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

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Abstract

Losses of keystone species that affect environmental quality through their ecosystem interactions can have large effects on social costs. However, crucial parameters for the management of their preservation are often not available. Determining an optimal recovery strategy requires knowing the benefits lost in their absence, defensive expenditures linked to their loss, as well as the direct rehabilitation costs. We study the above in the setting of vultures that serve a major public health role by preventing the spread of infectious diseases. Vulture populations fell in the Indian subcontinent due to the presence of a chemical residue in livestock carrion. The use of the chemical painkiller in livestock animals became widespread after its patent expired and generic versions of the drug made it widely accessible for veterinary uses. Using distribution range maps for the affected vulture species, we compare districts before and after the collapse in vulture populations. We estimate all-cause death rates increased, on average, by over three percent in the highly-vulture-suitable districts after vultures nearly went extinct. We find suggestive evidence that following the vulture die-offs, feral dog populations increased, and that water quality deteriorated in the affected regions.

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1 Introduction

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

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

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

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

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

The setting of the vultures in India provides a vivid example of biodiversity loss as an unforeseen externality. Once too numerous to count, with a population in the tens of millions, the number of birds in the wild fell by over ninety-five percent within a few short

years in the second half of the 1990s. Today just a few thousand vultures survive in India, with the three primary species all critically endangered. The historic presence of large and stable vulture populations simultaneously reduced pathogen and bacteria concentrations in the environment, as well as crowded out other mammalian scavengers such as dogs and rats that transmit various infectious diseases (Moleón et al. 2014).

The ecological and epidemiological dynamics of scavengers, pathogens, and infectious diseases help explain the causal link between diminishing vulture populations and human health. While some animal species will feed on carrion if available, for vultures, it is the only source of food. As a result, vultures have evolved as very efficient scavengers. Vultures have an extremely acidic stomach, that ranges from just above zero to two pH.[1] In comparison, an average human has a pH level of two in their stomach, making it ten to a hundred times less acidic than that of a vulture. This is one of the key adaptations that allows vultures to safely consume carrion, and also results in most bacteria not surviving their digestive system (D. L. Ogada et al. 2012; Roggenbuck et al. 2014).

The removal of carrion from the environment by vultures becomes more important in low to middle income countries where these birds have effectively substituted for expensive infrastructure to safely dispose of animal carcasses. In India, the interaction of widespread dairy cultivation with cultural practices regarding dead animals has resulted in a historically large reliance on scavengers. The sanitizing function vultures perform prevents livestock carcasses from rotting in open fields, and from transmitting their pathogens and diseases, such as anthrax, to other scavengers. Deadly pathogens can also get eroded by surface runoff and end up in the drinking water supply (Vijaikumar et al. 2002; R. T. Watson et al. 2004; Markandya et al. 2008; D. Ogada et al. 2016). Restricting the amount of carrion and time it remains in open fields is of specific importance in India because of prevailing social norms regarding the handling of meat. Hindus will not consume cows whereas Muslims will not consume animals not killed according to *halal* (Subramanian 2011).

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

While other scavenging species exist, they are imperfect substitutes for vultures. Vultures are extremely effective at reducing a carcass to its bones, and can consume the carrion of an entire cow within forty minutes (D. L. Ogada et al. 2012). Non-vulture scavengers are just not as effective, as they tend to leave some of the flesh behind (Subramanian 2011). Recent experimental evidence confirms that non-vulture species are not able to compensate and functionally replace vultures in terms of scavenging efficiency (Hill et al. 2018). In the absence of vultures, the composition of species that feed on carcasses changes towards species that are a reservoir of pathogens and diseases to which humans are susceptible to.

The limited availability of infrastructure such as animal incinerators have led to so called "animal landfills" on the outskirts of population centers across India. Anecdotal accounts describe how with vultures no longer available, the rotting meat and its scent builds up, attracting feral dogs (Subramanian 2011). Dogs and rats are vectors of infectious diseases and much less efficient scavengers than vultures, a combination that makes carcass dumps a breeding ground for disease (D. L. Ogada et al. 2012).

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

Because vultures were so abundant across India it was never deemed necessary to survey them and maintain a population count database. In the absence of data on the population densities, we leverage habitat suitability to infer treatment intensity. We classify districts into vulture suitability terciles using well-established species range distribution maps produced

by BirdLife International (BLI), considered the global authority on birds and their habitats. Explicitly, we overlap each district with the habitat range maps for each species, calculate the mean area overlap for the three diclofenac-affected vultures, and divide into terciles. This is similar to how Alsan (2015) uses suitability for the tsetse fly in Africa, but our approach does not rely on strict functional form assumptions.

Using a difference-in-differences strategy, we compare high to low-suitability districts, before and after the onset of diclofenac use. In our analysis, we focus on district level data, reported separately for urban and rural areas, on death rates from all causes. We also examine infant mortality rates, and birth rates. Our main finding is an increase of 0.3 deaths per-one-thousand people, reflecting an increase of over three percent, relative to a nationally representative mean. We do not find strong evidence for an increase in infant mortality. However, we do observe an increase in birth rates, suggestive of changes in fertility decisions in order to replace children lost to rabies or other infectious diseases.

To further investigate the mechanisms through which a decline in vulture can affect public health, we study the channel of dog population densities. Attacks by feral dogs are common, and they mostly represent an immediate deadly threat to small children. However, with India being a global epicenter for rabies, any animal bite can result in death (Subramanian 2011). Anecdotal accounts often describe feral dog populations increasing by a third after competition over food sources with vultures subsided.[2] As part of the livestock census, India records counts of cattle, sheep, and other domesticated farm animals, as well as counts of dogs. Using the same research design, we estimate, albeit imprecisely, that dog population were around eight to twenty percent higher in the years after the onset of diclofenac use in the high-suitability districts.

As previous work has demonstrated, livestock agriculture acts as a source of water pollution, especially when needing to dispose of dead animals (Engel et al. 2004; Kwon et al. 2017). We use data on annual measurements of water quality and find evidence of de-

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

graded water quality in most affected regions in our sample, across a variety of outcomes that have been linked to animal agriculture, and the absence of scavengers (Swift et al. 1979; Santori et al. 2020; Brundage 2021). We find more than a doubling of fecal coliforms in water bodies around urban areas, indicating potential leeching of additional pathogens and bacteria into drinking water.

The analysis in this paper relies on the collapse in vultures to be unforeseen, and not a direct consequence of our outcomes of interest. These assumptions are supported by the fact that the use of diclofenac expanded after its patent expired in 1993, and the connection to the demise of vulture was only made in 2004. A remaining potential threat to our identification strategy is differential reporting of mortality that is systematically correlated with the timing and location of the decline in vultures. To reject that reporting is increasing more in the high-suitability regions death rates, we verify the findings are robust to the inclusion of state-linear time trends. As differential pre-trends could also mean that our key identifying assumption is violated, we include results from a longer panel with more time periods, yet fewer balanced districts. Even in the extended sample, we do not observe systematically different pre-trends in death rates across the different vultrue suitability categories.

Our work adds to a growing body of literature in economics that uses quasi-experimental settings to estimate the social costs of biodiversity losses. This builds on a theoretical foundation in ecology that explores how declines in species that perform important ecosystem functions can have effects beyond the interactions within the ecosystem (Dirzo et al. 2014; D U Hooper et al. 2005; Estes et al. 2011; Martin et al. 2013; Ceballos et al. 2015; J. E. M. Watson et al. 2016; Luis et al. 2018; Dainese et al. 2019; Schmeller et al. 2020). Drawing on variation in ecosystems to construct research designs, these recent empirical findings demonstrate how key predictions from ecology can be tested. The resulting well-identified evidence contributes quantified estimates that previous theoretical work in economics highlighted as important for the optimal management of biodiversity as a resource (Weitzman 1992; Solow et al. 1993; Weitzman 1993; 1998; Nehring and Puppe 2002; Brock and Xepapadeas 2003).

Earlier work by Banerjee et al. (2010) used an invasive species as an income shock to study the effect it has on adulthood outcomes. Using variation in environmental suitability, Alsan (2015) studies the long-term effects that result from the presence of the tsetse fly on agricultural production and political institutions. Lange et al. (2009) and Bloome et al. (2017) study the economic and demographic implications of the invasion of the Boll Weevil, a crop pest that plagued cotton fields in the US South.

More recent papers study how farmers increase their use of insecticides to substitute for the loss of pest control following declines in insect eating bats (Frank 2021); how air pollution increases after tree die-offs caused by the emerald ash borer (Jones and McDermott 2018); how insecticide use increases in years in which cicadas emerge (Taylor 2021), and exploit the change in insecticide use to study their health impacts (Frank 2021; Taylor 2021); and how reintroducing wolves can change the behavior and density of deer and reduce deer-vehicle-collisions (Raynor et al. 2021). While the exact source differs, these studies exploit a plausibly exogenous shock to an ecosystem to learn about the causal treatment effect of a species of interest.

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

In what follows, we summarize the conditions that led to the decline of vulture popula-

tions, and proceed to describe the use of the data in our empirical strategy, along with the key findings of the analysis.

2 The Sudden Collapse of Vulture Populations in India

Vultures were once an ubiquitous sight across India. Their collapse is the fastest of a bird species in recent history and the largest in magnitude since the extinction of the passenger pigeon in the US. While several vulture species are still present in India, the three that made up the bulk of the population are considered critically endangered after declining by more than ninty-five percent from population counts in the millions.[3] Their collapse was due to chemical residue of diclofenac, a painkiller used by livestock farmers, in cattle and sheep carcasses. Within weeks, vultures that fed on a carcass with such residue would develop kidney failure and die.[4]

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

The sudden adoption of diclofenac among livestock farmers was due to the emergence of a generic version of the drug. Anecdotal accounts place the timing of the patent expiration in the early 1990s (Subramanian 2015). In order to more precisely determine the onset of diclofenac use, we draw on additional sources of data. We start with formal records regarding the patent and its expiration. The patent originally belonged to the pharmaceutical company

[3] The three common names (and scientific names) of the three affected vulture species are: slender-billed (*Gyps tenuerostris*), white-backed (*Gyps bengalensis*), and long-billed (*Gyps indicus*). There is one additional member of the gyps genus, Himalayan Griffon (*Gyps himalayensis*). However, as their name suggests, they are mostly found in the Himalayas, where they do not depend on livestock carcasses that have diclofenac residue that caused the collapse in the other species.

