# Blended Finance for Biodiversity Conservation

Davide Stocco* Peter Tankov† David O. Zerbib‡

February 28, 2026

**Abstract**

We develop a continuous-time model of heterogeneous farms with two-dimensional state - size and biodiversity - to compare a blended finance credit instrument with conventional direct payments for biodiversity conservation. Under blended finance, the government co-pays a fraction of private borrowing costs, lowering the cost of capital and conditioning the cost of capital on ecological outcomes. Farms endogenously choose capital and subsidised intensity, which translates into biodiversity effort. We calibrate the model on EU agricultural data (FADN, Farmland Bird Index, GLOBIO4) and we find that blended finance raises expected biodiversity by 28% relative to current EU policies, reverses the net biodiversity decline, and achieves these gains at 84% lower fiscal cost, with a biodiversity leverage ratio approximately 50 times that of direct payments. The instrument is automatically progressive - small farms, which face higher borrowing costs, benefit most from the proportional subsidy - and participation is voluntary via self-selection. Blended finance dominates direct payments across all 27 EU member states, under both high and low interest-rate regimes, and even when up to 60% of the subsidy is captured by financial intermediaries. In regions with severe biodiversity decline, combining blended finance with targeted direct payments nearly doubles the biodiversity gain of either instrument alone.

**Keywords:** biodiversity, blended finance, agricultural policy, credit subsidies, Common Agricultural Policy.

## 1 Introduction

Biodiversity loss is one of the defining environmental challenges of the twenty-first century. The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) reports that approximately one million species face extinction, with the rate of loss accelerating over the past fifty years (IPBES, 2019). Agricultural land use is the single largest driver of terrestrial biodiversity decline: the conversion, simplification, and intensification of farmland have reduced habitat heterogeneity, eliminated landscape features, and increased chemical pressure on non-target species (Newbold et al., 2015;

*CREST, ENSAE, Institut Polytechnique de Paris. Email: davide.stocco@ensae.fr.
†CREST, ENSAE, Institut Polytechnique de Paris. Email: peter.tankov@ensae.fr.
‡CREST, ENSAE, Institut Polytechnique de Paris. Email: david.zerbib@ensae.fr.

Tilman et al., 2017). In Europe, the Eurostat Farmland Bird Index (FBI)—the official agri-environment-climate biodiversity indicator—documents a persistent decline of approximately 1.5–2% per year across EU 27 countries since 1990, with no structural break at any Common Agricultural Policy (CAP) reform date (Gregory et al., 2019).

The European Union allocates roughly EUR 55 billion per year to agricultural support through the CAP, of which Pillar I direct payments account for approximately 70%. These payments are predominantly area-based and only weakly conditioned on environmental outcomes through cross-compliance and greening requirements. The programme evaluation literature finds near-zero additionality for broad-based environmental conditionality (Chabé-Ferret and Subervie, 2013; Pe'er et al., 2014), consistent with the FBI evidence of continued biodiversity decline throughout successive CAP reforms. Agri-environment-climate schemes under Pillar II offer more targeted support, but they reach only a fraction of agricultural land and face challenges of adverse selection and limited ecological ambition (Wuepper and Huber, 2022).

Blended finance—the strategic use of public or philanthropic capital to mobilise private investment toward sustainable development outcomes (OECD, 2018)—has gained traction in climate mitigation through green bonds and concessional lending (Flammer, 2021; Flammer et al., 2024), but its application to biodiversity conservation remains largely unexplored. The fundamental challenge is to design a financial instrument that conditions private capital on ecological outcomes while maintaining market discipline and fiscal efficiency. Unlike carbon, where emissions can be priced through a single metric, biodiversity is multidimensional, spatially heterogeneous, and difficult to monitor at farm level. These features create both the need for and the difficulty of designing incentive-compatible credit instruments.

This paper develops a structural model to evaluate a blended finance instrument for farmland biodiversity conservation and compares it quantitatively with a direct public funding similar to the existing CAP direct payment system. A continuum of heterogeneous farms evolves in continuous time with two state variables: farm size $s_t$ (following geometric Brownian motion) and a biodiversity index $b_t \in [0, 1]$ (following a diffusion with endogenous drift). The government designs a credit subsidy that co-pays a fraction $\sigma_0$ of private borrowing costs and lowers the cost of capital depending on the farm's biodiversity by $\phi\, b$ per unit of subsidised capital. Farms optimally choose total capital, the share financed through the subsidised channel and biodiversity effort. A central planner selects the BF design parameters, namely the BF cost of capital and the costs associated to such investment, to maximise social welfare subject to a budget constraint benchmarked to the DP fiscal envelope.

Our main finding is that optimised blended finance welfare-dominates direct payments across every dimension we examine. In the low interest-rate environment (2015–2020), BF raises expected biodiversity by 28% relative to DP and is the only regime that reverses the net biodiversity decline, while requiring 84% less public spending (629 versus 4,003 EUR per farm per year). The biodiversity effort elicited per euro of fiscal cost is approximately 50 times higher under BF than under DP. The instrument is automatically progressive—the proportional subsidy delivers a larger absolute rate reduction to small, credit-constrained farms—and 63% of farms participate voluntarily through self-selection. The dominance of BF holds across both pre-crisis and low interest-rate regimes, across all 27 EU member states, and even when up to 60% of the credit subsidy is captured by financial intermediaries. However, in regions with severe biodiversity pressure—such as intensive arable areas of France—BF alone cannot fully offset the exogenous ecological decline; combining BF with targeted direct payments nearly doubles the biodiversity gain, revealing a complementarity between the two instruments under extreme conditions.

The paper makes four main contributions. First, we introduce a two-dimensional heterogeneous-agent framework in which both farm size and biodiversity are endogenous state variables, extending the canonical entry–exit models of Hopenhayn (1992) and Miao (2005) to incorporate ecological dynamics. The stationary cross-sectional distribution, obtained via Fokker–Planck methods, provides a coherent basis for aggregating farm-level decisions into economy-wide welfare comparisons. Second, we provide, to the best of our knowledge, the first micro-founded quantitative evaluation of blended finance as an environmental policy instrument, contrasting it with the existing direct payment system on welfare, biodiversity, output, farm structure, and fiscal cost dimensions. Third, the credit-based instrument features automatic progressivity and voluntary participation via self-selection —properties that are desirable for political feasibility and that distinguish it from mandated environmental regulations. Fourth, the calibration exploits multiple EU-level data sources (FADN, Farmland Bird Index, GLOBIO4, IACS/LPIS) to ground the model in observed agricultural and ecological patterns, and we validate the results across 232 NUTS2 regions under GLOBIO4 land-use change scenarios.

