Bigger than Brexit?

Estimating the impact of wildlife on resource use*

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Abstract

Natural capital, including wildlife, is widely understood to enter the production function. Ex-ante, agents may not know its effect on productivity, leading risk-averse operators to neutralize threats. In wildlife management, such action may reduce welfare if it degrades ecosystem functioning, reduces biodiversity, or eliminates beneficial services provided by culled animals. But there remains little empirical evidence on the effect of so-called "nuisance" species in the absence of control operations. The beaver, an ecosystem engineer that produces a host of well-documented benefits, can hinder agricultural productivity by inundating fields, grazing crops, felling trees, and collapsing flood banks. A recent unauthorized reemergence of beavers in Scotland, where they had been extinct for centuries, has triggered a backlash among agricultural producers. Using a series of regional beaver surveys and high-resolution land use data, I exploit the rapid recolonization to test whether beavers impede agricultural operation. Contrary to conventional wisdom, preliminary results indicate that, on the extensive margin, beaver entry increases land use for agriculture 4.6 p.p. (11% relative to baseline). This effect is driven by changes in directly exposed landscape patches with high agricultural soil suitability. To validate that beavers alter their physical environment, I employ a network of in-situ hydrometry monitoring stations to measure changes in river level and flow following beaver arrival but find inconclusive evidence. Ongoing analysis aims to clarify the mechanism of the reduced-form effect.

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> [A farmer] said as far as he was concerned, the impact of beavers on his business was bigger than Brexit, and I'll now show you how this has come to fruition.
>
> *Martin Kennedy, President of National Farmers Union, Scotland* (Kennedy 2023)

A large literature has described the role natural capital plays in the economic production function. Climatic conditions, wildlife, and pests can alter productivity. But *ex-ante*, operators may not know the direction and magnitude of a given natural capital input's effect, leading risk-averse agents to neutralize potential threats. Such actions may be welfare reducing if control operations degrade ecosystem functioning, reduce biodiversity, or eliminate an ecosystem service's positive externalities. A salient example is the conflict between wildlife and agriculture. Farming operations have historically relied on the producer's or state's ability to control habitats, cull livestock predators and crop grazers, repel potential vectors of disease transmission, and eliminate crop-eating pests with mass insecticide use. In the past two decades, the US Department of Agriculture's Wildlife Services has killed 56 million wild animals, including some otherwise protected by the Endangered Species Act, to protect livestock capital (Torrella 2024). Pesticides used to protect crop yields have harmed human and animal health (Larsen et al. 2017), leading in part to the formation of the modern American environmental movement (Woodwell 1984). The Chinese "Four Pests Campaign" encouraged the mass killing of sparrows, believed to feed on grain reservoirs, and inadvertently eliminated sparrows' valuable pest-control services, contributing to the subsequent mass famine (Frank et al. 2024).

Balancing risk-reduction for economic production on one hand and ecosystem protection on the other is complicated by climate change, habitat destruction, and biodiversity loss (Cardinale et al. 2012). Indeed, a poor understanding of keystone species can distort wildlife management policy; recent evidence has challenged long-held beliefs about so-called "nuisance" species (e.g., Raynor et al. 2021). But scant empirical evidence details how economic production adapts to changes in the composition of natural capital.

The beaver (*Castor canadensis* in North America[1] and *Castor fiber* in Europe and Western Asia[2]) occupies the dual roles of agricultural menace and ecosystem engineer. Since the advent of modern agriculture, farm operators have killed beavers and destroyed their colonies to avoid the flooding, crop grazing, and timber felling. After being hunted to near-extinction until the 19th Century, the beaver has been reintroduced in much of North America and Europe. Beavers produce a plethora of positive externalities, from wetland preservation (Hood and Bayley 2008), temperature regulation (Dittbrenner et al. 2022), and carbon storage (Wohl 2013, Johnston 2014) to wildfire resistance (Fairfax and Whittle 2020) and species richness (Wright et al. 2002).

A recent illegal reemergence of beavers in Scotland, where hunting had driven them extinct centuries earlier, illustrates the challenge of poorly understood natural capital inputs. Consistent with a widespread history of farmer-led opposition to beavers, Scottish agricultural groups have opposed beaver incursion into the fertile region around the River Tay, with the president of the National Farmers Union, Scotland (NFUS) warning that beavers pose a greater threat than Brexit to his constituents (Castle 2021). In nearby Southern England, which has seen several controlled releases, residents have posted public banners vowing to oppose beaver invasion (ITookSomePhotos 2022). In Bavaria, where beavers reemerged in the 20th Century, farming organizations campaigned for their complete eradication (Campbell-Palmer et al. 2015). But anecdotal evidence suggests heterogeneous effects. While some Scottish farmers cite enormous monetary damages incurred by beaver activity (Hamilton and Moran 2015), others report no adverse effects from nearby colonies (Campbell, R.D. et al. 2012).

Scottish beaver establishment provides a ripe setting for causal inference. Exogenous to agricultural policy, climate change,[3] or wildlife management regime shifts, the rapid spread facilitates the estimation of beavers' impact on agricultural land use. To identify beaver movement over time, I employ a set of regional surveys, conducted between 2012 and 2020. To test whether beaver presence affects cropping behavior, I match high-resolution land use data to beaver arrival. To verify that beavers do indeed alter their physical environment, I measure changes in river level and flow rate after beaver entry.