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

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

Novartis. Using documents from the Federal Drug Administration regarding drug patents, we are able to trace its expiration to 1993. See the Appendix for additional details. Further evidence, in the form of recall surveys from 2004, helps to more precisely determine the timing of diclofenac use for veterinary purposes in India. The survey responses found that diclofenac was widely available for sale in veterinary clinics by 1994 (Cuthbert et al. 2014). With these three sources of information, we classify 1994 as the first year in which diclofenac was widely used, and assign it as the year of treatment onset.

Following the large-scale use of diclofenac, reports of vulture declines began to emerge. Field observations in 1996 found only half of the three-hundred and fifty-three nesting vulture pairs recorded in 1984 in Keoladeo National Park outside Delhi. Surveys conducted in 1996 reported dead vultures around the nests, in bushes, and hanging from the trees. By 1999, there was not a single living vulture pair documented at the site (Subramanian 2011). After Dr. Vibhu Prakash, at the time a PI in the Bombay Natural History Society, communicated his findings to colleagues, they reported similar patterns they thought were simply idiosyncratic to their study sites. Population declines were so rapid that in 2000, all three species were classified as critically endangered (2011).[6]

At first, several conjectures were made regarding the potential cause. Plausible causes were considered to be an emergence of a new wildlife disease or the effect of pesticide accumulation, as well as deliberate poisoning by western countries (2011). However, it was not until 2004 when Oaks et al. (2004) used both autopsy data, and experimental exposure of vultures to diclofenac, that the causal link was established. Even small trace amounts of diclofenac in the carcasses that vultures feed on result in lethal kidney failure. As a result, the Indian government banned the veterinary use of diclofenac in 2006 (Prakash et al. 2012; D. L. Ogada et al. 2012). However, surveys conducted up to 2018 documented continued and rampant illicit use of diclofenac in livestock (Galligan et al. 2020).

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

Despite the 2006 ban on veterinary use, vulture populations are far from fully recovered. This is not surprising given that vultures have a low fecundity. A female vulture will lay at most a single egg each year. Vultures take five years until they reach sexual maturity. Assuming they find a mating pair, construct a nest for six weeks, lay a single egg, and successfully feed and ensure the survival of the offspring for four months, a new vulture gets on the path toward reproducing in about five years (Subramanian 2011; D. L. Ogada et al. 2012). The ongoing use, even if at lower capacity, of diclofenac prevents seeing large population gains as vultures can still feed on carcasses with diclofenac residue.

3 Vulture Presence, Health, & Livestock Census Data

In this section, we briefly summarize the data sets for the presence of vulture species, health outcomes, and dogs counts we use in the analysis. We provide additional details in the Appendix. In Table 1, we provide descriptive statistics for the main variables we used in the analysis.

3.1 Vulture Habitat Ranges

We obtain maps from BirdLife International (BLI) on the species distribution ranges of all bird species. In our analysis, we extract the range maps for vulture species, and perform two spatial calculations with the 1981 district boundaries: whether the district intersects with the range map, and the area of overlap between the range map and the district. We use the area of overlap to calculate the share of area for each vulture species in each district. Our approach assigns each district a suitability category for diclofenac-affected vultures by dividing the mean overlap of species ranges into terciles. This provides us with a proxy for the abundance of vultures and their prevalence across the district. This approach is more flexible, and less dependent on functional form assumptions previously used to relate environmental suitability to outcomes of interest (Alsan 2015).

Vulture species that are negatively affected by diclofenac are mostly concentrated inland. Figure 1 shows the spatial distribution of the classification into high, medium, and low suitability categories for diclofenac-affected vultures. In general, the further you move away from the center of the country, the lower the suitability is for the three affected species. The highest suitability is concentrated just south the Himalayas. In Figure C1, we summarize the relationship between the number of species present, the mean overlap of the habitats with the district area, and the classification into each tercile. The first tercile is composed of districts that have a partial overlap with one or two species, while the second and third terciles are predominately composed of districts that overlap entierly with two or three habitat ranges for the diclofenac-affected vultures.

The data in the species distribution maps provided by BLI is regarded as the most complete source of information regarding the habitat areas of bird species around the world. BLI also assess the conservation status and extinction risk as part of the Red List, produced by the International Union for Conservation of Nature.[7] BLI uses both published and unpublished sources of information to determine the boundaries of each range. Some unpublished sources of information include specific interviews with local experts, as well as confidential records.[8] The maps are known to err on the side of including areas that might not contain the species (Ramesh et al. 2017). This means that the true distribution of the species is a subset of the area in the distribution map.

3.2 Sales & Product Entry of Pharmaceuticals in India

We use data provided by the Indian Statistical Institute on the sales and the patent dates of drugs across India from 1991 to 2001. The data include information about the main active ingredient, the concentration, usage (topical, oral, or injection), as well as data quantity sold, value sold, and the year when product was launched. We focus on painkillers where

[7] The Red List is a set of species assessments that classifies species as threatened or non-threatened with respect to extinction risk, across several sub-categories.

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

diclofenac is the active ingredient, especially the products that are injected as those are more likely to be used in livestock animals. In Figure 2a, we plot the quantity sold and new product entries across India of veterinary-related diclofenac products. While we see an increasing trend from 1991 to 1996, there is a clear jump in sales in 1997 along with the entry of five new products.

3.3 Observation Records of Bird Species

There are no detailed survey data on vulture populations that allow us to compare changes in the presence of vultures at the district level. In the absence of scientifically collected data on the population levels of birds, we instead use citizen science data from eBird on the recorded observations of different bird species at the national level. We count the number of observations of either the diclofenac-affected-vultures, all vulture species, all bird species, and all bird species that have non-zero observations reported each year during 1990 to 2004.

We summarize the share of the diclofenac-affected-vultures relative to each other group. Taking the ratio between two groups allows us to account for growing trends in reporting, as long as those are not changing differentially over time for different bird species. In Figure 2b, we report a decline in the share of diclofenac-affected-vultures relative to each other group of bird species, which coincides with the timing of when reports of large vulture die-offs began to emerge in 1996.

3.4 Health Outcomes

In the paper we use mortality data at the district level from the Vital Statistics of India (VSI), reported as part of the Civil Registration System (CRS). The VSI-CRS data provide us with fine-spatial resolution data, yet are known to suffer from issues of underreporting (we compare the extent of under-reporting in a following subsection). The data include information regarding live births, deaths from all-causes, and infant deaths. Most districts have areas defined as either rural or urban, and the data are reported separately. Areas

classified as urban are not necessarily similar to a city, and might simply be a denser village. The official classification classifies an area as urban if it has a population above 5,000 people, or if more than 75% of men work in non-agricultural jobs (Burgess et al. 2017).

There is 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 2001, that data are reported as counts. We use population data from the censuses to calculate population growth rates, and use an exponential growth function to interpolate population during inter-censal years. We then calculate birth rates, death rates, infant death rates, and total death rates (that include infant deaths) using the interpolated population data. In our main results, we use the data from 1988 to 2001 as the data were calculated differently and are perhaps less comparable. In the Appendix, we provide the results for the full 1981 to 2001 period.

Using the classification into suitability terciles, we plot the mean population weighted all-cause death rates across in Figure 2c. We observe an increase in death rates in the high and medium suitability categories following the onset of diclofenac use in 1993. However, no similar change in magnitude or trend is observed in the lowest suitability category. The three habitat suitability groups overlap quite strongly in the years leading to the collapse in diclofenac-affected-vulture populations, yet diverge from each other following the onset of diclofenac use in livestock – the cause of the vulture collapse.

3.5 Stable District Boundaries

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

Hanna (2014).

3.6 Livestock Census

In addition to a population census and an industrial census, India also reports a livestock census. The data include counts of different livestock animals such as cattle, sheep etc. Surprisingly, the data in the livestock census also report a count of dogs at the district level.[9] We collected data from the states on their livestock censuses that were conducted in 1992, 1997, and 2003.[10] Our main variable of interest is the number of dogs recorded at the district level as it allows us to provide empirical evidence for two key mechanisms for an adverse health shock following the decline in vulture populations. If dog populations increased more in the high suitability areas for diclofenac-affected vultures, then that is consistent with the anecdotal evidence regarding the increase in feral dogs, animal bites, and rabies cases. In addition, it provides a proxy for an increase in mammalian scavengers such as dogs and rats.

3.7 Quantifying Under-Reporting in the CRS Data

In the main analysis, we rely on data from the Civil Registration System (CRS) as it offers data at the district level. One known limitation of the CRS data is that many vital statistics events go unrecorded, and as a result, the CRS under-reports the true magnitude of mortality. As our analysis compares changes over time using the CRS data, we are still able to recover the level differences. However, when we interpret those effects relative to a baseline level of mortality, using the mean mortality reported in the CRS data will over-estimate the relative

[9] 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. The Livestock census figures nonetheless give an indication of the trends in populations of dogs that depend, at least partly, upon food other than that directly provided by humans. The estimates are, however, likely to be underestimates of total dog populations."

[10] Ongoing collection efforts are expanding both the temporal and spatial coverage on the livestock census data.

change in mortality.

In order to obtain a nationally representative baseline of mortality, we use the Sample Registration System (SRS) data. The SRS samples less than 1% of the population, but is designed to recover a nationally representative sample (Rao and Gupta 2020). While we do find that at the national level, the CRS underestimates mortality rates by about a factor of two relative to the SRS, when controling for state and year fixed effects, both recover similar trends in mortality rates (see Figure C2). Consequently, in the interpretation of the analysis, we interpret the magnitude of the coefficients relative to the mean level from the SRS data, which reflect the national-level death rate, while also we presenting the mean mortality level from the CRS, to reflect the data in the estimation sample.