The paper relates to several strands of the literature. The biodiversity–agriculture nexus is studied by Tilman et al. (2017) and Moretti and Benzaquen (2024). The farm heterogeneity and structural transformation literature includes Adamopoulos and Restuccia (2014) and Adamopoulos et al. (2022), who document size-dependent distortions in agriculture. The subsidy pass-through literature—Lucca et al. (2019), Agarwal et al. (2018), Philippon (2015)—motivates our robustness analysis on intermediary rent extraction. The ecosystem services–productivity link, surveyed by Bareille and Largier (2026) and estimated by Liang et al. (2016), informs the biodiversity-output elasticity.

The remainder of the paper is organised as follows. Section 2 presents the model. Section 3 describes the calibration. Section 4 outlines the numerical methodology. Section 5 presents the results. Section 6 examines robustness. Section 7 concludes.

# 2 Model

We consider a continuous-time economy populated by a continuum of farms, each characterised by a two-dimensional state $z_t = (s_t, b_t)$, where $s_t > 0$ is farm size (hectares or a scale variable) and $b_t \in [0, 1]$ is a biodiversity index. Farms make within-period capital and participation decisions and an optimal exit decision modelled as a stopping time. A stationary cross-sectional distribution is sustained by exogenous entry and exit. A central planner designs the blended finance instrument and is benchmarked against a conventional direct payment scheme.

## 2.1 State dynamics

Farm size follows a geometric Brownian motion with constant drift $\mu_s$ and volatility $\sigma_s$:

$$ds_t = \mu_s \, s_t \, dt + \sigma_s \, s_t \, dW_t. \tag{1}$$

The drift $\mu_s > 0$ captures the secular trend toward larger farms driven by mechanisation and land consolidation (Adamopoulos and Restuccia, 2014; Hopenhayn, 1992); the volatility $\sigma_s$ reflects idiosyncratic shocks to productivity, weather, and market conditions.

The biodiversity index follows a diffusion with endogenous drift:

$$db_t = \big[\mu_b + a(s_t, b_t)\big]\, dt + \hat{\sigma}_b\, d\widehat{W}_t, \qquad b_t \in [0,1]. \tag{2}$$

The drift decomposes into two economically distinct components. The exogenous baseline degradation $\mu_b < 0$ captures the oberved decline in farmland biodiversity driven by landscape simplification, climate change, pesticide pressure, and habitat fragmentation—forces external to individual farm decisions that no single farmer can reverse. The endogenous drift $a(s,b) \geq 0$ captures the positive effect of farm-level conservation spending on ecological outcomes. The key economic implication is that maintaining biodiversity requires active investment: without conservation effort ($a = 0$), biodiversity degrades at rate $\mu_b$, and the policy problem is precisely to design instruments that elicit sufficient effort to offset or reverse this baseline decline.

The driving Brownian motions of size and biodiversity are correlated: $d\langle W, \widehat{W}\rangle_t = \rho\, dt$, $\rho \in [-1,1]$. Factors like weather, climate, and land-use shocks jointly affect farm profitability and biodiversity, creating co-movement between the two state variables.

## 2.2 Private credit pricing

The private loan interest rate is a size-dependent schedule motivated by default risk and lending frictions:

$$\delta(s) = r_f + \rho_{ag} + c_\delta\, s^{-\psi}, \qquad c_\delta > 0,\ \psi > 0, \tag{3}$$

where $r_f$ is the risk-free rate, $\rho_{ag}$ is the agricultural sector spread (illiquidity, sector risk), and $c_\delta\, s^{-\psi}$ is a size-dependent monitoring/information premium that decays as a power law in farm size. As $s \to \infty$, $\delta(s) \to r_f + \rho_{ag}$; as $s \to 0^+$, $\delta(s) \to \infty$, reflecting extreme informational frictions for the smallest holdings. The power-law specification generates a heavier tail than an exponential law, matching the empirical observation that spreads flatten gradually for large farms. Economically, the schedule captures the well-documented pattern that small farms face severe credit rationing: lenders charge higher rates to compensate for greater default risk, limited collateral, and higher per-unit monitoring costs (Stiglitz and Weiss, 1981; Petersen and Rajan, 1994). This size-dependent credit friction is a key mechanism in Adamopoulos et al. (2022), who show that it is a primary driver of farm size misallocation and agricultural productivity losses.

## 2.3 Policy instruments

### 2.3.1 Blended finance (BF)

The government co-pays a fraction $\sigma_0 \in (0,1)$ of the borrowing cost and adds a component which lowers the cost of capital for higher biodiversity levels ($\phi\, b$) per unit of subsidised capital. The farmer's subsidised interest rate is

$$\delta_{BF}(s,b) = \delta(s)(1-\sigma_0) - \phi\, b, \qquad \sigma_0 \in (0,1),\ \phi \geq 0, \tag{4}$$

so the per-unit subsidy rate is $\sigma(s,b) = \sigma_0\, \delta(s) + \phi\, b$. The minimum subsidised rate, attained at the largest farms with $b = 1$, is

$$\delta_{BF,\min} = \delta_{\min}(1-\sigma_0) - \phi, \qquad \delta_{\min} := \lim_{s\to\infty} \delta(s) = r_f + \rho_{ag}. \tag{5}$$

The economic content of (4) is best understood through its three design parameters. The proportional subsidy $\sigma_0$ reduces the borrowing cost by a fixed fraction: a farmer facing a private rate of 8% with $\sigma_0 = 0.40$ pays an effective rate of 4.8% before the biodiversity premium, representing a 320 basis-point reduction. Because $\sigma_0$ multiplies $\delta(s)$ and $\delta'(s) < 0$, the absolute rate reduction is larger for small farms (*progressivity*): the subsidy relaxes credit constraints precisely where they are most binding. The biodiversity premium $\phi\, b$ further reduces the borrowing cost for farms with high biodiversity, creating a continuous incentive gradient $\partial\delta_{\text{BF}}/\partial b = -\phi < 0$: improving biodiversity lowers the cost of capital at the margin (*biodiversity targeting*). Finally, because the subsidy scales with $\delta(s)$ and thus remains attractive even for the smallest farms, BF achieves broad coverage without mandating participation (*universal eligibility*).

### 2.3.2 Direct payments (DP)

The benchmark policy provides a per-hectare transfer:

$$DP(s) = \lambda\, s, \qquad \lambda \geq 0. \tag{6}$$

A linear rule mimics per-hectare support ($\lambda$) typical of CAP-style payments. DP serves both as a benchmark spending policy and as a reference budget constraint for BF. The aggregate DP fiscal cost is

$$C_{DP} = \iint_{s > y^*(b)} DP(s)\, f(s,b)\, ds\, db, \tag{7}$$

where $f(s,b)$ is the size-biodiversity distribution steady state of farms.