[1] Kuhl, 1820

[2] Linnaeus, 1758

[3] Not all beaver expansion has been uncorrelated with the identification-confounding effects of climate change. In recent decades, the warming Arctic tundra has proved fertile habitat for beavers (Tape et al. 2022), which are further altering the environment by producing methane (Clark et al. 2023).

In my preferred specification, beaver arrival increases the share of landscape patches[4] devoted to agriculture by 4.6 p.p., an 11% change relative to baseline agricultural land share. This effect is stable across treatment cohorts but is driven by landscape patches directly adjacent to watercourses and those with soil suitable to arable cropping. River level and flow rate, which beavers alter (Swinnen et al. 2019), lower noisily in response to beaver introduction. In preliminary analysis, it remains unclear whether the beaver-caused increase in share of land devoted to agriculture reflects true changes in cropping, or either a) the greening effect of beaver colonization, b) reallocation of cropping within farm properties (where the formerly cropped land remains "switched on" in the satellite data), or c) evidence of lowered marginal productivity of land. Further study, using satellite data to measure water saturation and flooding directly at a finer scale, as well as agricultural census data on farm productivity and income, may clarify underlying mechanism.

This paper contributes an understanding of how natural capital enters the agricultural production function. A large literature details the impact of weather and climate on agricultural yield and economic growth (Mendelsohn et al. 1994, Schlenker et al. 2006, Schlenker and Roberts 2009, Hsiang and Jina 2014, Taylor and Schlenker 2021), while a small but growing field of studies treat wildlife as a natural capital input (Frank 2024, Rucker et al. 2019, Champetier et al. 2015, Kawasaki 2023, Devkota et al. 2024). This study also adds to findings on the valuation and accounting of natural capital more broadly (Lewis et al. 2024, Raynor et al. 2021, Fenichel et al. 2016, Fenichel and Abbott 2014, Kareiva 2011). Finally, it contributes evidence of the environmental effects of beaver habitation to a mature literature in ecology, whose findings I discuss further in section 1.2. To my knowledge, I am the first to study, in a quasi-experimental setting, the impact of the beaver on the extensive margin of resource use.

The paper is structured as follows. Section 1 describes the Scottish agricultural industry, beaver ecology, and the recent beaver recolonization of Scotland. Section 2 reviews the data. Section 3 describes my empirical approach. Section 4 presents results on reduced-form beaver impacts on agriculture, as well as suggestive evidence on mechanisms. Section 5 concludes and points toward future research.

[4] I refer to these units synonymously as landscape grid cells.

1 Background

Below, I describe the Scottish agriculture industry; beaver ecology, territorial expansion, and behavior, including potential threats to agricultural productivity; and the recent reemergence of beavers in Scotland.

1.1 Agriculture in Scotland

Scottish agriculture, established in a cool and marshy environment, has relied on an array of floodbanks constructed along rivers to protect cropland from inundation (Goldfarb 2018). In 2023, 69% (5.33 million hectares) of Scotland's land area was devoted to agricultural use (Cabinet Secretary for Rural Affairs 2023), with much of the arable farming concentrated along the eastern lowlands. The industry employs a 66,000-person workforce (1.2% of the country's population), contributing approximately 2.5 billion EUR of Standard Output.[5] 13% of the agricultural workforce is concentrated in Tayside, which relies heavily on casual and seasonal employees (*Economic Report on Scottish Agriculture* 2016). In 2016, Scotland's farm income was estimated at 667 million GBP, roughly 18% of the United Kingdom's total farm income (*Economic Report on Scottish Agriculture* 2016). The majority of farms report a farm business income below 30,000 GBP, with 22% operating at a loss (*Economic Report on Scottish Agriculture* 2016). Most arable farming in Scotland is dedicated to above-ground cereals, including barley, oats, and rye, and to a lesser extent, crops such as oilseed rape and beans (Cabinet Secretary for Rural Affairs 2023). In contrast to the rugged northwest highlands, where holdings are largely under five hectares, in the arable lowlands, where beavers reemerged, farm size is distributed evenly from less than one hectare to over 200 hectares (*Economic Report on Scottish Agriculture* 2016). This distribution informs my construction of 1km^2 (100-hectare) landscape patches as the study's unit of analysis.

1.2 Beavers

Beavers, including the North American *Castor canadensis* and the Eurasian *Castor fiber*, are large herbivorous rodents. Requiring water, they tend to settle on streams, rivers, lakes, or ponds, preferring areas with little to no gradient (Müller-Schwarze 2011), narrow watercourses (Dittbrenner

[5] A standard EU metric for measuring the economic weight of agricultural activity, standard output equals the average value of output per hectare of farmland or per head of livestock.

et al. 2018), slow-moving water, and nearby vegetation (Swinnen et al. 2019). Constructed from timber, branches, mud, and leaves, beaver dams may stretch up to hundreds of meters and form ponds, often with several to a single colony (Müller-Schwarze 2011). The family unit resides in a lodge, either burrowed into the river bank or built as a free-standing structure in the impounded water. The beaver often dredges a network of canals around the colony to facilitate movement between its lodge and feeding areas (2011). The typical litter has 3 or 4 kits, which spend around two years with their parents before departing to establish homes, usually within 5 kilometers of the parents (Hartman 1997, Müller-Schwarze 2011). Dispersal typically occurs along a permanent watercourse, though temporary waterways formed during spring snow melt can facilitate passage across otherwise inaccessible land.