4 Empirical Strategy

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

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

Under the assumptions that the ecosystem was in equilibrium, and that diclofenac use

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

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

4.1 Differences-In-Differences Design

Exploiting the sudden and drastic change in the presence of vultures across the Indian sub-continent we examine different outcomes of interest. Specifically, health outcomes and concentrations of dog populations. For an outcome of interest, y_{dast}, in district d, rural or urban area a, state s, and time period t, we estimate the following event-study-like difference-in-differences (DD) regression specification:

$$y_{dast} = \sum_{\substack{\tau \in \{\underline{T},\ldots,\overline{T}\} \\ \tau \neq 1993}} \beta_\tau (\text{High Vulture Suitability})_d \times \mathbb{1}\{t = \tau\} + \lambda_{da} + \delta_t + \boldsymbol{X_{dast}\theta} + \varepsilon_{dast} \quad (1)$$

Where high vulture suitability is a dummy variable that equals one for districts that we classify as highly suitable for the three vulture species affected by the exposure to diclofenac. Explicitly, we define high suitability as either being in the third tercile, or the third and second terciles of the overlap between vulture ranges and districts areas (see Figures 1 and C1). We interact the suitability dummy that defines that treated group with year dummies. We omit 1993 as the baseline year as that is the year in which the patent on diclofenac expired, allowing the generic pharmaceutical industry to produce the painkiller, making it widely accessible for farmers (see section 2 for more details).

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

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

[11] In mainland India, there are only two districts that do not overlap with any of the ranges of the diclofenac-affected vulture species. See Figure C1 for more details.

able 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.[12]. We flexibly control for pooled time trends using year fixed effects in the more parsimonious specifications. For robustness, we also allow time trends to vary differently in rural and urban areas, and we include area-by-year fixed effects.

In addition to secular time trends that could explain variation in outcomes, we also control for 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 health outcomes and dog counts, then we could interpret the spurious correlation in reporting and high suitability as the effect of vulture population collapse. To account for potential variation in reporting over time, we also produce results with state level linear time trends. We do not include state-year fixed effects as we do not have sufficient variation in our suitability measure at the state level (see Figure 1).

To further test that any observed results are strictly driven by the interaction of vulture suitability and diclofenac use onset, we also include other control variables, $\boldsymbol{X_{dast}}$. We include weather variables, economic activity indicators, and land-use. Any unobserved variation is captured by the error term, ε_{dast}. We allow standard errors to be correlated across years and across urban and rural area within a district. In our baseline results, standard errors are not correlated across districts. In the Appendix, we relax the assumption on no spatial clustering.

To summarize the average treatment effects, we collapse the post-treatment coefficients using the following specification:

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

$$y_{dast} = \beta(\text{High Vulture Suitability}) \times (\text{Post Diclofenac Use}) +$$
$$\lambda_{da} + \delta_t + \boldsymbol{X_{dast}\theta} + \varepsilon_{dst} \quad (2)$$

5 Results for Health Outcomes

Following the onset of diclofenac use in 1994, and the first observed signs of large-scale decline of vultures in 1996, we find that in the high vulture suitability districts death rates from all causes increased. We estimate that, on average, death rates were higher by over three percent, and that by 2001 they were ten percent higher relative to the nationally representative mean level in 1993 reported in the SRS data.[13] Our analysis also finds considerable heterogeneity between urban and rural areas, as well as between the top of the suitability distribution and its middle terciles. We do not find strong evidence for an increase in infant mortality rates, or for an effect on birth rates (see Appendix Tables A1 and A2).

5.1 All-Cause Death Rates

Death rates increased in areas that overlap with the range of diclofenac-affected vultures after farmers started using diclofenac in 1994. In Figure 3, we report the 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. By the end of the sample, in 2001, death rates were higher by about one death per-one-thousand people, reflecting an increase of ten percent relative to the mean level in 1993 in the SRS data.

The increase in death rates did not start immediately in 1994. When comparing the third and second habitat overlap terciles to the first tercile (Figure 3a, we observe an in-

[13] Using the CRS data allows us to recover level differences, but a correct interpretation of the relative change requires using the nationally representative baseline from the SRS data. See section 3.7: Quantifying Under-Reporting in the CRS Data for additional details.

crease in death rates that started in 1996. From 1998 onward, we can reject the null of no difference in death rates between the two groups of districts. We should expect to see a dynamic response where effects intensify over time given that the decline in vulture populations took place in stages, and their functional capacity was not immediately removed from the environment in 1994. This observed heterogeneity of treatment effects over time is also consistent with diclofenac use taking time to become prevalent instead of a discontinuous jump to full utilization by farmers in 1994.

The effects on death rates are larger, and show up earlier, when comparing the third tercile of the habitat overlap to the first tercile (Figure 3b). For the districts that are most suitable for the diclofenac-affected vultures, death rates started to increase in 1995, and continued to diverge year-by-year from the least suitable districts. By 1999, we estimate that in this most suitable districts, death rates were higher by one to one-and-a-half deaths per-one-thousand persons.

Urban areas experienced the largest increase in death rates relative to rural areas. In Figure 4 we estimate the difference between the third and second to the first tercile of habitat overlap for urban and rural areas separately. Death rates in both urban and rural areas increased after 1994, but they increased more and earlier in the urban areas. This is consistent with the location of carrion landfills at the outskirts of areas classified as urban. Urban areas also have higher population densities that can result in higher infection rates following the loss of sanitation functions provided by vultures, and an increase in dog and rat populations.

On average, death rates were higher in the post-diclofenac period of 1994 to 2001. In Table 2, we summarize the magnitude of the effects across the different comparisons in Figures 3 and 4. We observe the two largest increases in death rates when comparing urban areas, third and second terciles, and rural areas, third tercile, to the first tercile (columns two and six). These estimates average together the period right after the onset of diclofenac use, while the effects on vultures were not yet observed and potentially not yet having an impact

on the ecosystem dynamics. As a result, the point estimates are lower and less precise than the results reported near the end of the sample in Figures 3 and 4.

In additional analysis, we verify that extending the sample to cover 1981 to 2001 allows us to recover similar estimates (Figure A6, and that the results are robust to including state-linear time trends (Table A3. As we use the two top habitat overlap terciles as the treated group, we also classify districts into treatment and control status if they are above or below the median level of habitat overlap with the affected species and obtain similar results to those in the main analysis (Figure A7.

6 Results for Dog Populations

Using the data on dog counts, collected as part of the livestock census, we find suggestive evidence for an increase in the number of dogs following the decline in vulture populations. These findings are consistent with the anecdotal reporting of increasing dog counts following the decline in vultures. However, we do not interpret these estimates as strong evidence for the mechanism of higher rabies disease burden resulting from the use of diclofenac and its subsequent effects on vultures. The census data are reported roughly every five years, and we compare data from 1997 and 2003 to data from 1992.[14] In Table 3, we report the estimates from the specification of Equation 2 when comparing the third and second terciles to the first tercile of the diclofenac-affected-vulture district habitat overlap.

The estimates across all sample years and weighting options are all positive, yet are imprecise. When weighting districts equally, we estimate a four to five percent increase in dog counts (Table 3, columns one and two). These estimate increase to eight and twenty-six when weighting districts by their census population level in 1991 (columns three and four). We recover similar estimates when weighting by the count of dogs in the 1992 livestock census (columns five and six).[15]

[14] We are collecting more data to establish baseline dog counts pre-1992, and to extend the coverage of districts for 2003 onward.

[15] Weighting by baseline values allows the estimates to be driven more by the districts that already had a

7 Results for 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). We use data on water quality outcomes that are most directly linked to a larger presence of carrion when disposal by scavengers declines: dissolved oxygen and fecal coliforms. The higher availability of organic matter decomposing in the water consumes oxygen, lowering the amount of dissolved oxygen. The higher availability of carrion that were not fully consumed by scavengers increases the availability of gut pathogens, such as fecal coliforms.

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

8 Conclusions

Using a panel of health outcomes at the district level, we provide evidence on the public health implications from the decline of vulture species in India. The collapse in their populations was a direct effect of the use of the painkiller diclofenac in livestock. Affected vulture species would die within weeks if they fed on a carrion with diclofenac residue. We find that all-cause death rates increased in the districts that are considered more suitable to the diclofenac-affected vultures to those less suitable before and after the large-scale onset

large dog population in 1992. Meaning, that a doubling from ten to twenty receives more weight than a doubling from one to two.

of diclofenac use. Our results inform current vulture recovery efforts in India, and global conservation policies more broadly. Further research is needed on the role that other species play in key outcomes of interest.