## 2.4 Farm decisions under BF

At each instant, a farm chooses the total capital $k \geq 0$ and a subsidised share $x \in [0,1]$, maximising its net profits. Biodiversity spending associated to BF products are linear in subsidised intensity: $\kappa_B(1-b)\, k\, x$, with $\kappa_B > 0$. We consider convex administrative costs in $x$ with scale economies: $\kappa_A\, k\, x^\gamma\, s^{-\chi}$, with $\kappa_A > 0$, $\gamma > 1$, $\chi \geq 0$.

A farmer's instantaneous profit under BF is

$$\Psi_{BF}(s,b;p) = \max_{k \geq 0,\, x \in [0,1]} \; p\, s^\alpha \left(\frac{k}{s}\right)^\beta (1+b)^\xi - [\delta(s) + r_d]\, k + x\, k\, \Delta(s,b) - \kappa_A\, k\, x^\gamma\, s^{-\chi} - c_F\, s^\zeta, \tag{8}$$

where the net per-unit gain from participating in the subsidised channel is

$$\Delta(s,b) := \underbrace{\sigma_0\, \delta(s) + \phi\, b}_{\text{credit benefit}} - \underbrace{\kappa_B(1-b)}_{\text{conditionality cost}}. \tag{9}$$

The first two terms represent the benefit of BF participation: the proportional rate reduction $\sigma_0\delta(s)$ and the biodiversity premium $\phi\, b$. The third term is the cost of the environmental conditionality attached to subsidised capital: farms must spend $\kappa_B(1-b)$ per unit on biodiversity, with the cost declining as biodiversity improves. A farm participates ($x^* > 0$) if and only if $\Delta(s,b) > 0$—that is, when the credit benefit outweighs the conditionality cost. Since $\delta'(s) < 0$, the gain is progressive: $\partial\Delta/\partial s = \sigma_0\, \delta'(s) < 0$, so smaller farms, which face higher private rates, have a larger incentive to participate.

For an interior $x \in (0,1)$, the first-order condition yields

$$x^*_{BF}(s,b) = \left( \frac{\Delta(s,b)\, s^{\chi}}{\kappa_A\, \gamma} \right)^{\frac{1}{\gamma-1}}, \tag{10}$$

clipped to $[0,1]$, with $x^*_{BF} = 0$ when $\Delta(s,b) \le 0$.

Evaluating the capital first-order condition at $x = x^*_{BF}$, we get the farmer's optimal capital $k^*_{BF}$ explicitely defined as

$$k^*_{BF}(s,b) = \left( \frac{\delta(s) + r_d - x^*\, \Delta(s,b) + \kappa_A\, (x^*)^{\gamma}\, s^{-\chi}}{p\, \beta\, (1+b)^{\xi}} \right)^{\frac{1}{\beta-1}} s^{\frac{\beta-\alpha}{\beta-1}}. \tag{11}$$

## 2.5 Farm decisions under DP

Under DP the farmer receives $DP(s) = \lambda\, s$ and borrows privately at rate $\delta(s)$. Instantaneous profit is

$$\Psi^{DP}(s,b;p) = \max_{k\ge 0}\, p\, s^{\alpha} \Big( \frac{k + DP(s)}{s} \Big)^{\beta} (1+b)^{\xi} - \delta(s)\, k - c_F\, s^{\zeta} \\ - r_d\, k - \kappa_{DP}(1-b)\, DP(s), \tag{12}$$

where $\kappa_{DP} > 0$ is a reduced-form intensity of biodiversity spending per euro of DP. The economic content of (12) highlights the fundamental difference between the two instruments. DP enters the production function as an unconditional capital supplement $k + DP(s)$, crowding out private borrowing one-for-one at the margin: the farmer reduces $k$ by the amount of the transfer and pockets the saved interest. Biodiversity effort under DP is governed by the term $\kappa_{\mathrm{DP}}(1-b)\, DP(s)$, which is mechanical rather than incentive-based: the farmer does not choose $\kappa_{\mathrm{DP}}$, and the FBI panel evidence implies it is near zero ($\kappa_{\mathrm{DP}} = 0.03$). BF, by contrast, elicits effort endogenously through the conditionality parameter $\kappa_B$ and the biodiversity premium $\phi$, creating a feedback loop between ecological outcomes and the cost of capital.

Let $\kappa := k + DP(s)$. The optimal total capital is

$$\kappa^*(s,b) = \left( \frac{\delta(s) + r_d}{p\, \beta\, (1+b)^{\xi}} \right)^{\frac{1}{\beta-1}} s^{\frac{\beta-\alpha}{\beta-1}}, \tag{13}$$

and private borrowing is $k^*_{DP}(s,b) = \max\{0,\, \kappa^*(s,b) - DP(s)\}$.

## 2.6 Endogenous biodiversity drift

A common biodiversity effectiveness parameter $\vartheta_B > 0$ maps effective biodiversity spending into a drift in biodiversity. Under BF, the biodiversity added drift $a_{BF}(s,b)$ is defined as the effort that the farm makes towards biodiversity conserving policies, i.e. how much it spends per unit of BF project aquired, times the efficacy of such spending:

$$a_{BF}(s,b) = \vartheta_B\, \frac{\kappa_B(1-b)\, k^*_{BF}(s,b)\, x^*_{BF}(s,b)}{s}. \tag{14}$$

Under DP, effective biodiversity spending is proportional to per-hectare transfers and the ecological gap:

$$a_{DP}(s,b) = \vartheta_B\, \kappa_{DP}\, (1-b)\, \lambda. \tag{15}$$

Both drifts are expressed per hectare, ensuring that the biodiversity index evolves at a rate independent of farm size for a given spending intensity. The economic distinction is crucial. Under BF, the drift $a_{\mathrm{BF}}$ depends on the farm's endogenous choices of capital $k^*_{\mathrm{BF}}$ and subsidised intensity $x^*_{\mathrm{BF}}$, both of which respond to the BF design parameters $(\sigma_0, \phi, \kappa_B)$ and to the farm's own state $(s, b)$. Under DP, the drift $a_{\mathrm{DP}}$ depends only on the exogenous policy rate $\lambda$ and the calibrated effectiveness $\kappa_{\mathrm{DP}}$; it does not vary with farm-level decisions. In particular, under DP the farms have no legal obligations in investing a certain amount in biodiversity policies, which is enforced under the BF framework. This is the core of the BF advantage: the credit channel creates a price signal that induces farms to internalise biodiversity as a factor affecting their cost of capital.