As a keystone species and ecosystem engineer, the beaver reforms its habitat. A large literature documents a host of positive externalities from beaver colonization. Species richness and biodiversity rise (Hossack et al. 2015, Wright et al. 2002, Leidholt-Bruner et al. 1992, Bouwes et al. 2016, Fedyń et al. 2023, Kemp et al. 2012, Stringer and Gaywood 2016, Law et al. 2016), carbon storage and water filtration mitigate climate extremes and pollution (Hood and Bayley 2008, Dewey et al. 2022, Johnston 2014, Fairfax and Small 2018, Fairfax and Whittle 2020, Lazar et al. 2015, Wohl 2013), and local economies see increased tourism revenue (e.g., Campbell et al. 2007 and Auster et al. 2020).

But the beaver's consumption of natural resources, for construction materials and food, may conflict with human land use. Impounded water floods roads during periods of high precipitation and snow melt, particularly when beavers plug highway culverts (Jensen et al. 2001). Beavers may target vegetation on residential property, with many homeowners viewing beavers as a nuisance species in need of control (Jonker et al. 2006). The beaver's largest potential economic impact is on agriculture (Hamilton and Moran 2015, *Beavers in Scotland* 2017, Mikulka et al. 2020, Janiszewski and Hermanowska 2019, Campbell-Palmer et al. 2015). Burrowing into river banks may collapse flood defenses, impounding moving water may flood fields, beavers may graze on crops, and may fell orchard trees for use as construction materials.

1.2.1 Scottish Reemergence

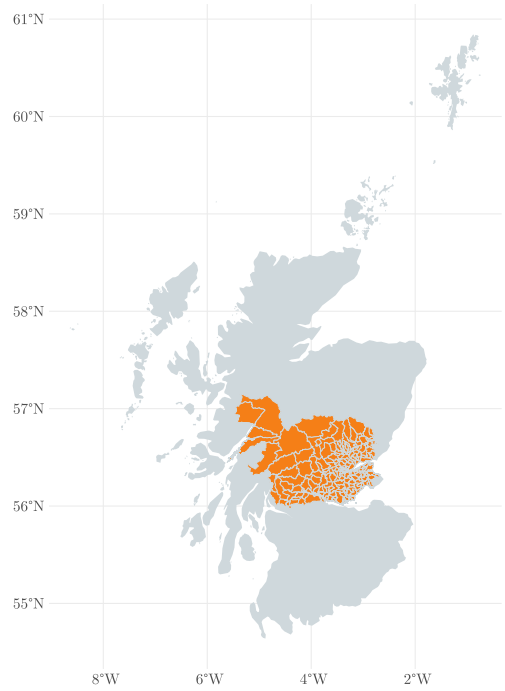
In 1996, the Scottish public conservancy began studying the feasibility of reintroducing the Eurasian beaver (Kitchener and Conroy 1997). Though present in the British Isles since at least 10,000 years ago, the beaver went locally extinct sometime between the 12th and 16th Centuries, hunted for its pelt, meat, and prized *castoreum* oil (1997). Similar local extinctions and near-extinctions occurred throughout mainland Europe and North America in the 18th and 19th Centuries. Over the past century, beavers have repopulated, through both controlled release and natural expansion, much of their former range (Dzięciołowski and Gozdziewski 1999, Janiszewski and Hanzal 2021, Schwab and Schmidbauer 2003, Hartman 1995, Dijkstra 1999). Despite the beaver's least concern status, the EU Habitats Directive now lists *Castor fiber* as an Annex IV(a) species[6].

Around 2000, wild beavers emerged in Scotland, either illegally released or escaped from a nearby enclosure (Goldfarb 2018, Campbell-Palmer et al. 2015, Gaywood 2018). This population spread around the agriculturally productive lowland River Tay region (Fig. 1). In 2012, the national conservancy conducted its first regional survey of the Tayside beavers, followed by two others in 2017 and 2020, respectively. In 2016, despite opposition from agricultural groups (Castle 2021, Kennedy 2023, Werth, Christopher 2017), the Scottish government, citing the EU Habitats Directive, allowed the Tayside population to continue expanding its range naturally (*Beavers in Scotland* 2017). In 2019, beavers became a protected species in Scotland (*Beavers given protected status* 2019). While official policy encourages non-lethal mitigation measures, hundreds of beavers have been culled since obtaining protected status, with 202 individuals killed between 2019 and 2021, not including unauthorized killing undertaken by private landowners (Williams 2021).

The fear of conflict between beavers and agricultural operations has historical precedent. During the expansion of beaver populations in Bavaria in the 20th Century, the landscape of intensive agriculture, supported by a network of engineered waterways–similar to the eastern Scottish lowlands–saw large-scale disruption (Campbell-Palmer et al. 2015). By the 1990s, several farmers organizations were advocating for beaver extirpation. In Scotland, official estimates of beaver-caused damage are scarce. To my knowledge, Hamilton and Moran (2015) is the only study of

[6] Article 12(1) of the Habitats Directive prohibits the killing and capture of Annex IV(a) species, as well as disturbance of their habitat (*Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora* 2013) Annex IV(a) protection does not apply to the populations of Finland, Sweden, Latvia, Lithuania, Estonia, and Poland, which are listed under the softer Annex V, allowing for management action to avoid over-exploitation.