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

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

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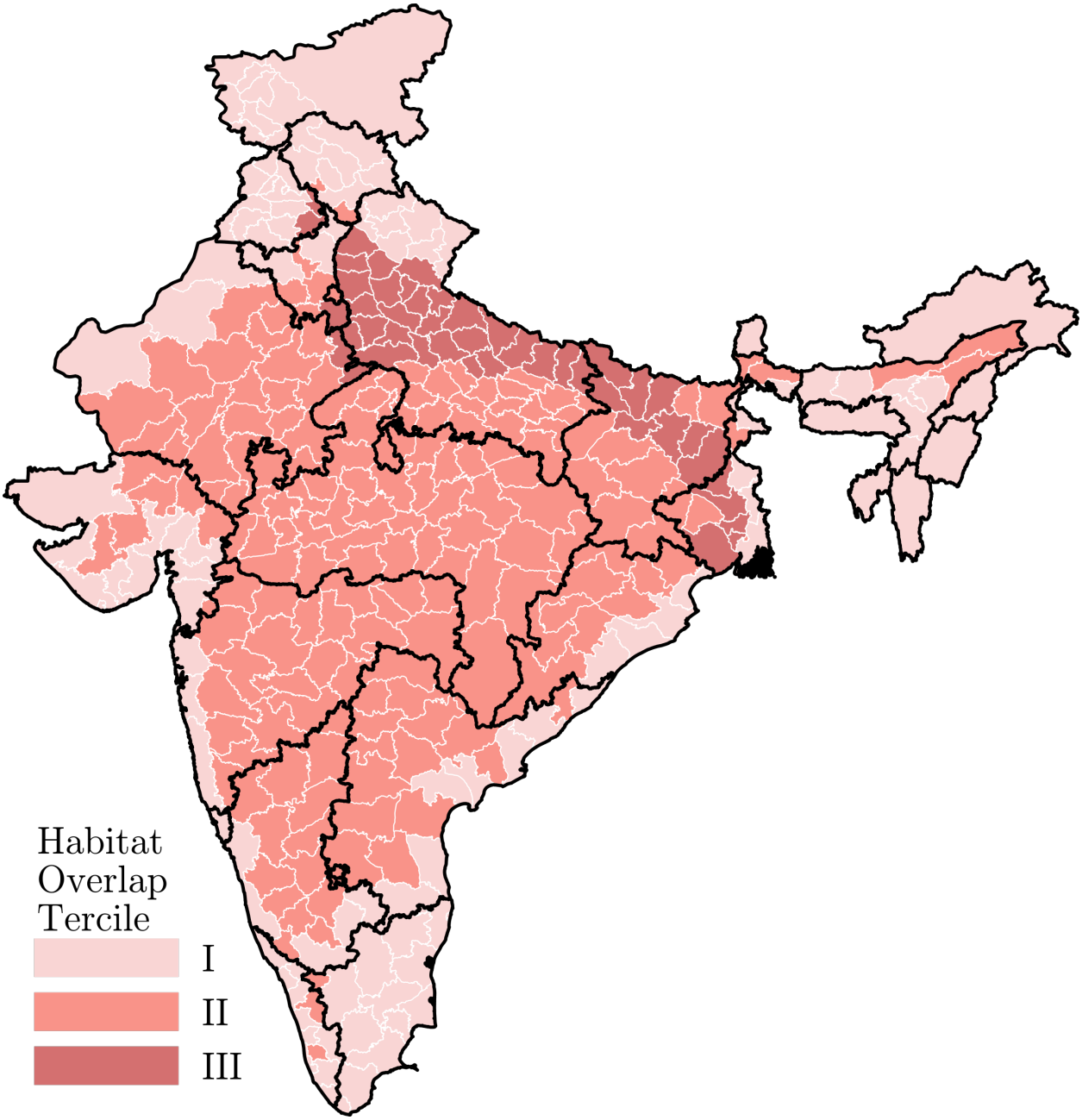
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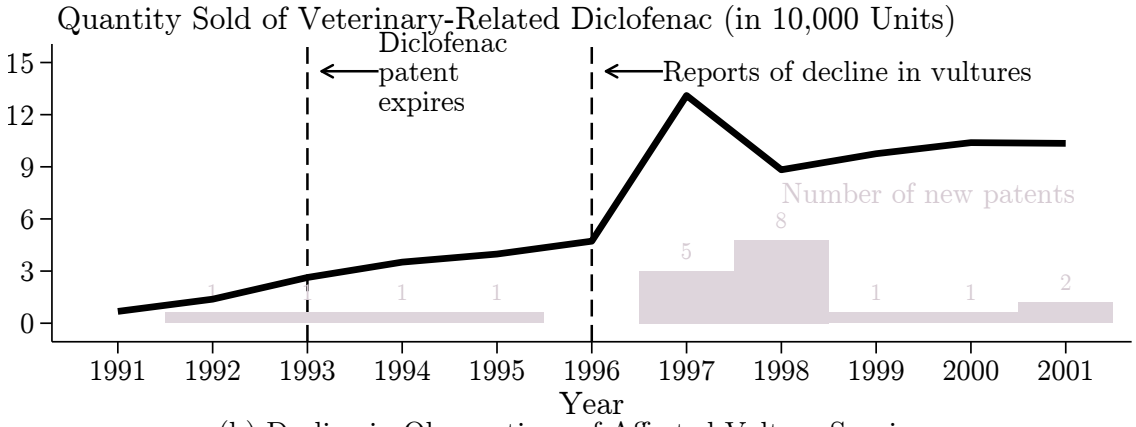
Figure 1: Spatial Distribution of Diclofenac-Affected-Vulture Ranges



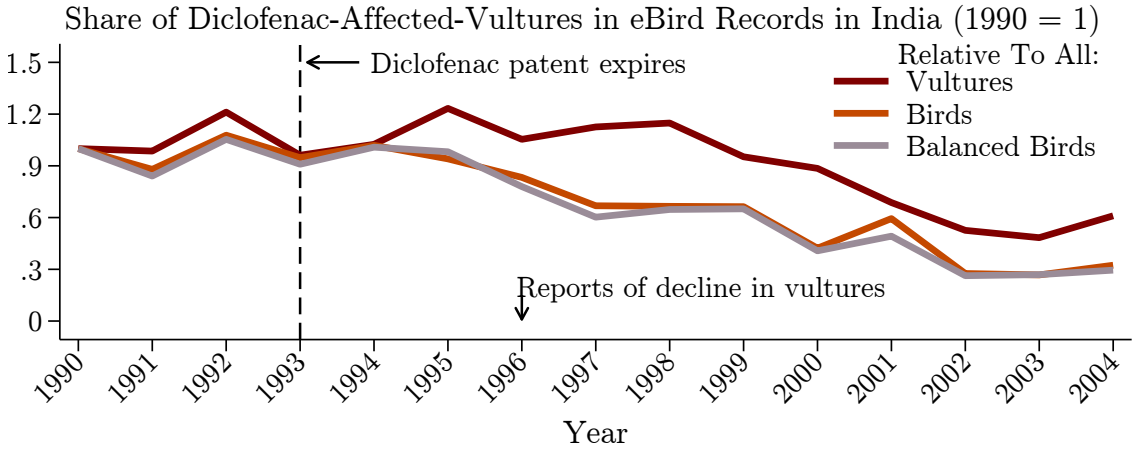
Notes: Districts in India, at their stable 1981 geographic borders, classified as high, medium, and low exposure to diclofenac-vulture-collapse (see text for more details).
Source: Range distribution maps from Birdlife International.

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

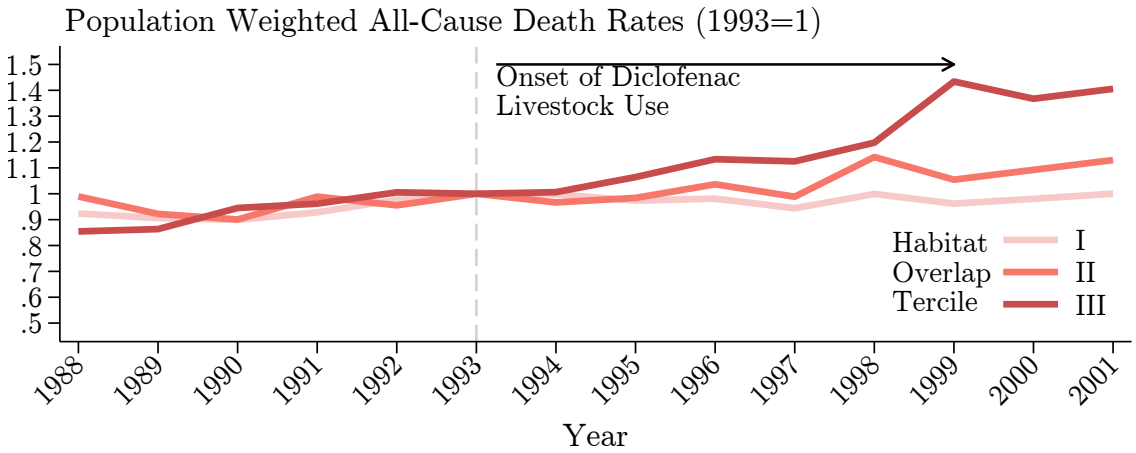
(a) Expansion in Diclofenac Use After Patent Expired



(b) Decline in Observations of Affected Vulture Species



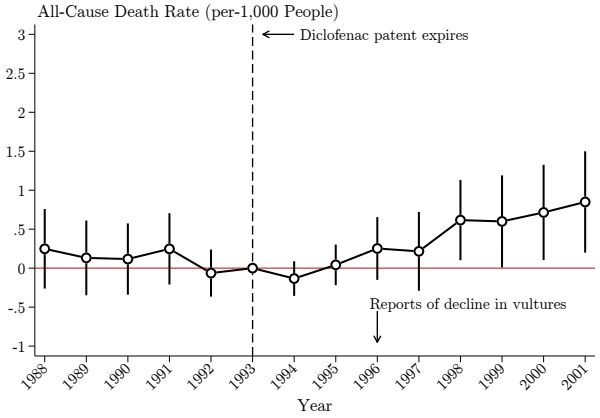
(c) All-Cause Death Rates by Vulture Habitat Suitability



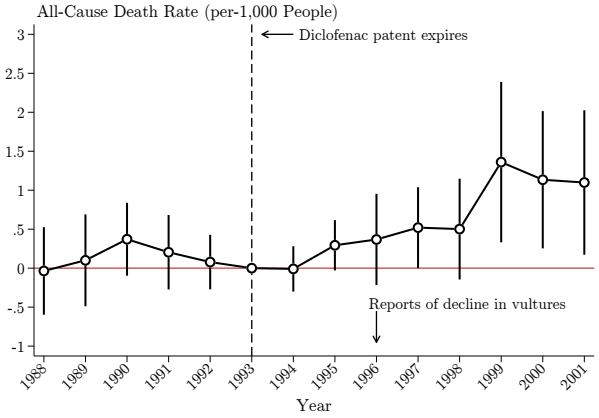
Notes: (a) Veterinary-related diclofenac sales, and the number of new product entries. (b) The share of diclofenac-affected-vultures relative to all other vultures species, all bird species, and all bird species that are consistently reported every year. (c) Mean all-cause death rates (balanced and not residualized) by tercile of habitat overlap with diclofenac-affected-vultures.
Source: Observations records of birds from eBird. Diclofenac sales and patents data from the Indian Statistical Institute. Mortality data from Vital Statistics of India.

Figure 3: All-Cause Death Rates DD Estimation Results

(a) Terciles III & II Relative to I



(b) Tercile III Relative to I



Notes: Estimation results from Equation (1). Comparing the high and medium suitability districts (terciles III and II) to the low suitability districts (tercile I), for the diclofenac-affected-vultures (see Figure 1 for the spatial distribution of vulture suitability). Observations are population-weighted. Standard errors are clustered at the district level.
Source: Data on district and habitat overlap of diclofenac-affected-vultures from Birdlife International. Mortality data from Vital Statistics of India.