## 2.7 Optimal exit

Farms solve a zero-stopping-payoff optimal stopping problem:

$$V(s_0, b_0) = \sup_{\tau} \ \mathbb{E}^{s_0,b_0}\left[\int_0^{\tau} e^{-\varrho t}\, \Psi(s_t, b_t; p)\, dt\right], \qquad \varrho := r_f + \eta, \tag{16}$$

leading to a free boundary $y^*(b)$ such that exit occurs when $s \leq y^*(b)$. The value function satisfies the variational inequality:

$$\max\Big\{\mathcal{L}V(s,b) + \Psi(s,b;p) - \varrho\, V(s,b),\ -V(s,b)\Big\} = 0, \tag{17}$$

where the infinitesimal generator is

$$\mathcal{L} = \mu_s s\,\frac{\partial}{\partial s} + [\mu_b + a(s,b)]\,\frac{\partial}{\partial b} + \tfrac{1}{2}\sigma_s^2 s^2\,\frac{\partial^2}{\partial s^2} + \rho\,\sigma_s s\,\hat{\sigma}_b\,\frac{\partial^2}{\partial s\,\partial b} + \tfrac{1}{2}\hat{\sigma}_b^2\,\frac{\partial^2}{\partial b^2}. \tag{18}$$

The free boundary $y^*(b)$ is identified as the boundary of the region $\{(s,b) : V(s,b) > 0\}$: a farm exits when its size falls below $y^*(b)$, the threshold at which the present value of future profits reaches zero. The exit boundary is biodiversity-dependent: farms with higher $b$ enjoy higher output (through $(1+b)^{\xi}$) and, under BF, lower borrowing costs, so they survive at smaller sizes. Policy parameters that raise continuation values—a more generous $\sigma_0$ or higher $\phi$—shift $y^*(b)$ downward, extending the survival of small farms and increasing the mass of the stationary distribution.

## 2.8 Stationary distribution

Let $f(s,b)$ denote the stationary cross-sectional density of surviving farms. Farms exit endogenously when $s \leq y^*(b)$ and exogenously at hazard $\eta > 0$ (retirement, land conversion). New farms enter at rate $\nu > 0$ with density $g(s,b)$.

On the interior domain $\Omega := \{(s,b) : s > y^*(b),\ b \in (0,1)\}$, $f$ solves the stationary Fokker–Planck equation:

$$\begin{aligned} 0 = -\frac{\partial}{\partial s}\big(\mu_s s\, f\big) - \frac{\partial}{\partial b}\big([\mu_b + a(s,b)]\, f\big) + \frac{1}{2}\frac{\partial^2}{\partial s^2}\big(\sigma_s^2 s^2\, f\big) \\ + \frac{1}{2}\frac{\partial^2}{\partial b^2}\big(\hat{\sigma}_b^2\, f\big) + \rho\,\sigma_s\,\hat{\sigma}_b\,\frac{\partial^2}{\partial s\,\partial b}\big(s\, f\big) - \eta\, f + \nu\, g(s,b), \end{aligned} \tag{19}$$

with absorbing boundary at exit ($f = 0$ for $s \leq y^*(b)$), reflecting boundaries at $b \in \{0,1\}$, and unit mass normalisation. Economically, $f(s,b)$ describes the long-run cross-sectional distribution of farms across size and biodiversity that emerges from the interplay of individual decisions (capital, effort, exit) and aggregate forces (entry, exogenous shocks). The stationary distribution provides a coherent benchmark for computing aggregate welfare, fiscal costs, and biodiversity outcomes: all moments reported in the results are expectations under $f$, ensuring internal consistency between micro-level incentives and macro-level outcomes.

## 2.9 Policy costs and leverage

The BF fiscal cost per unit time is the subsidy rate times subsidised capital:

$$c_{BF}(s,b) = \big[\sigma_0\,\delta(s) + \phi\, b\big]\, x^*_{BF}(s,b)\, k^*_{BF}(s,b). \tag{20}$$

Aggregate BF spending is $C_{BF} = \iint_{s>y^*(b)} c_{BF}(s,b)\, f(s,b)\, ds\, db$.

Biodiversity leverage ratios quantify the ecological improvement elicited per euro of public spending:

$$\mathcal{L}^{bio}_{BF} = \frac{\mathbb{E}_f[a_{BF}]}{\mathbb{E}_f[c_{BF}]}, \qquad \mathcal{L}^{bio}_{DP} = \frac{\mathbb{E}_f[a_{DP}]}{\mathbb{E}_f[DP(s)]}. \tag{21}$$

The ratio measures how effectively each instrument converts fiscal resources into biodiversity outcomes. A high leverage ratio indicates that the instrument mobilises substantial private co-investment: under BF, the government pays only the spread between private and subsidised rates, while the farmer finances the remainder at the reduced rate, so each public euro leverages private capital toward conservation.

## 2.10 Central planner

The central planner chooses BF policy parameters $\pi = (\sigma_0, \phi, \kappa_B)$. The programme structure parameter $\gamma$ (administrative cost convexity) is held fixed as a structural feature of the BF instrument; $\kappa_B$ (biodiversity spending intensity) is optimised, representing the chosen level of environmental conditionality.

Under policy $\pi$, the output function evaluated at optimal capital is

$$q_{BF}(s,b) = p\, s^{\alpha}\Big(\frac{k^*_{BF}(s,b)}{s}\Big)^{\beta}(1+b)^{\xi}. \tag{22}$$

The planner observes aggregate moments $\mathbb{E}_\pi[q]$, $\mathbb{E}_\pi[b]$, and $\mathrm{Var}_\pi(b)$ computed against the stationary distribution $f_\pi$.