Figure 1: Study region



Notes: Study region (orange) within Scotland (gray), capturing a buffer area around beaver expansion in the Tay, Earn, South Esk, Lunan, and Perth Coastal catchments. Agricultural parish boundaries are drawn in gray.

the Scottish beaver's economic impact. In contrast to my ex-post analysis of realized land use changes, Hamilton and Moran (2015) project potential impacts by scaling up landowner-reported costs, estimating damaging totaling 179,900 GBP in Tayside, almost all (173,500 GBP) incurred in the agriculture-intensive eastern lowlands. The majority of reported costs come from labor needed to process and replace of felled trees and repair flood banks. Compared to bank, crop, and tree disruption, survey respondents rarely cited field flooding as the primary cause of damage.

2 Data

To estimate the impact of beaver reintroduction on agricultural land use, I obtain data on (1) the Tayside beaver expansion from three comprehensive regional surveys conducted over a decade, (2) high-resolution satellite-derived land use classifications, (3) soil data on agricultural suitability, (4) river levels and flow rates from a network of in-situ monitoring stations, and (5) weather patterns from a standard reanalysis product. To measure the impacts of beaver entry on agriculture, I

employ the land use data, exploiting its spatiotemporal variation, spanning pre- and post-beaver entry periods, to test for land use change. To detect physical environment changes following beaver entry, I use the hydrometry data. In robustness checks, I use high-resolution data on soil type to run models on subsamples. In all analyses, the unit of observation is a $1km^2$ landscape patch in a grid constructed by tessellating the study region shown in Fig. 1. Subsamples of river-adjacent cells use an Ordnance Survey watercourse layer (*OS Open Rivers* 2023).

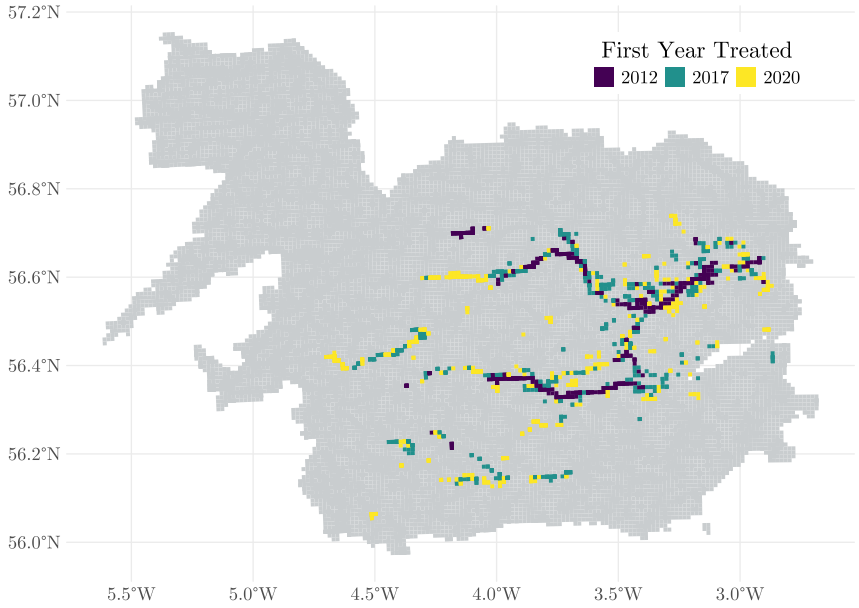
2.1 Beaver Expansion

Beavers reemerged in Tayside around 2000, with anecdotal reports beginning in the middle of the decade. In 2006, NatureScot, Scotland's national conservancy, acknowledged their presence. In 2012, NatureScot conducted the first comprehensive survey of beaver habitation (Campbell, R.D. et al. 2012). Two follow-up surveys were conducted in 2017-18 and 2020-21, respectively, with each resurveying the extent of its predecessor as well as extensions along suitable habitat corridors. The surveys were carried out by canoe and on foot, with the 2012 survey covering 690km of contiguous watercourse and 450km of non-contiguous river bank around the rivers Tay, Tummel, Isla, Almond, and Earn.[7] The 2020-21 survey covered 1,760km of contiguous watercourse and 1,238 non-contiguous spot checks along river banks (Campbell-Palmer, R. et al. 2021). Of the 1,522 field signs recorded in 2012, most were cutting, with only six direct beaver sightings. 98% of field signs were found within 10m of a watercourse. I obtain beaver survey records via the National Biodiversity Network Trust's NBN Atlas repository, which contains a limited number of the surveyor-recorded variables. In data aggregation, I focus solely on the extensive margin of beaver expansion for several reasons. First, privacy constraints did not allow for collection of the full variable set needed to estimate beaver population.[8] Second, a change in GPS data collection methods in the 2017-18 survey inflated the number of observations linked to any given field sign, a discrepancy for which the full set of variables would be needed to adjust (Campbell-Palmer, R. et al. 2018). Finally, because much of the survey work was conducted during summer months, vegetation cover obscured many beaver signs, making precise quantification unreliable.

[7] An additional 310 km of river bank had been surveyed over several preceding years, informing the initial targets of the 2012 survey.

[8] These include activity type (e.g., dam, foraging, scent mound), estimated age, distance from water, effected area, and river width/depth, among others.

Figure 2: Beaver Expansion



Notes: Beaver detection by survey year in study region. Observation units are 1km^2 landscape patches. Data source: NatureScot and affiliated survey contractors. See main text for details.