Figure 4: Urban and Rural All-Cause Death Rates DD Estimation Results

(a) Urban Areas



(b) Rural Areas



Notes: Estimation results from Equation (1). Comparing the third and second terciles to the first tercile of vulture habitat overlap. Observations are weighted by population levels. Standard errors are clustered at the district level.
Source: See Figure 3.

Table 1
Summary Statistics for Main Estimation Sample

Variable	Mean	SD	Min	Max	N
Death Rate	5.32	2.44	0.21	26.91	4,998
Infant Mortality Rate	17.57	17.90	0.04	442.24	4,998
Birth Rate	22.08	15.81	0.41	417.26	4,998
Population	140.79	127.85	1.72	1197.84	4,998
Urban Area	0.48	0.50	0	1	4,998
Mean Habitat Overlap	0.60	0.14	0	0.75	4,998
Dogs (in Levels)	7.84	4.50	1.41	27.07	132
Dogs (in Log Points)	11.81	0.55	10.25	13.20	132

Notes: Summary statistics for the main estimation sample spanning 1988 to 2001. The sample for the health outcomes is balanced across all outcomes. Observations for the health outcomes are at the district-area-year level while observations for the livestock census data are at the district-census year level (1992, 1997, and 2003). Death and birth rates are per-one-thousdand people, and infant mortality rate is per-one-thousdand live births. Population are in ten-thousand people. Population data are interpolated using an exponential growth function. See main text for more details. Dog counts are in ten-thousand dogs.
Source: Health and population data from Vital Statistics of India.

Table 2
District Death Rates DD Estimates
Outcome: Death Rates, All Causes, Per-One-Thousand People
Treatment: High Vulture Suitability (HVS) & Post Diclofenac Use

Comparing Terciles	III & II To I			III To I		
	U&R	U	R	U&R	U	R
	(1)	(2)	(3)	(4)	(5)	(6)
Treat×Post	0.32	0.60	0.15	0.56	0.35	0.52
	(0.15)	(0.20)	(0.16)	(0.21)	(0.48)	(0.15)
$\overline{\text{Death Rate}}_{1993}$	4.86	5.99	4.39	5.17	6.11	4.71
R^2	0.796	0.765	0.778	0.786	0.777	0.770
N	4,998	2,380	2,618	2,478	1,176	1,302
Clusters	198	170	187	98	84	93
District FE	X	X	X	X	X	X
Year FE	X	X	X	X	X	X

Notes: Estimation results for the specification in Equation (2). Comapring the third and second tercile of diclofence affected vultures to the first tercile (columns one to three), or the third to the first tercile (columns four to six), before and after the onset of diclofenac use. Reporting results for the the urban and rural sample (U&R), or urban (U) and ruarl (R) separately. Observations are weighted by population levels. For interpretation, we include the mean of the dependent variable in 1993 (prior to the onset of diclofenac use). Standard errors are clustered at the district level.
Source: see Figure 2.

Table 3
District Livestock Census DD Estimates
Outcome: Logged Dog Counts
Treatment: High Vulture Suitability (HVS) & Post Diclofenac Use

	(1)	(2)	(3)	(4)	(5)	(6)
Treat×Post	0.05	0.04	0.08	0.26	0.10	0.23
	(0.06)	(0.23)	(0.06)	(0.15)	(0.07)	(0.15)
Initial Year	1992	1992	1992	1992	1992	1992
Terminal Year	1997	2003	1997	2003	1997	2003
$\overline{\text{Dog Counts}}_{1992}$	11.74	11.65	11.83	11.85	12.02	11.99
R^2	0.976	0.714	0.982	0.725	0.977	0.770
N	124	102	124	102	124	102
Clusters	62	34	62	34	62	34
District FE	X	X	X	X	X	X
Year FE	X	X	X	X	X	X
Weighted	E	E	P	P	B	B

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. Observations are weighted equally (E), by census population levels in 1991 (P), or by baseline dog counts in 1992 (B). Standard errors are clustered at the district level.
Source: see Figure 2.

Table 4
District Water Quality DD & DDD Estimates
Outcomes: Water Quality Parameters
Treatment: High Vulture Suitability & Post Diclofenac Use

	Log(Dissolved Oxygen)			Log(Fecal Coliforms)		
	U&R		U	U&R		U
	(1)	(2)	(3)	(4)	(5)	(6)
Treat×Post	0.01	0.05	-0.07	0.06	-0.29	0.79
	(0.03)	(0.03)	(0.04)	(0.31)	(0.40)	(0.33)
Post×Urban		0.07			-0.46	
		(0.05)			(0.34)	
Treat×Post×Urban		-0.12			1.03	
		(0.05)			(0.46)	
$\overline{\text{Dep. Var}}_{\leq 1993}$	1.91	1.91	1.89	6.87	6.87	6.86
R^2	0.630	0.632	0.655	0.777	0.779	0.841
N	3,265	3,265	1,541	2,467	2,467	1,153
Clusters	196	196	116	176	176	102
District-Type FE	X	X	X	X	X	X
Year FE	X	X	X	X	X	X

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. The 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.). Observations are population-weighted. Standard errors are clustered at the district level.
Source: Water quality data from Greenstone and Hanna (2014).

Appendix

A Additional Results

A.1 Results for Additional Vital Statistics Outcomes

A.1.1 Infant Mortality Rates

We fail to find strong evidence that infant mortality rates (IMR) increased following the collapse of vulture populations. In Figure A1 we report the same estimation as in Figure 3 but for IMR. When comparing the third to the first vulture suitability tercile, we observe an increase in IMR of more than three deaths per-one-thousand live births (see Table A1, column four). However, as seen in Figure A1b, IMR is already increasing the high suitability districts. The result of higher IMR in the high suitability districts is not robust to the inclusion of state-linear trends, which we report in Figure A4.

A.1.2 Birth Rates

Deteriorating environmental conditions that result in higher morbidity and mortality can also affect household fertility decisions. Throughout the demographic transition the pattern is often of improved conditions leading to reductions in fertility rates (Galor and Weil 2000; Chatterjee and Vogl 2018). However, as the theory on the demographic transition suggests, an increase in mortality rates can lead to a delay or even reverse the transition to lower birth rates. Empirical work in Sub-Saharan-Africa finds evidence for such a reversal due to the high mortality spikes caused by the HIV epidemic (Kalemli-Ozcan and Turan 2011; Gori et al. 2020).

If the mortality effect is disproportionately affecting infants and children, then it is plausible that birth rates will go up in response as households might choose to replace children lost to rabies or other infectious diseases. We fail to find strong evidence for an increase in IMR, and we there are no district-level mortality data by age for the time period of interest.

However, we can use the data from the vital statistics on birth rates to directly test for an effect. In Figure A2 and Table A2, we report similar pattern as that of IMR. While there are higher birth rates in the affected districts following the collapse of vultures, the effects are concentrated in the highest suitability tercile, and they are not robust to the inclusion of state-linear time trends (see Figure A5).

A.2 Including State-Linear Time 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 state-linear trends. The inclusion of state-linear trends potentially absorbs large share of the signal of interest in this case as there is little sub-state variation in habitat suitability overlap. The results for death rates, Figure A3, third and second terciles, are largely unaffected. However, for the comparison of just the third tercile to the first tercile the results are are now more indicative of a reversal of a declining trend in death rates in the high-suitability districts. When including state-linear time trends, there are no longer any apparent effects for infant mortality rates (Figure A4) or birth rates (Figure A5).

A.3 Extending the Panel to 1981 to 2001

In the main text we use the data from 1988 to 2001 for two main reasons. First, there is an abrupt shift in the reporting regime in 1988 where the vital statistics start reporting vital event counts instead of rates. As this allows to fully control the conversion to rates, we prefer to use data reported under the same regime. Second, the number of urban and rural areas in districts that are fully balanced from 1988 to 2001 are 198 and 202, while they are only 130 and 131 for the 1981 to 2001 period. When extending the panel to the full 1981 to 2001 period, and losing about thirty-five percent of the districts, we recover qualitatively similar results with lower precision (Figure A6).

A.4 Comparing Above Median Suitability

Throughout the main text, we base the definition of treatment and control groups using the classification into terciles of habitat overlap. We consider the third and second terciles to be more intensely affected by the onset of diclofenac use relative to the first tercile. The division into terciles allows us to compare high and medium suitability to low suitability districts. However, if we were to classify suitability using the median value of habitat overlap then we would end up with fewer districts in the high suitability group, and more district in the low suitability group. This would reclassify some districts with mean habitat overlap values of 0.5 to 0.6 as low suitability. Meaning, more districts in the control group will actually be exposed to a more intense treatment than in the first tercile group we use. We should expect to recover lower estimates as some of the treated units are classified as control units. We estimate the specification in Equation (1) using the classification based on the median value of mean habitat overlap. The results in Figure A7 recover an increasing dynamic response for all-cause death rates, albeit less precise.

A.5 Additional Water Quality Parameters

Here we report additional results on water quality for biological and chemical oxygen demands (BOD and COD), as well as turbidity. In general, as the demand for oxygen in the water system increases with more substances that react with it, we see dissolved oxygen levels decline (as seen in Table 4), as well as increasing levels of BOD and COD. Because BOD only captures biological uses of oxygen, it will be below the COD level which capture both organic and inorganic uses of oxygen. We should expect to see both BOD and COD levels increase with a greater availability of carrion in the environment.

