high livestock with postcollapse has a meaningful effect on the all-cause death rate only in the high-vulture-suitability subsample.

E. *Sanitation Channels: Dogs*, *Rabies*, *Water Quality*

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

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

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

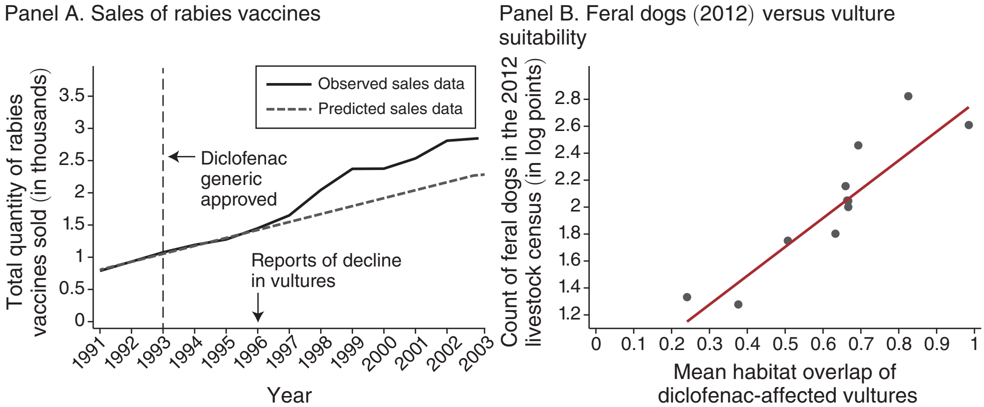

FIGURE 5. SUGGESTIVE EVIDENCE FOR FERAL DOG MECHANISM

Notes: Panel 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. Panel B: District-level data on feral dogs were counted for the first time during the 2012 livestock census.

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 a lifesaving treatment after an animal bite, although there are sadly many people in India who still die from rabies because they delay reporting to hospitals.[25] In Figure 5, panel A, 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

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

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

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

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

F. *Sensitivity Analysis and Robustness Checks*

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

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

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

Table 5—District Water Quality DD and DDD Estimates

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

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

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

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

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 online Appendix Figure A9 and report the average treatment effects in online Appendix Table A7. For both of the alternative classifications, we estimate an increase of more than 0.5 deaths per 1,000 people. This analysis confirms that our results are not driven by a specific functional form for the vulture suitability and that the results are not sensitive to the exact definitions of the treatment and control groups.

We further examine the sensitivity of the results to compositional changes in the sample by estimating two leave-one-out versions of the DD specification in equation (2). Specifically, we either omit one district at a time or one state at a time. We plot the resulting narrow distribution of the estimated treatment effects in online Appendix Figures A13 and A14. Lastly, we perform a permutation inference analysis, where we randomly assign treatment status and reestimate 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 nonrandomly assigned treatment is in the right tail of the distribution.

V. Benchmarking Mortality Effects

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

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

The literature supports large improvements in mortality for other interventions that improve water and sanitation, just as we might expect vultures to do. Geruso and Spears (2018) estimate a reduction in infant mortality rate in India by 8 percent for a 10 percent decrease in open defecation. In the context of privatizing water provision to improve sanitation and quality, Galiani, Gertler, and Schargrodsky (2005) find that child mortality drops by 8 percent, on average, and as much as 26 percent in the poorest regions. Cutler and Miller (2005) estimate an even larger drop, of

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

43 percent, in infant mortality rates from the improvements to water quality in US cities around 1900. In Mexico, where water chlorination went up from 58 percent to 90 percent, Bhalotra et al. (2021) find that child mortality dropped by 45 percent to 67 percent. These comparisons are tabulated in online Appendix Table D1.

Other environmental risk factors such as pollution have also been found to have large effects on mortality. Ebenstein et al. (2017) suggest that China's policy of providing free heating coal increased all-cause mortality by 20–26 percent. Tanaka (2015) finds that air pollution regulations instituted in Chinese provinces in 1998 reduced infant mortality by 20 percent. Carleton et al. (2022) study the mortality effects of exposure to future high temperatures due to climate change. One of the countries projected to be most negatively affected by heat deaths is India. The estimates in this study suggest an increase in death rates by 0.6 per 1,000 in 2099 under an RCP 8.5 warming scenario (a relatively pessimistic "business as usual" projection of future emissions and warming). This is comparable in magnitude to our estimate of a 0.48 increase in deaths from losing the sanitation services provided by vultures. Of course, deaths due to heat exposure are only one aspect of climate costs and mortality due to climate change, but the comparison is nevertheless striking and underscores the importance of keystone species to human welfare.

Incinerator Costs.—A third way to think about these damages is to consider what it would cost to avoid them. The most straightforward alternative to vultures is to build out a network of incinerators (carcass-rendering machines) to dispose of livestock carcasses. Ishwar et al. (2016) carry out a detailed analysis of the costs of operating mechanical incinerators using data from 2014 to 2015. 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 ₹8,346,097 ($\sim$ \$139,000).

In 2019, India's livestock population was over 500 million, with about 300 million of those being cattle (twentieth Livestock Census). Although it is illegal to slaughter cows in India, they do not survive long after their productive life as milch animals because farmers may set them free, effectively denying them access to sufficient food or medicines. Assuming an average life-span of about 10 years suggests an annual burden of about 30 million cow carcasses alone. This number suggests annualized costs of operating a nationwide network of carcass-rendering machines of about \$768 million (in 2014–2015 US dollars), solely for cows. This estimate ignores air pollution damages from the incinerators.

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

Vulture Recovery.—Finally, we might wonder what it would cost to bring back vultures. We do not venture to place a monetary cost on this option for two reasons. First, a key element of any such recovery would be a successful ban on diclofenac and its derivatives. The leading alternative to this drug is Meloxicam, which is similarly priced but takes much longer than diclofenac to be effective in cattle (roughly 4 hours against 15 minutes). Second, the most significant hurdle involved in restoring vultures to the point where they might once again provide these services is the time it would take. Vultures, much like humans, reproduce relatively slowly. They mate for life, reach sexual maturity at five years, and lay only one or two eggs each year.

VI. Conclusions

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

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

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

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

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

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

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

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