Over the decade of surveys, beaver populations expanded rapidly, more than doubling between each survey. The 2012 survey detected 39 beaver groups in the Tayside region. In 2018, this had risen to 114 groups. In 2021, 251 groups were found (Campbell-Palmer, R. et al. 2021). Fig. 2 shows beaver expansion throughout the tessellated study region. Grid cells are colored according to the first year in which a NatureScot survey detected any beaver sign. The observed pattern accords with ecological literature on beaver colonization. Dispersal occurs linearly along waterways, both downstream and upstream (Müller-Schwarze 2011), with separate groups initially far apart to avoid territorial conflict, then slowly filling over time (Hartman 1995).

2.2 Land Use

To capture spatiotemporal variation in agricultural land use, I employ the UK Center for Ecology & Hydrology's repeated Land Cover Map product (LCM) at 25m resolution, which exists for 1990, 2000, 2007, 2015, 2017, 2018, 2019, 2020, 2021, and 2022. The LCM covers the entirety of Great Britain and North Ireland, deriving its classifications from Sentinel-2 10-band seasonal composite

image patches[9], with additional context layers on height, aspect, slope, distance to built structures and water bodies, foreshore, and woodland to ejudicate spectral confusion (*The UKCEH Land Cover Map for 2022* 2024). Modern UKCEH models (since 2015) classify 10m pixels into 21 land use classes, based on the Biodiversity Action Plan Broad Habitats (Jackson 2000), using a random forest. As ground truth, UKCEH uses pixels classified in previous years with high accuracy (>80%) and no observed change over three consecutive years. Predictions made on 10m pixels are then aggregated to land parcels and from there rasterized at 25m resolution. I consider agricultural any pixel classified as "arable" in the UKCEH LCM, which includes cropped and freshly ploughed land (Fig. 3a). I do not include the closely related class of "improved grassland" due to its low recall and precision (74.3% and 91.1%, respectively) compared to "arable," which has the highest recall and second highest precision of any class (97.6% and 92.7%, respectively), when validated on a set of reference points sourced from countryside surveys, National Forest Inventory, Rural Payment Agency, manual Sentinel-2 image interpretation, and UKCEH field collection.[10] Nearly all false negative arable pixels were mislabeled as improved grassland (93.4%). Similarly, the majority of false positive arable-classified pixels were in fact improved grassland (55.8%). I aggregate the 25m "arable" mask (Fig. 3a) up to the $1km^2$ grid cells to produce a measure on the unit interval of agricultural land share, weighting by overlap proportion for raster cells not entirely contained within the $1km^2$ landscape patch (Fig. 3b).

2.3 Hydrometry

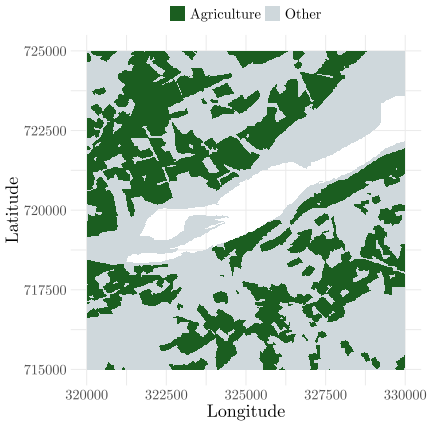
To detect beaver impacts on their physical environment, I obtain data on river characteristics. The Scottish Environmental Protection Agency (SEPA) operates a network of in-situ hydrometry monitoring stations, 167 of which reside in the study region. Stations collect data on rainfall, groundwater levels, river levels, river flow (or discharge), and tidal level. I extract historical records of river level, groundwater level, and river flow–all of which may be altered by beaver colonization. I drop groundwater level from analysis due to high missingness rates. I aggregate river level (recorded monthly) and river flow (recorded daily) to annual statistics to match the temporal resolution of

[9] Sentinel-2 was launched in 2015. 1990-2015 maps used Landsat 30m resolution imagery. In 2017, UKCEH revised its historical classifications to allow for comparability with post-2015 products.

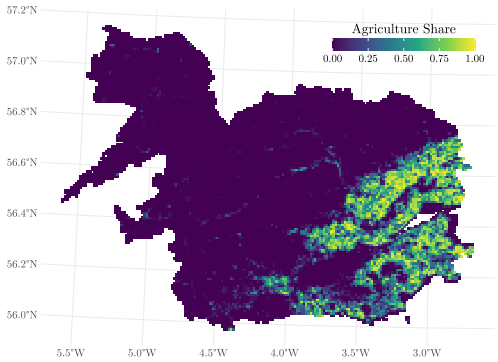
[10] This is largely due to the difficulty of differentiating between grassland types, including improved, neutral, calcareous, and acid, which exist on a spectral continuum.

Figure 3: Agricultural land use

(a) Agricultural land in 25m raster



(b) Agricultural raster aggregated to 1km^2 landscape grid cells



Notes: Data from the UK Center for Ecology & Hydrology Land Cover Map (25m rasterized land parcels, GB). Fig. 3a shows the extracted agricultural layer in a subregion around the Firth of Tay. Fig. 3b includes the entire study region. 2022 data is used for illustration. See main text for details.