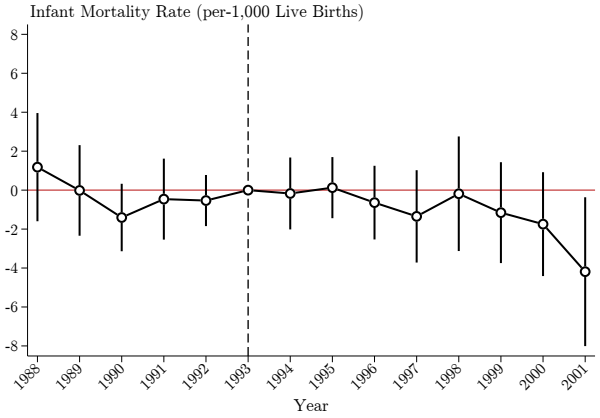
Turbidity is a measure of water quality that generally show improvement in water quality as it goes down, however, in the case of a decline in scavengers, turbiidty goes up. This is because scavengers tend to increase turbidity through the act of tearing carrion flesh. As shown in other aquatic environments, the absence of scavengers reduces turbidity (Santori

et al. 2020).

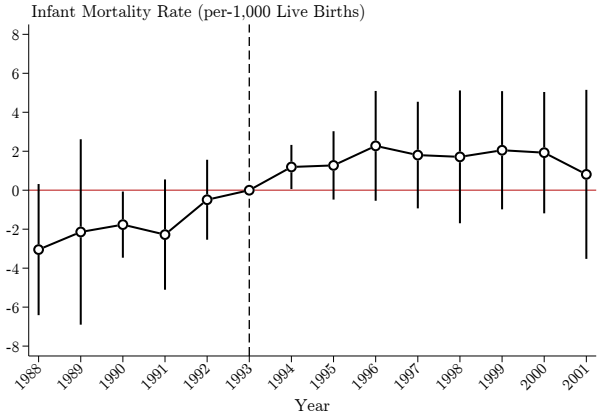
In Table A3, we report results that are consistent with the above predictions, albeit, imprecisely estimated. BOD and COD values increase in the high vulture suitability district after the onset of diclofenac use in cattle. This effect is entirely driven by the urban district (columns 2,3, 5, and 6), similar to how the decline in dissolved oxygen and increase in fecal coliforms was as well (see Table 4). Turbidity declines in water bodies monitored in urban district (columns 8 and 9), which is consistent with previous findings on decline in scavengers.

Figure A1: Infant Mortality Rates DD Estimation Results

(a) Terciles III & II Relative to I



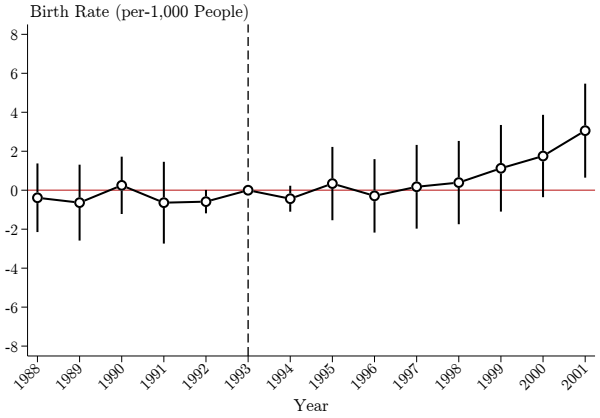
(b) Tercile III Relative to I



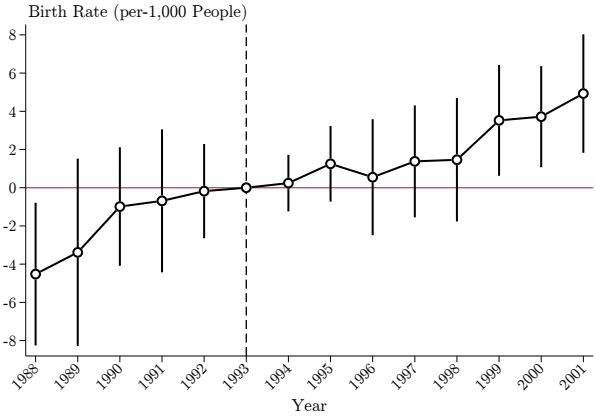
Notes: Estimation results from Equation (1). Observations are weighted by population levels. Standard errors are clustered at the district level.
Source: See Figure 2.

Figure A2: Birth Rates DD Estimation Results

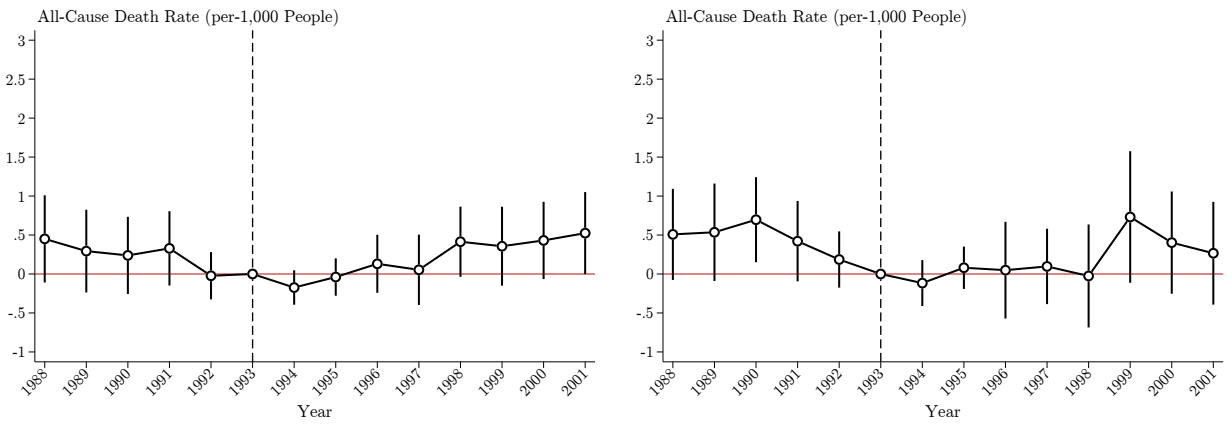
(a) Terciles III & II Relative to I



(b) Tercile III Relative to I



Notes: Estimation results from Equation (1). Observations are weighted by population levels. Standard errors are clustered at the district level.
Source: See Figure 2.



(a) Terciles III & II Relative to I

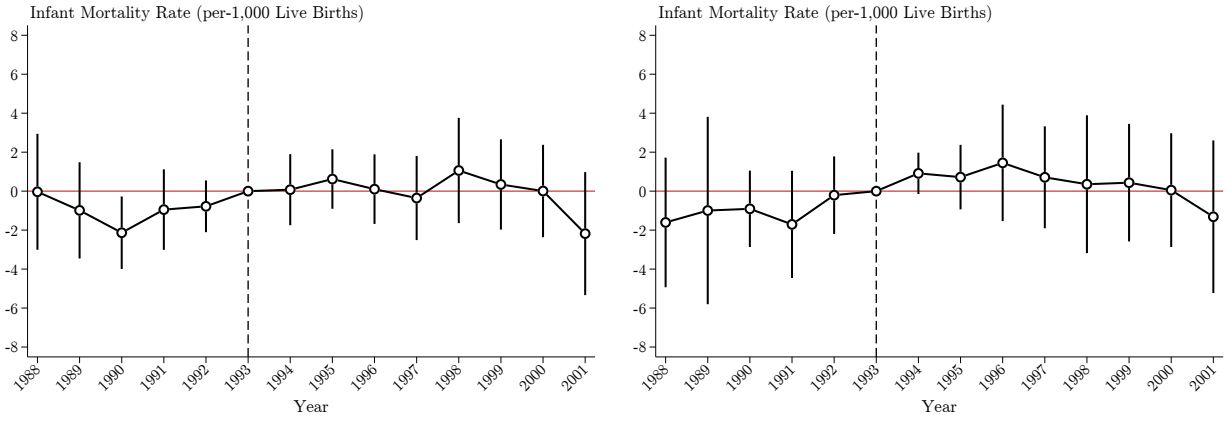
(b) Tercile III Relative to I

Figure A3: All-Cause Death Rates With State-Linear Trends

Notes: Estimation results from Equation (1). Observations are weighted by population levels. Standard errors are clustered at the district level.

Source: See Figure 2.

Figure A4: Infant Mortality Rates With State-Linear Trends

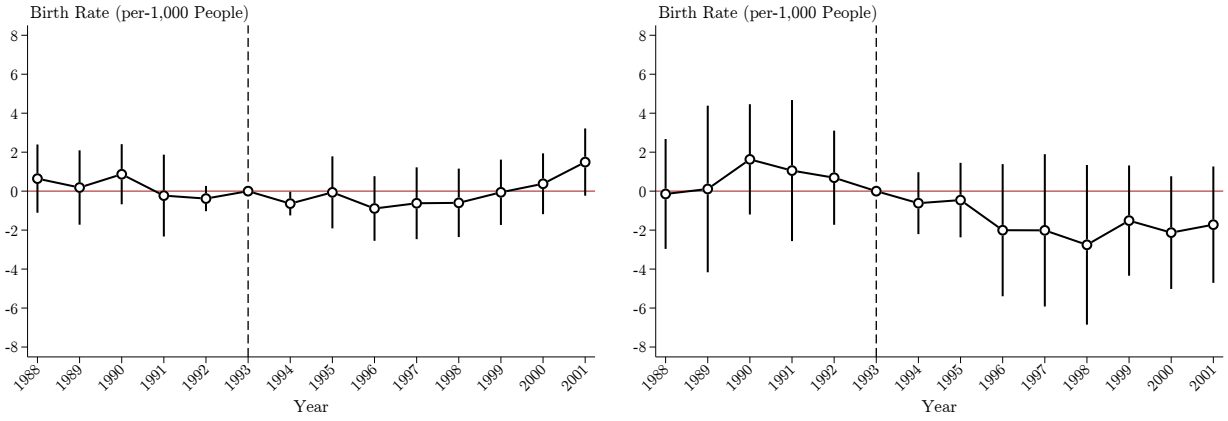


(a) Terciles III & II Relative to I

(b) Tercile III Relative to I

Notes: Estimation results from Equation (1). Observations are weighted by population levels. Standard errors are clustered at the district level.
Source: See Figure 2.