The total cost of the BF programme combines a linear deadweight loss of taxation, a convex programme implementation cost, and a soft anti-arbitrage penalty:

$$\mathcal{C}(C_{BF}, \sigma_0, \varphi) = \kappa_1\, C_{BF} + c_{\text{prog}}\, \sigma_0^{\gamma_{\text{prog}}} + c_{\text{rate}}\, \big[\max\big(0,\ v(\sigma_0,\varphi)\big)\big]^{\gamma_{\text{rate}}}, \tag{23}$$

where

$$v(\sigma_0,\varphi) = \frac{r_f - \delta_{BF,\min}(\sigma_0,\varphi)}{\rho_{ag}} \tag{24}$$

measures how far below $r_f$ the minimum BF rate has dropped, normalised by the agricultural spread. The three terms capture distinct economic costs. The linear term $\kappa_1 C_{\text{BF}}$ is the deadweight loss of raising public revenue through distortionary taxation ($\kappa_1 \approx 0.30$): each euro of BF spending costs society $1+\kappa_1$ euros. The programme cost $c_{\text{prog}}\, \sigma_0^{\gamma_{\text{prog}}}$ captures institutional and administrative expenses that scale convexly with programme generosity—enforcement, compliance verification, ecological monitoring, and political economy costs. The convexity ($\gamma_{\text{prog}} > 1$) reflects the empirical regularity that programme costs accelerate as coverage expands (Heckman and Smith, 2004), ensuring an interior optimum for $\sigma_0$. The soft anti-arbitrage penalty discourages designs where the subsidised rate falls below $r_f$: if $\delta_{\text{BF}} < r_f$, a farmer could borrow at the subsidised rate and lend at the risk-free rate, earning a riskless profit. The penalty allows small crossings (reflecting monitoring imprecision) while making large violations prohibitively expensive.

The planner objective is identified by the following problem

$$\begin{aligned} \max_{(\sigma_0,\phi,\kappa_B)} \quad & W(\sigma_0,\phi,\kappa_B) = w_q\,\mathbb{E}_\pi[q] + w_b\,\mathbb{E}_\pi[b] - w_v\,\mathrm{Var}_\pi(b) - \mathcal{C}\big(C_{BF}(\pi),\sigma_0,\varphi\big) \\ \text{s.t.} \quad & C_{BF}(\pi) \le C_{DP}(\lambda;\pi), \\ & \sigma_0 \in (0,1),\ \phi \ge 0,\ \kappa_B \ge 0. \end{aligned} \tag{25}$$

The welfare function reflects four normative considerations. The term $w_q\,\mathbb{E}[q]$ values aggregate food output. The term $w_b\,\mathbb{E}[b]$ values biodiversity as natural capital, reflecting the social willingness to pay for ecosystem services (pollination, pest control, soil health) that farmland biodiversity provides. The penalty $w_v\,\mathrm{Var}(b)$ discourages policies that raise average biodiversity at the cost of spatial fragmentation —a concern in conservation biology, since connected habitats are ecologically more valuable than isolated patches. The cost function $\mathcal{C}$ captures the deadweight loss of taxation, institutional programme costs, and the anti-arbitrage penalty. The hard budget constraint $C_{\text{BF}} \le C_{\text{DP}}$ ensures that BF is evaluated against the same fiscal envelope as the existing DP system, so that any welfare improvement reflects genuine efficiency gains rather than higher public spending.

# 3 Calibration

## 3.1 Data sources

We consider two datasets for calibrating the farm-size distribution: the Eurostat Farm Structure Survey and the Integrated Administration and Control System.

The Eurostat Farm Structure Survey (FSS) provides cell-level averages of Utilised Agricultural Area (UAA) in hectares, representative at the NUTS 2 level and covering at least 98% of the total UAA in each Member State.[1] Published aggregates are binned by size class. The IACS/LPIS (Integrated Administration and Control System / Land Parcel Identification System) provides georeferenced, parcel-level data on agricultural land use, including exact hectare measurements with polygon boundaries. A harmonised dataset covering 19 EU Member States with time series up to 17 years has been published by the Europe-LAND Horizon Europe project.[2]

We adopt a mixed-source strategy: IACS/LPIS parcel-level data for the 19 available countries, and FSS aggregates for the remaining Member States. Farm size enters the entry distribution as $s \sim \chi^2(\nu_s)$, capturing the right-skewed size distribution of European farms.

Two complementary indicators are used for what concerns biodiversity.

The Eurostat Farmland Bird Index (FBI, dataset `env_bio2`) provides annual country-level indices of common farmland bird populations for 24 EU/EFTA countries over 1990–

[1] Eurostat FSS metadata: `https://ec.europa.eu/eurostat/cache/metadata/en/ef_sims.htm`.
[2] Harmonised IACS inventory: `https://zenodo.org/records/15692199`.

2023. The FBI is the European Commission's primary agri-environment-climate biodiversity indicator. The Mean Species Abundance (MSA) from the GLOBIO 4 model (Schipper et al., 2020) measures the mean abundance of original species relative to undisturbed ecosystems. Available at 300 m resolution, it integrates land-use intensity, infrastructure, nitrogen deposition, and climate pressures, with each cell scored in $[0, 1]$.

The biodiversity index FBI enters the entry distribution as $b \sim \text{Beta}(\alpha_g, \beta_g)$, with shape parameters reflecting the ecological state of European farmland at NUTS 2 level. Both marginals are combined via a Gaussian copula with correlation $\rho$. The MSA is considered to analyse further BF policy under stressed scenarios.

## 3.2 Parameter estimation

Table 1 summarises all model parameters, their role, calibration source, and classification.

The GBM parameters in (1) are estimated from the FADN panel at NUTS 2 level (2007–2020). Cross-sectional dispersion of $\ln s$ divided by $\sqrt{T}$ ($T \approx 25$ years) yields $\sigma_s$; the drift $\mu_s$ is recovered from the time trend with the Itô correction $\mu_s = \hat{\mu} + \sigma_s^2/2$, ensuring consistency between the log-normal model and observed mean growth rates.

The biodiversity drift $\mu_b$ in (2) is estimated from the FBI 1990–2023 trend. We normalise FBI values to $[0, 1]$ and compute year-on-year changes for each country. The exogenous drift is $\mu_b = \overline{\Delta b}$, reflecting the persistent secular decline in farmland biodiversity across EU member states. The volatility is the median of per-country temporal standard deviations: $\hat{\sigma}_b = \text{median}_c \, \text{std}_t(\Delta b_{c,t})$.

The elasticities in (8) and (12) are estimated from a Cobb–Douglas specification:

$$\ln q_i = \ln p + \alpha \ln s_i + \beta \ln(k_i/s_i) + \xi \ln(1 + b_i) + \text{fixed effects} + \varepsilon_i,$$

using weighted least squares on FADN data at NUTS 2 level. Farm Net Value Added (SE410) is the dependent variable; capital $k = \text{SE455} + \text{SE460}$ (buildings plus machinery). The restriction $\alpha > \beta$ ensures decreasing returns to capital intensity $k/s$ at given farm size, while $\alpha + \beta < 1$ implies decreasing returns to scale in the two conventional factors—consistent with the standard result in the agricultural economics literature that farm-level production exhibits mild decreasing returns (Adamopoulos and Restuccia, 2014).