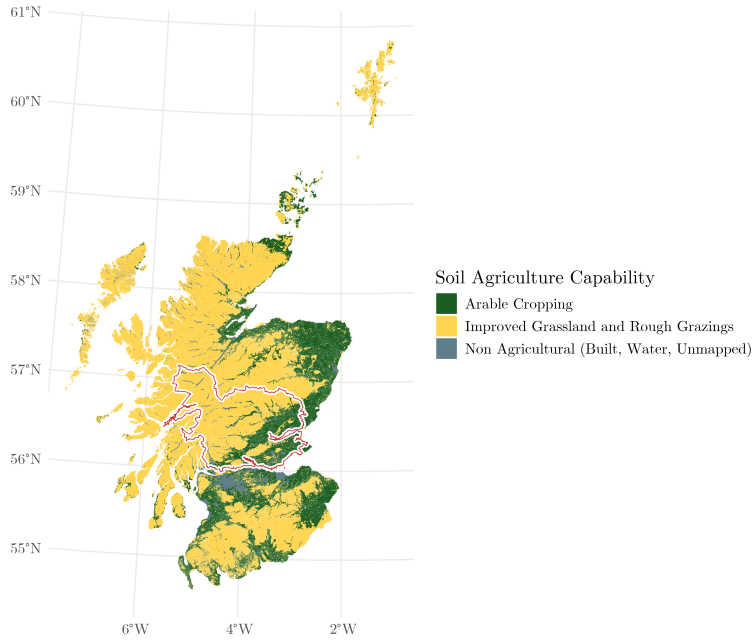
the beaver expansion data.

2.4 Soil

To distinguish between high and low agriculture-propensity areas, I use soil data from the James Hutton Institute. The soil map bins Scottish land into 13 main soil classes, ranging from soil suitable for a broad array of crops[11] to soil that could be used only for rough grazing (e.g., rugged or extremely wet terrain). An additional three classes capture built, water, and unmapped island areas. In Fig. 4, I group the 13 bins into three broad classes: Arable Cropping, Improved Grassland and Rough Grazings, and Non-Agricultural (including built, water, and unmapped areas). To achieve full cover, I employ two versions of the soil map: a 1:50,000 scale map that is considered definitive where coverage exists, mainly in eastern lowlands, where arable land is widespread (1987) and a 1:250,000 scale map that provides lower resolution coverage in highlands areas not mapped by the 50k scale map (1981). I calculate the share of each landscape grid cell that is occupied by a given class and determine the dominant class (>50%) in each cell.

[11] This highest cropping suitability category is characterized by "well-drained deep loam, sandy loams, silty loams or their related humic variants with good reserves of moisture. Sites are level or gently sloping and the climate is favourable. There are no or only very minor physical limitations affecting agricultural use" (Soil Survey of Scotland Staff 1981)

Figure 4: Soil agriculture capability classes



Notes: Data from James Hutton Institute. Classes have been aggregated from the 13 original classifications. Study region outlined in red. See main text for details.

2.5 Weather

To control for temporal variation in weather patterns, I extract temperature, precipitation, and vegetation cover records from the European Centre for Medium-Range Weather Forecasts' Reanalysis v5 (ERA5), which provides hourly estimates at 31km resolution, spanning 1950 to present (Hersbach et al. 2020). Using data from 1990 to 2022, I calculate annual total precipitation and average two-meter temperature. I assign ERA5 annual grid cell values to 1km^2 landscape patches intersecting the ERA5 cell, weighting by overlap proportion when multiple ERA5 cells intersect a single landscape patch.

3 Methods

I estimate response functions of both land used for agriculture and local environmental characteristics to beaver colonization.

I construct a two-period panel, using the 1km^2 landscape patches. I index calendar years as $t \in \{1990, ..., 2022\}$ and event-relative periods as $\gamma \in [0, 1]$ Year $t \in [1990, 2000] \mapsto \gamma = 0$. For

the sample including all treatment cohorts, $t \in [2020, 2022] \mapsto \gamma = 1$. For the sample including only 2012 and 2017 treated cohorts, $t \in [2017, 2022] \mapsto \gamma = 1$. For the sample including only the 2012-treated cohort, $t \in [2012, 2022] \mapsto \gamma = 1$. To rule out one potential violation of model assumptions, I calculate trends in agriculture land share in the pre-treatment years for which I have land use data (1990 and 2000). In Fig. A1, all the treated cohorts display similar, slightly negative trends over this time. While the control group is much less agriculture-intensive, its trend does not appear to significantly differ from those of the treated cohorts. In estimation, iteratively omitting cohorts does not meaningfully affect results.

Because beaver habitation is typically generations long, and dams can persist long after resident beavers abandon sites, I treat beaver arrival as an absorbing treatment.

I assign hydrometry monitoring stations to the grid cell in which they reside. Because the beaver survey data does not permit me to locate dams, I do not distinguish between upstream and downstream stations. Assuming a random distribution of underlying dam locations relative to monitoring station placement, the equal presence of downstream and upstream stations may cancel out any signal.

To measure the impact of beaver habitation on agricultural land use, I estimate the classic two-period difference-in-difference model

$$y_{i\gamma} = \beta^b D_{i\gamma} + \mathbf{X}_{i\gamma} + \alpha_i + \gamma + \epsilon_{i\gamma}, \tag{1}$$

where $y_{i\gamma}$ is the agricultural land use outcome, β^b is the effect of being treated by beaver presence, $D_{i\gamma}$ captures beaver treatment, $\mathbf{X}_{i\gamma}$ is a vector of local precipitation and temperature controls, and ϵ is a random error term. α_i and γ capture grid cell and pre-and post-period fixed effects, respectively.

4 Results

In Table 1, I report estimates from Equation (1) of agriculture land use changes following beaver entry.