Figure A5: Birth Rates With State-Linear Trends

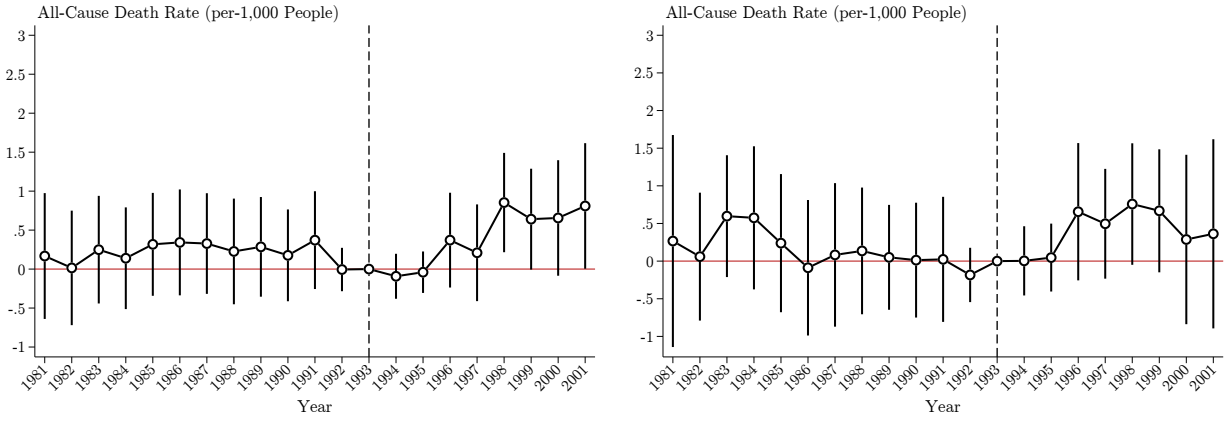


(a) Terciles III & II Relative to I

(b) Tercile III Relative to I

Notes: Estimation results from Equation (1). Observations are weighted by population levels. Standard errors are clustered at the district level.
Source: See Figure 2.

Figure A6: All-Cause Death Rates Extended Panel

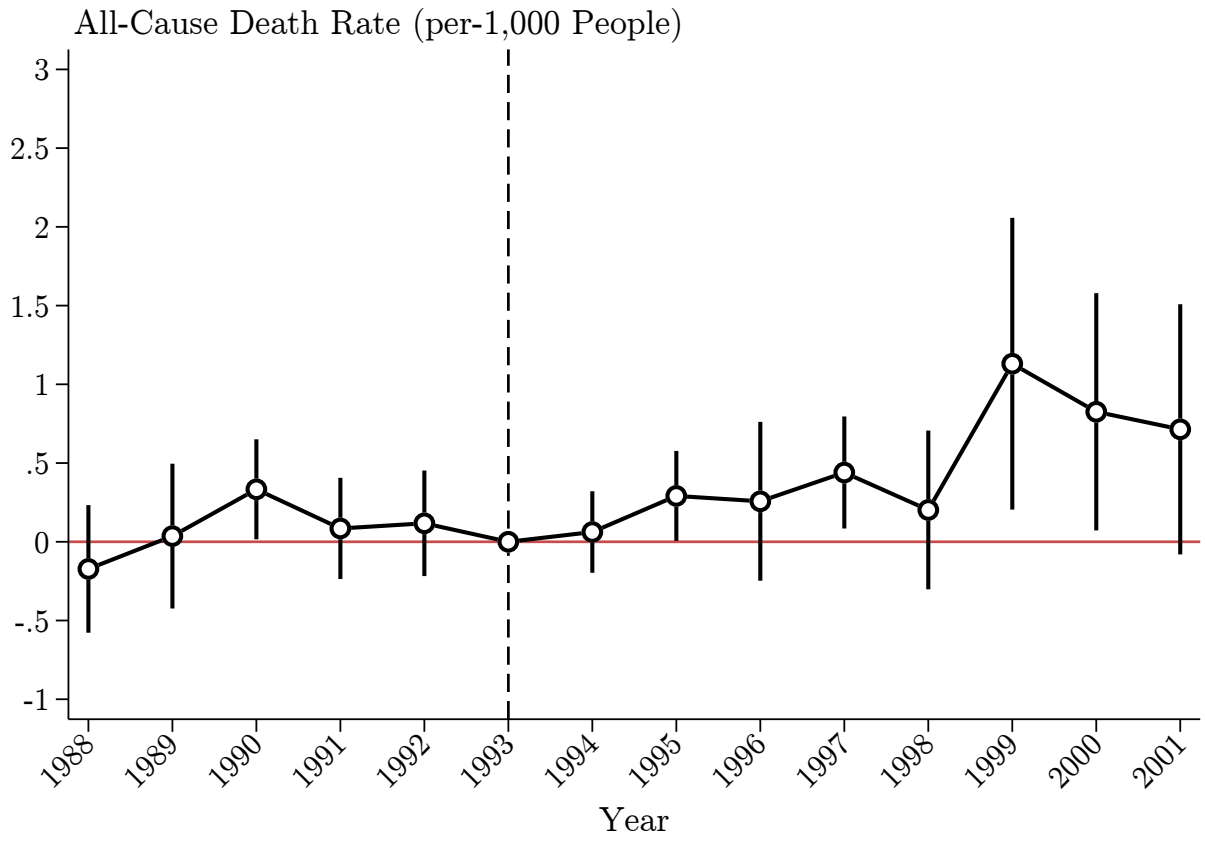


(a) Terciles III & II Relative to I

(b) Tercile III Relative to I

Notes: Estimation results from Equation (1). Observations are weighted by population levels. Standard errors are clustered at the district level.
Source: See Figure 2.

Figure A7: All-Cause Death Rates: Above Median Habitat Overlap



Notes: Estimation results from Equation (1). Comparing the district above the median of the mean habitat overlap to those below the median. Observations are weighted by population levels. Standard errors are clustered at the district level.
Source: See Figure 2.

Table A1
District Infant Death Rates DD Estimates
Outcome: Infant Death Rates, Per-One-Thousand Live Births
Treatment: High Vulture Suitability (HVS) & Post Diclofenac Use

Comparing Terciles	III & II To I			III To I		
	U&R	U	R	U&R	U	R
	(1)	(2)	(3)	(4)	(5)	(6)
Treat×Post	-0.45	-0.07	-0.51	3.49	2.19	3.92
	(1.21)	(2.11)	(1.56)	(1.38)	(3.66)	(1.70)
$\overline{\text{Infant Death Rate}}_{1993}$	18.14	21.27	16.83	16.99	21.00	15.04
R^2	0.776	0.711	0.796	0.740	0.697	0.752
N	4,998	2,380	2,618	2,478	1,176	1,302
Clusters	198	170	187	98	84	93
District FE	X	X	X	X	X	X
Year FE	X	X	X	X	X	X

Notes: Estimation results for the specification in Equation (2). Comapring the third and second tercile of diclofence affected vultures to the first tercile (columns one to three), or the third to the first tercile (columns four to six), before and after the onset of diclofenac use. Reporting results for the the urban and rural sample (U&R), or urban (U) and ruarl (R) separately. Observations are weighted by population levels. For interpretation, we include the mean of the dependent variable in 1993 (prior to the onset of diclofenac use). Standard errors are clustered at the district level.
Source: see Figure 2.

Table A2
District Birth Rates DD Estimates
Outcome: Birth Rates, Per-One-Thousand People
Treatment: High Vulture Suitability (HVS) & Post Diclofenac Use

Comparing Terciles	III & II To I			III To I		
	U&R	U	R	U&R	U	R
	(1)	(2)	(3)	(4)	(5)	(6)
Treat×Post	1.20	2.83	0.44	3.79	2.38	3.89
	(0.82)	(1.02)	(1.01)	(1.45)	(1.81)	(1.49)
$\overline{\text{Birth Rate}}_{1993}$	16.79	26.21	12.88	17.49	26.18	13.25
R^2	0.756	0.612	0.698	0.776	0.736	0.600
N	4,998	2,380	2,618	2,478	1,176	1,302
Clusters	198	170	187	98	84	93
District FE	X	X	X	X	X	X
Year FE	X	X	X	X	X	X

Notes: Estimation results for the specification in Equation (2). Comapring the third and second tercile of diclofence affected vultures to the first tercile (columns one to three), or the third to the first tercile (columns four to six), before and after the onset of diclofenac use. Reporting results for the the urban and rural sample (U&R), or urban (U) and ruarl (R) separately. Observations are weighted by population levels. For interpretation, we include the mean of the dependent variable in 1993 (prior to the onset of diclofenac use). Standard errors are clustered at the district level.
Source: see Figure 2.

Table A3
District Water Quality DD & DDD Estimates
Outcomes: Water Quality Parameters
Treatment: High Vulture Suitability & Post Diclofenac Use

	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)
Treat×Post	0.63	0.22	1.46	2.72	-0.96	9.57	-0.03	2.39	-5.26
	(0.47)	(0.48)	(0.93)	(2.86)	(1.74)	(6.46)	(3.68)	(3.83)	(6.85)
Post×Urban		-0.68			-6.69			4.07	
		(0.62)			(5.72)			(5.58)	
Treat×Post×Urban		1.17			10.25			-7.19	
		(0.96)			(6.51)			(6.95)	
$\overline{\text{Dep. Var}}_{\leq 1993}$	3.99	3.99	5.00	25.20	25.20	28.41	36.59	36.59	40.59
R^2	0.790	0.790	0.821	0.771	0.773	0.793	0.851	0.851	0.858
N	3,236	3,236	1,525	3,262	3,262	1,540	2,890	2,890	1,323
Clusters	195	195	115	195	195	115	188	188	109
District-Type FE	X	X	X	X	X	X	X	X	X
Year FE	X	X	X	X	X	X	X	X	X

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. The 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.). Observations are population-weighted. Standard errors are clustered at the district level.
Source: Water quality data from Greenstone and Hanna (2014).