The biodiversity-productivity elasticity $\xi$ is estimated via a two-step Mundlak (1978) decomposition to separate genuine ecosystem-service effects from endogenous sorting (intensive farms in low-biodiversity regions). The between-country component captures time-invariant cross-country differences in ecosystem-service provision, while the within-country component absorbs endogenous farm-type sorting. Our estimate is consistent with the meta-analytic evidence of Bareille and Largier (2026) and Liang et al. (2016).

The output price $p$ is extracted as the geometric mean of the residual scale factor from the regression intercept and fixed effects, ensuring internal consistency with cost parameters denominated in euros.

The parameters $c_F$ and $\zeta$ are estimated by regressing FADN overheads (SE336) on farm size in a log-log specification. The sub-linear estimate $\zeta < 1$ implies economies of scale in overhead, consistent with Hopenhayn (1992) and Adamopoulos and Restuccia (2014). The depreciation rate $r_d$ is a weighted average across asset types: machinery (12%, weight 0.6), buildings (3.5%, weight 0.3), and biological assets (8%, weight 0.1). The risk-free rate $r_f$ is set to the average ECB main refinancing rate over the estimation period.

The three-parameter power-law schedule (3) is fitted by nonlinear least squares to FADN effective interest rates (SE380/SE485) by economic size class. ECB bank lending survey rates serve as out-of-sample validation. The estimated schedule is steeply decreasing: the smallest size class faces rates three to five times higher than the largest, consistent with the size-dependent credit rationing documented by Adamopoulos et al. (2022).

In the six IACS countries with farm-level identifiers (DE, FR, ES, IT, NL, PL), farm survival and entry are tracked year-by-year. The exogenous exit rate $\eta$ captures retirement, land conversion, and non-economic exit; it is isolated by subtracting the endogenous exit rate produced by the model's free boundary $y^*(b)$ from the observed total exit rate. The entry rate $\nu$ is jointly calibrated with $\eta$ via the stationarity condition that ensures mass normalisation of the Fokker–Planck distribution (19).

The DP effectiveness $\kappa_{DP}$ is calibrated from the FBI regression evidence presented in Section 3.3: the absence of any structural break in the FBI decline at CAP reform dates implies near-zero ecological additionality of existing conditionality.

The admin cost level $\kappa_A$ is calibrated from Philippon (2015), who measures the unit cost of financial intermediation at approximately 1.5–2% per year. The scale economy exponent $\chi$ is calibrated from Featherstone and Moss (1994) and Gloy et al. (2005), who document significant economies of scale in agricultural banking.

The parameter $\vartheta_B$ in (14) and (15) converts monetary biodiversity spending into a drift contribution in the biodiversity SDE. Wuepper and Huber (2022) compare action- and results-based payments in Switzerland, finding that results-based schemes deliver higher biodiversity outcomes per euro; converting their estimated elasticities to our drift parametrisation yields the calibrated value. This parameter governs how effectively financial incentives translate into ecological outcomes and is therefore a key determinant of the relative performance of BF and DP.

The per-hectare DP rate is directly observable from CAP Pillar I payment data (DG AGRI), computed as a weighted average across EU27 Member States.

The welfare weights $w_q$, $w_b$, $w_v$ and cost parameters $\kappa_1$, $c_{\text{prog}}$, $\gamma_{\text{prog}}$, $c_{\text{rate}}$, $\gamma_{\text{rate}}$ in (25) and (23) are chosen normatively. Since $\mathbb{E}[q]$ is in EUR/farm ($\sim$30,000) while $\mathbb{E}[b]$ is dimensionless ($\sim$0.3), $w_b$ must carry units of EUR per biodiversity unit to render the two objectives commensurable. The biodiversity weight is anchored to the TEEB (2010) grassland ecosystem service valuations, which estimate the annual flow of pollination, water regulation, and soil fertility services at several thousand EUR per hectare—translating into the calibrated $w_b$ when scaled to the model's per-farm accounting. The deadweight loss parameter $\kappa_1 \approx 0.30$ is standard in the public finance literature (Heckman and Smith, 2004). All normative parameters are explored via sensitivity analysis.

Table 1: Parameter calibration overview.

| Parameter | Symbol | Description | Data source / method | Type |
|---|---|---|---|---|
| *Stochastic dynamics* | | | | |
| Size drift | $\mu_s$ | GBM drift of farm size | FADN NUTS 2 panel (2007–2020); time trend in mean $\ln s$ + Itô correction | Calibrated |
| Size volatility | $\sigma_s$ | GBM volatility of farm size | Cross-sectional $\text{std}(\ln s)/\sqrt{T}$, $T \approx 25\,\text{yr}$ | Calibrated |
| Biodiversity drift | $\mu_b$ | Exogenous baseline degradation | FBI 1990–2023 trend; no structural break at any CAP reform | Calibrated |
| Biodiversity vol. | $\hat{\sigma}_b$ | Diffusion volatility of $b_t$ | FBI median per-country $\text{std}(\Delta b)$; 24 countries | Calibrated |
| Correlation | $\rho$ | Brownian correlation | Spearman rank at NUTS 2 between farm size and biodiversity; sensitivity over $\rho \in [-0.5, 0]$ | Calibrated |
| *Production function* | | | | |
| Scale elasticity | $\alpha$ | Combined land + capital ($\alpha = \hat{a}+\hat{b}$) | WLS on FADN NUTS 2: $\ln(\text{FNVA}) \sim \ln s + \ln(k/s)$ + FE | Calibrated |
| Capital-intensity | $\beta$ | Returns to capital per ha ($\alpha > \beta$) | Same WLS regression | Calibrated |
| Ecosystem service | $\xi$ | Output gain from biodiversity via $(1+b)^{\xi}$ | Two-step Mundlak on FADN×MSA; consistent with Bareille and Largier (2026), Liang et al. (2016) | Calibrated |
| Output price | $p$ | Scale factor | Geometric mean of $\text{FNVA}/(s^{\alpha}(k/s)^{\beta})$ | Calibrated |
| *Costs* | | | | |
| Fixed cost level | $c_F$ | Operating fixed cost | FADN overheads (SE336): WLS $\ln \text{SE336} \sim \ln s$ + FE | Calibrated |
| Fixed cost exponent | $\zeta$ | Scaling of fixed cost in $s$ | Same regression; consistent with Hopenhayn (1992) | Calibrated |
| Depreciation | $r_d$ | Capital depreciation rate | Weighted average: machinery 12%, buildings 3.5%, biological 8% | Calibrated |
| Discount rate | $r_f$ | Risk-free rate | ECB main refinancing rate; $\varrho = r_f + \eta$ | Calibrated |
| *Private credit pricing:* $\delta(s) = r_f + \rho_{ag} + c_{\delta} s^{-\psi}$ | | | | |
| Sector spread | $\rho_{ag}$ | Agri risk premium over $r_f$ | NLS power-law fit on FADN SE380/SE485 by size class; ECB MIR validation | Calibrated |
| Size-premium coeff. | $c_{\delta}$ | Monitoring/info premium level | Same NLS fit | Calibrated |
| Size-premium exp. | $\psi$ | Power-law decay of risk premium | Same NLS fit; Stiglitz and Weiss (1981), Adamopoulos et al. (2022) | Calibrated |
| *Population dynamics* | | | | |
| Exit rate | $\eta$ | Hazard of farm exit | Eurostat structural statistics; Hopenhayn (1992) | Calibrated |
| Entry rate | $\nu$ | Rate of new farm creation | Stationarity condition | Calibrated |
| Entry size | $\nu_s$ | Chi-squared shape for $s$ entry | IACS/LPIS parcels (19 countries) or FSS size histograms | Calibrated |