In column (1), which includes all treatment cohorts and grid cells, beaver entry increases the share of land devoted to agriculture by 2.8 p.p. (22.4% relative to baseline). The effect is driven

Table 1
Impact of beaver habitation on land share in agricultural use by treatment cohort

	All Treated Cohorts		2012 and 2017 Cohorts		2012 Cohort	
	All cells	River cells	All cells	River cells	All cells	River cells
	(1)	(2)	(3)	(4)	(5)	(6)
Beaver Presence	0.028***	0.031***	0.030***	0.034***	0.027***	0.029***
	(0.005)	(0.006)	(0.006)	(0.007)	(0.009)	(0.009)
Observations	27,970	17,598	27,464	17,138	26,936	16,700
Within R^2	0.039	0.039	0.024	0.025	0.024	0.027
Mean Dep. Var.	0.125	0.125	0.122	0.120	0.118	0.114

Notes: Estimation results from Equation (1). Each regression includes grid cell and time period fixed effects. Samples vary by column. Regression includes average two-meter temperature and average total precipitation covariates. Standard errors are clustered at the grid cell level.
* 0.10 ** 0.05 *** 0.01

by cropping changes in landscape cells adjacent to rivers (cols (2), (4), and (6)), which supports anecdotal accounts that beaver effects remain localized to within tens of meters of their residence. Across the table, I vary the composition of the treatment group. In columns (1) and (2), I include all treated units. In columns (3) and (4), I remove the cohort treated in 2020. In columns (5) and (6), I further remove the 2017 treatment cohort. Across the cohort samples, the effect remains stable.

In Table 2, to test whether beaver entry effects vary by land type, I divide the sample by three soil types described in Section 2. Consistent with reports that beavers impact intensive cropping more than grasslands, in columns (3) and (4), the effect appears to be driven by activity in land classified as suitable for arable cropping. My preferred specification, column (4) includes all treatment cohorts but restricts the sample to only on-river cells with arable cropping soil. Here, beaver entry increases the proportion of cropped land by 4.6 p.p. (11.3% relative to the baseline, equivalent to 4.6 hectares). Columns (5) and (6), which report results on the subsample with soil suitable only for improved grassland or rough grazing, suggest beavers caused little to no change in agricultural cropping. This is consistent with the areas hosting little cropping (with a sample mean of <1% land area cropped). A further useful placebo test is shown in columns (7) and (8), which report results in non-agricultural soil areas, including built, water, and unmapped areas. If

Table 2
Impact of beaver habitation on land share in agricultural use by soil class

	All Soil Types		Arable		Grassland		Non Agri	
	All cells	River cells	All cells	River cells	All cells	River cells	All cells	River cells
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
Beaver Presence	0.028***	0.031***	0.045***	0.046***	-0.005	-0.003	-0.002	0.006
	(0.005)	(0.006)	(0.007)	(0.008)	(0.005)	(0.005)	(0.012)	(0.014)
Observations	27,970	17,598	7,668	5,128	18,982	11,534	1,312	934
Within R^2	0.039	0.039	0.037	0.045	0.032	0.032	0.060	0.061
Mean Dep. Var.	0.125	0.125	0.432	0.407	0.005	0.005	0.057	0.047

Notes: Estimation results from Equation (1). Each regression includes grid cell and time period fixed effects. Samples include all treatment cohorts. Regression includes average two-meter temperature and average total precipitation covariates. Standard errors are clustered at the grid cell level.
* 0.10 ** 0.05 *** 0.01

the remote-sensed land use data is trustworthy, one would expect to see no change in cropping in such areas. The results point to such a placebo test succeeding.

In case a small number of outlier units are driving the large positive results in Tables 1 and 2, I run a jackknife resampling exercise, in which I repeat the main specification in Table 2, column (4), leaving one landscape grid cell out each time. In Fig. A2, the resulting distribution is spread evenly around the coefficient reported in Table 2, with the tails extending approximately 0.1 p.p. in either direction.

To verify that beavers do, in fact, leave a physical imprint after entry, I perform a provisional test for changes in river characteristics. Using the hydrometry measures from in-situ monitoring stations, I regress river level and flow rate on beaver treatment using Equation 1. In Table A1, I report results from a small number of landscape patches which contain monitoring stations. Beaver entry causes a small but imprecise decrease in both river level and flow rate. While the qualitative effect is consistent with evidence that beaver dams lower water levels and peak flow rates downstream (Swinnen et al. 2019), the distribution of monitors relative to beaver dams is unknown, rendering interpretation unclear. Still, if one assumes *any* downstream-of-dam monitors in the data, then the effect shown may be a lower bound. But the results do not provide clear evidence of flooding–a potential damage mechanism–which would most likely appear as an increase

in the maximum observed river levels *upstream* of the dam.

5 Conclusion

Natural capital plays a crucial role in the economic production function, but less is known about the effect of wildlife considered to be nuisance species. While some provide beneficial ecosystem services, others disrupt economic operations. Because agents may not know the true direction on this effect *ex-ante*, many risk-averse operators will undertake potentially welfare-reducing control operations to neutralize potential threats. Using the case of unauthorized Scottish beaver reemergence, following a long period of local extinction, I provide evidence for the effect of one keystone species and ecosystem engineer on agricultural land use. On the extensive margin, land devoted to agriculture increases significantly in response to beaver entry, relative to comparison landscape patches that did not experience beaver habitation. Consistent with literature on beaver ecology and beaver-farm interactions, the positive effect is driven by landscape patches directly adjacent to watercourses with high arable cropping suitability. Using hydrometry measures, I observe suggestive evidence that beavers do indeed alter their physical environment, lowering flow rates and water levels, though the small sample and unknown spatial distribution of monitoring stations relative to beaver dams limits interpretability. Ongoing analysis aims to provide further evidence of the mechanism driving the reduced-form effect, includes tests of cropping reallocation and measurement error.