B Diclofenac Use Onset

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

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

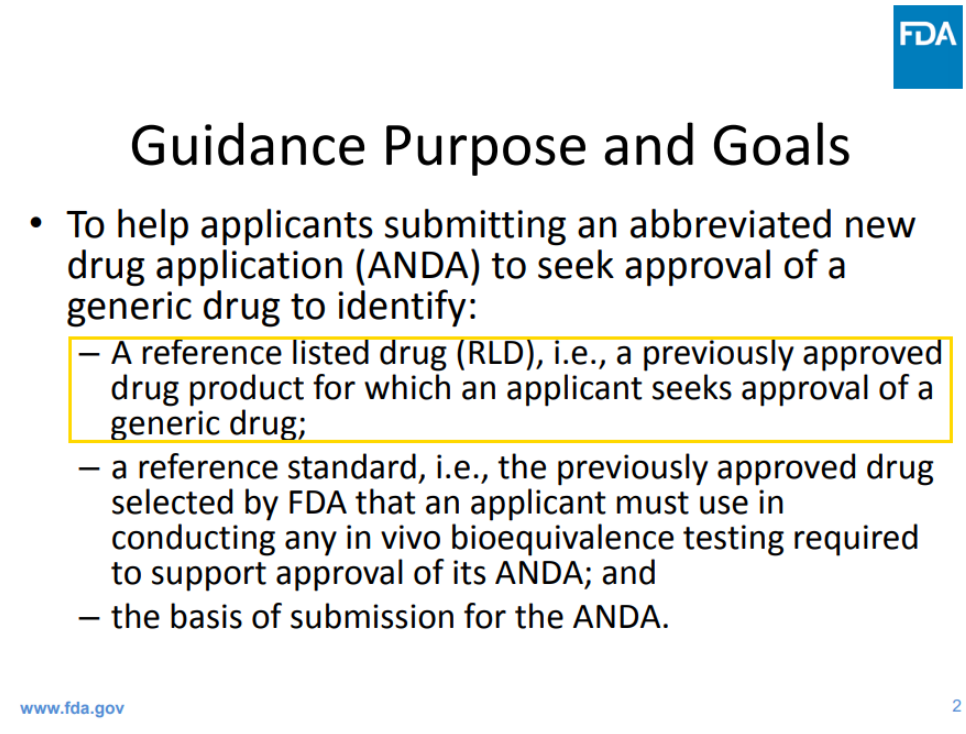
Figure B1: FDA Documents Regarding Diclofenac & Generic Drug Approval

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

(a) Change to Novartis' Diclofenac Patent in 1993

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

(b) Change Code CRLD



(c) Documentation Regarding RLD Changes

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

C Data

C.1 BirdLife International Species Distribution Maps

We requested access to the geodatabase with all the digitized maps for all bird species maintained by BirdLife International (BLI). Access is provided for non-commercial uses.[16] The data include information about whether the species are extant or extinct, along with discrete categories regarding the likelihood of the two. The data also include information on whether the species is native or not, and whether their presence is yearly, during the breeding season, or other form of seasonality.[17]

C.2 Examining the Reporting Accuracy of the CRS Data

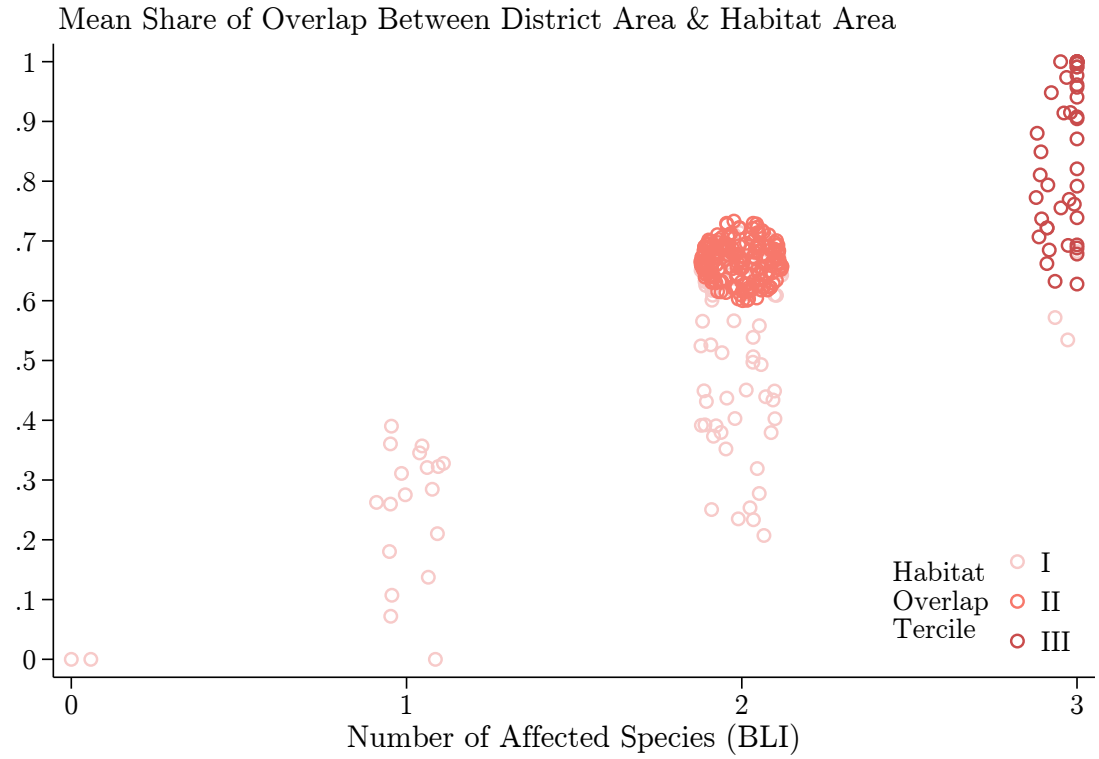
We compare the CRS data to the SRS data in order to evaluate if underreporting of mortality in the CRS data is introducing bias in the trends in addition to underestimating the magnitude. The data in the SRS are reported at the state level. To compare the CRS and SRS, we take a population weighted mean of the district- or state-level data, respectively, to obtain a national-level estimate for the all-cause death rate. We plot the levels of all-cause death rates, by source of data, by year, in Figure C2. There is a clear difference in levels (dashed lines) between the all-cause death rate in the CRS relative to the SRS data. The SRS death rate is nearly double that the CRS reported death rate. However, when residualizing the death rates on a set of unit and time fixed effects (reported in the solid lines), the two death rates follow similar trends.[18]

We interpret the agreement between the residualized levels in Figure C2 as evidence that conditional on fixed effects, the CRS data manage to capture similar trends to those in the SRS data. In addition, the results from this comparison also highlight that the correct

[16] Application can be filled out at: http://datazone.birdlife.org/species/requestdis
[17] BLI provides a summary of these categories here: http://datazone.birdlife.org/species/spcdistPOS
[18] Specifically, we include district-by-area or state-by-area, for urban and rural areas, fixed effects, as well as year fixed effects.

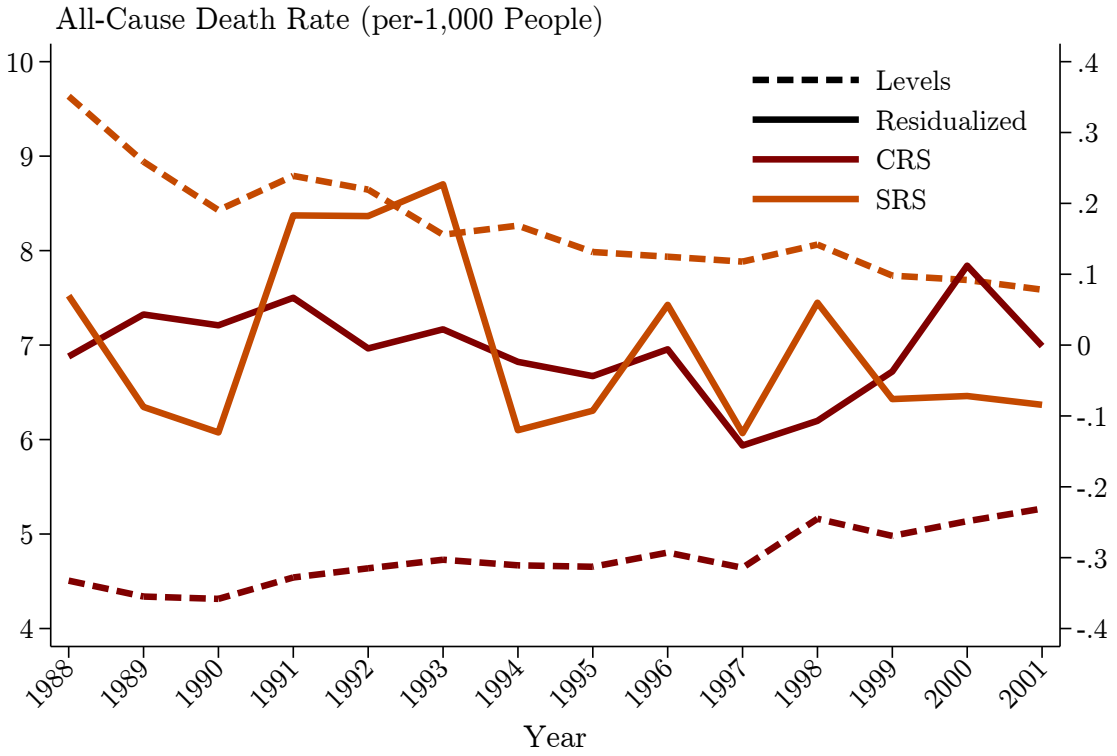
baseline level that we should use when comparing the relative change in mortality is nearly twice as large, reducing the relative size of the effect when using the CRS mean level by half. Throughout the analysis, we present the mean mortality level from the CRS, to reflect the data in the estimation sample, but interpret the magnitude of the coefficients relative to the mean level from the SRS data, which reflect the national-level death rate.

Figure C1: Classification of Districts Into Habitat Suitability Terciles



Notes: Each hollow circle is a district showing the number of vulture species that are adversely affected by diclofenac, and that range intersects with the district, along with the mean level of overlap between the Birdlife International habitat range map, and the district. Districts are classified into three habitat overlap terciles based on their mean share of overlap. Circles are jittered to allow for easier visual inspection.
Source: Data on vulture habitat areas from Birdlife International range maps.

Figure C2: 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) by area (urban or rural), as well as year fixed effects.
Source: Data from Vital Statistics of India (CRS), and the SRS database.