*Continued on next page*

*Table 1 continued from previous page*

| Parameter | Symbol | Description | Data source / method | Type |
|---|---|---|---|---|
| Entry biodiversity | $(\alpha_g, \beta_g)$ | Beta shape for $b$ entry | FBI or MSA scores at NUTS 2; cross-validated | Calibrated |
| *Direct payments (observable policy)* | | | | |
| DP rate | $\lambda$ | Per-hectare direct payment | CAP Pillar I data (DG AGRI); EU27 weighted average | Calibrated |
| *Biodiversity spending and admin.* | | | | |
| DP bio-spending | $\kappa_{DP}$ | Bio-spending per euro of DP | FBI: no structural break at 2013 greening | Calibrated |
| Admin cost level | $\kappa_A$ | BF administrative cost | Philippon (2015): intermediation cost $\approx$ 1.5–2%/yr | Calibrated |
| Scale economy | $\chi$ | Admin cost reduction with size | Featherstone and Moss (1994), Gloy et al. (2005) | Calibrated |
| *Biodiversity effectiveness* | | | | |
| Effectiveness | $\vartheta_B$ | Spending-to-biodiversity conversion | Wuepper and Huber (2022): AES elasticities | Calibrated |
| *Blended finance (optimised)* | | | | |
| Proportional subsidy | $\sigma_0$ | Fraction of $\delta(s)$ co-paid | Optimised by planner; constraint (5) | Policy |
| Biodiversity premium | $\phi$ | $b$-dependent rate reduction | Optimised by planner | Policy |
| Policy stringency | $\gamma$ | Convexity of admin costs in $x$ | Structural: baseline $\gamma = 2$; sensitivity analysis | Structural |
| BF bio-spending | $\kappa_B$ | Biodiversity spending per unit of subsidised capital | Optimised by planner | Policy |
| *Planner preferences (normative)* | | | | |
| Output weight | $w_q$ | Weight on aggregate output | Output in EUR | Normative |
| Biodiversity weight | $w_b$ | Social value of biodiversity | TEEB (2010); sensitivity analysis | Normative |
| Variance penalty | $w_v$ | Penalty on biodiversity dispersion | Sensitivity analysis | Normative |
| Linear fiscal cost | $\kappa_1$ | DWL of taxation | Fiscal multiplier literature | Normative |
| Programme cost level | $c_{\text{prog}}$ | Institutional cost of BF | Enforcement and compliance costs | Normative |
| Programme cost conv. | $\gamma_{\text{prog}}$ | Convexity in $\sigma_0$ | Ensures interior optimum; Heckman and Smith (2004) | Normative |
| Anti-arbitrage pen. | $c_{\text{rate}}$ | Soft penalty for $\delta_{BF} < r_f$ | Normalised by $\rho_{ag}$ | Normative |
| Anti-arbitrage conv. | $\gamma_{\text{rate}}$ | Convexity of AA penalty | Allows small violations | Normative |

### 3.3 Empirical evidence: CAP reforms and farmland biodiversity

The model assigns a near-zero ecological effectiveness to direct payments ($\kappa_{\text{DP}} = 0.03$) and treats the biodiversity drift $\mu_b < 0$ as an exogenous secular force unaltered by existing policy. These are strong assumptions. This subsection documents the empirical basis for both, exploiting two identification strategies applied to the Eurostat Farmland Bird Index (FBI) and to GLOBIO 4 satellite land-use data.

#### 3.3.1 Structural break analysis.

The European Union undertook two major reforms of agricultural environmental conditionality during our sample period: the 2003 Mid-Term Review, which introduced cross-compliance linking direct payments to environmental standards, and the 2013 CAP reform, which added explicit Pillar I greening requirements (crop diversification, permanent grassland maintenance, ecological focus areas). If these reforms generated meaningful biodiversity additionality, the FBI—the European Commission's official agri-environment-climate indicator—should exhibit a structural break in its trend at one or both dates.

We test this hypothesis by estimating piecewise linear trends in $\ln(\text{FBI}_{it})$ with country fixed effects:

$$\ln(\text{FBI}_{it}) = \alpha_i + \beta\, t + \gamma_1\, \mathbb{1}_{t\geq 2003} \cdot (t - 2003) + \gamma_2\, \mathbb{1}_{t\geq 2013} \cdot (t - 2013) + \varepsilon_{it}, \tag{26}$$

where $\alpha_i$ absorbs time-invariant country characteristics and $\gamma_1$, $\gamma_2$ capture slope changes at the two reform dates. Standard errors are clustered at the country level throughout.

Table 2 reports the results. The baseline decline is 1.2% per year (column 2, $p < 0.01$), and neither break coefficient is individually or jointly significant: the $F$-test for $\gamma_1 = \gamma_2 = 0$ yields $p = 0.71$. Far from attenuating the decline, the point estimates indicate that biodiversity loss *accelerated* after the 2003 reform ($\hat{\beta} + \hat{\gamma}_1 = -1.7\%/\text{yr}$) and continued at $-1.6\%/\text{yr}$ after the 2013 greening requirements entered into force. These findings are robust to using 2015—the actual implementation year of the greening payment—as the break date (column 3), to allowing level shifts at reform dates (column 4), and to expanding the sample from 12 to 15 countries with shorter time series (column 5). In every specification, the trend coefficient $\hat{\beta}$ is precisely estimated and the break coefficients are individually and jointly insignificant.

#### 3.3.2 Subsidy intensity and biodiversity outcomes.

The structural break analysis tests the timing of policy effects; a complementary approach tests whether the *intensity* of CAP support predicts biodiversity across countries. We regress $\ln(\text{FBI}_{it})$ on $\ln(\lambda_{\text{CAP},it})$, the log of Pillar I direct payments per hectare, over 2013–2023 (the period for which CAP expenditure data are available at country level).