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Appendix

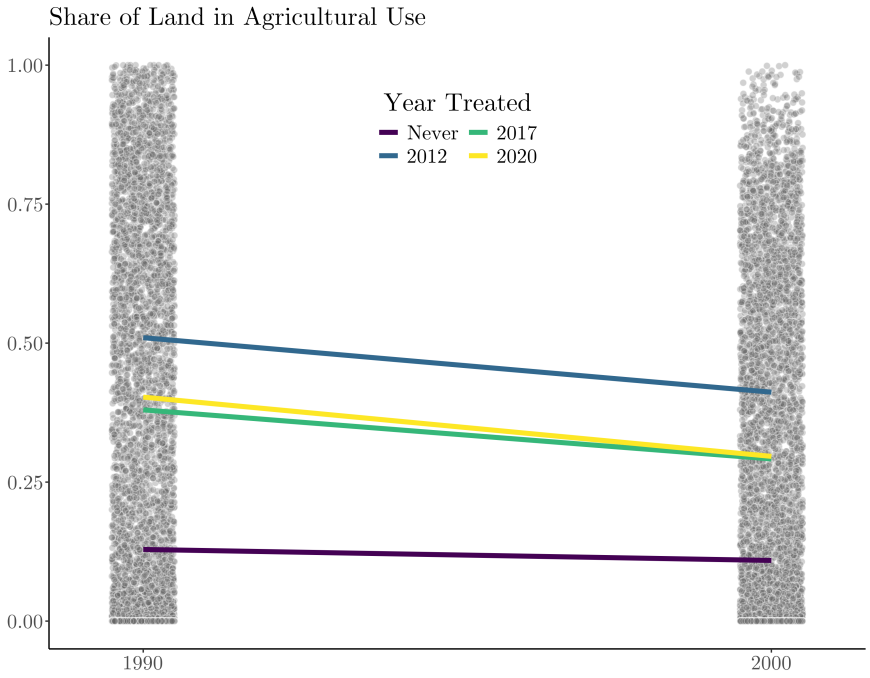
A Additional Results

Table A1
Impact of beaver habitation on agricultural land share, river level, and river flow rate

	Share Agri.	River Level (mean)	River Level (max)	River Flow (mean)
	(1)	(2)	(3)	(4)
Panel A: All grid cells				
Beaver Presence	0.028***	-0.042	-0.017	-0.149
	(0.005)	(0.027)	(0.070)	(0.266)
Observations	27,970	136	136	108
Within R^2	0.039	0.050	0.266	0.038
Mean Dep. Var.	0.125	0.683	2.354	17.907
Panel B: River grid cells				
Beaver Presence	0.031***	-0.041	-0.018	-0.149
	(0.006)	(0.027)	(0.071)	(0.266)
Observations	17,598	134	134	108
Within R^2	0.039	0.049	0.262	0.038
Mean Dep. Var.	0.125	0.573	2.251	17.907

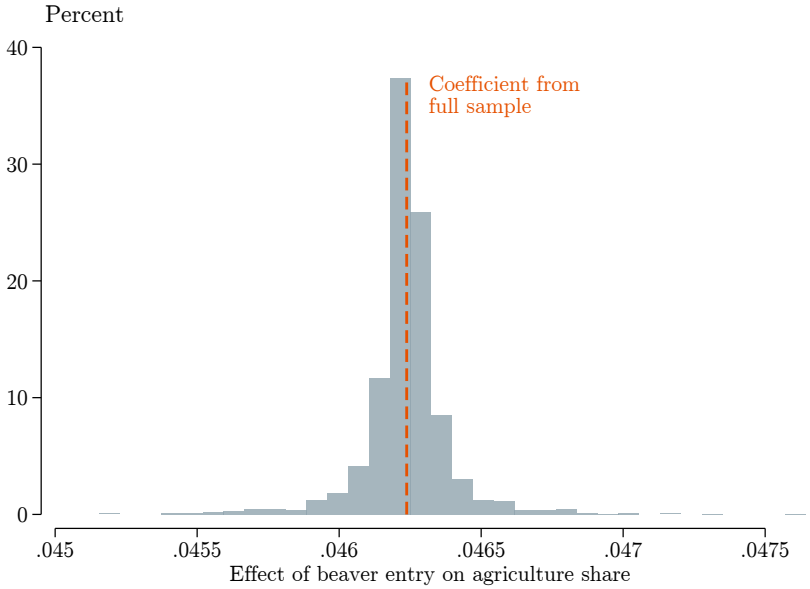
Notes: Estimation results from Equation (1). Each regression includes grid cell and time period fixed effects. Sample includes all treatment cohorts. Standard errors are clustered at the grid cell level.
* 0.10 ** 0.05 *** 0.01

Figure A1: Outcome pre-trends by treatment cohort.



Notes: Grey points represent landscape patches. The share of landscape patch classified as "arable" is plotted on the y-axis.

Figure A2: Jackknife leave-one-out resampling estimation



Notes: The distribution of coefficients from a jackknife resampling exercise, where one landscape patch is left out each time, is plotted on the x-axis. Full sample specification is shown in Table 2, column (4).