Table 3 presents four specifications: pooled OLS (column 1), country fixed effects (column 2), two-way fixed effects (column 3), and first differences (column 4). In all four, the elasticity of the FBI with respect to subsidy intensity is economically small, negative in sign, and statistically indistinguishable from zero. The country fixed-effects estimate ($\hat{\beta} = -0.051$, SE $= 0.069$) implies that a doubling of per-hectare CAP spending is associated with a *decrease* of 5% in the FBI, though this estimate is far from significant. These cross-sectional and within-country results reinforce the structural break evidence: not only has CAP reform failed to arrest the aggregate biodiversity decline, but countries that spend more per hectare on direct payments do not achieve better ecological outcomes. The

Table 2: Farmland Bird Index: no structural break at CAP reforms

| | Long sample (12 countries) | | | | Broad sample |
|---|---|---|---|---|---|
| | (1) | (2) | (3) | (4) | (5) |
| $t$ (annual trend) | $-0.0152^{***}$ | $-0.0116^{***}$ | $-0.0115^{***}$ | $-0.0128^{***}$ | $-0.0157^{***}$ |
| | (0.0027) | (0.0035) | (0.0035) | (0.0035) | (0.0044) |
| $t \times \mathbb{1}_{t\geq 2003}$ | | $-0.0055$ | $-0.0059$ | $-0.0059$ | $-0.0026$ |
| | | (0.0078) | (0.0072) | (0.0073) | (0.0063) |
| $t \times \mathbb{1}_{t\geq 2013}$ | | $+0.0007$ | | $+0.0021$ | $+0.0046$ |
| | | (0.0099) | | (0.0091) | (0.0086) |
| $t \times \mathbb{1}_{t\geq 2015}$ | | | $+0.0019$ | | |
| | | | (0.0115) | | |
| $\mathbb{1}_{t\geq 2003}$ | | | | $+0.0196$ | |
| | | | | (0.0206) | |
| $\mathbb{1}_{t\geq 2013}$ | | | | $+0.0080$ | |
| | | | | (0.0202) | |
| Country FE | Yes | Yes | Yes | Yes | Yes |
| Observations | 362 | 362 | 362 | 362 | 439 |
| Countries | 12 | 12 | 12 | 12 | 15 |
| $R^2$ | 0.754 | 0.756 | 0.756 | 0.756 | 0.744 |
| $F$-test (breaks = 0) | | 0.35 | 0.38 | 0.51 | |
| $p$-value | | [0.710] | [0.696] | [0.731] | |

*Notes:* Dependent variable: $\ln(\text{FBI}_{it})$, normalised to $\text{FBI}_{i,2000} = 100$. Standard errors clustered by country in parentheses. The long sample includes 12 EU/EFTA countries with continuous FBI data from at least 1998 to 2020; the broad sample extends to all countries with $\geq 15$ annual observations spanning 2000–2018. The $F$-test reports the joint test of all break coefficients equal to zero. $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$.

findings are consistent with the programme evaluation literature documenting near-zero additionality of broad-based environmental conditionality (Chabé-Ferret and Subervie, 2013; Pe'er et al., 2014; Wuepper and Huber, 2022).

#### 3.3.3 Land-use composition, biodiversity, and CAP spending.

A natural question is whether the FBI decline reflects land conversion (the extensive margin) or intensification within existing agricultural land (the intensive margin). We investigate using GLOBIO 4 land-use maps at 300 m resolution (Schipper et al., 2020), matched to the FBI panel at country-year level. For each of 25 EU countries, we compute the agricultural share (cropland + pasture as a fraction of total land area) and the natural share (forests, shrubland, grassland, wetlands, and secondary vegetation).

Table 4 provides three findings. Columns (1)–(3) regress $\ln(\text{FBI}_{it})$ on land-use composition in pooled cross-section (within-country land-use variation accounts for only 14% of total variation over six quinquennial observations, making fixed-effects identification unreliable). The cross-sectional gradient is as expected: countries with larger agricultural shares have lower FBI ($\hat{\beta} = -0.053$, $p < 0.01$), while higher natural shares predict higher FBI ($\hat{\beta} = +0.080$, $p < 0.01$). Crucially, the annual trend of $-1.6\%$/yr persists after controlling for land-use composition, implying that the biodiversity decline operates through intensification within existing agricultural land—habitat simplification, pesticide

Table 3: CAP subsidy intensity and farmland biodiversity

| | (1) Pooled | (2) Country FE | (3) Two-way FE | (4) First diff. |
|---|---|---|---|---|
| $t$ (trend) | $-0.0193^{***}$ | $-0.0202^{***}$ | | |
| | (0.0061) | (0.0053) | | |
| $\ln(\lambda_{\text{CAP}})$ | $-0.0132$ | $-0.0506$ | $-0.0154$ | |
| | (0.0322) | (0.0694) | (0.0551) | |
| $\Delta \ln(\lambda_{\text{CAP}})$ | | | | $-0.0262$ |
| | | | | (0.0413) |
| Country FE | No | Yes | Yes | Yes |
| Year FE | No | No | Yes | — |
| Observations | 138 | 138 | 138 | 122 |
| $R^2$ | 0.080 | 0.880 | 0.891 | 0.083 |

*Notes:* Columns (1)–(3): dependent variable is $\ln(\text{FBI}_{it})$; column (4): $\Delta \ln(\text{FBI}_{it})$. $\lambda_{\text{CAP}}$ denotes CAP Pillar I subsidy intensity (EUR/ha). Robust standard errors in column (1); standard errors clustered by country in columns (2)–(4). Sample: 16 EU countries, 2013–2023 (overlap of FBI and CAP expenditure data). $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$.

use, and reduction of landscape features—rather than through land conversion. This is an important distinction for policy design: it means that instruments targeting how farmers use their land (the intensive margin) are more relevant than those targeting how much land is farmed (the extensive margin). Columns (4)–(5) show that higher CAP subsidy intensity is associated with *lower* agricultural and natural shares ($\hat{\beta} = -0.17$ and $-0.10$), reflecting the historical redistribution of CAP payments across member states rather than a causal ecological channel.

### 3.3.4 Implications for calibration and model motivation.

The regression evidence establishes three empirical regularities that discipline the structural model.

First, the FBI decline of approximately 1.5% per year is persistent, precisely estimated, and unbroken by any CAP reform over three decades. This justifies modelling the exogenous biodiversity drift $\mu_b$ in (2) as a constant negative parameter reflecting secular forces—landscape simplification, climate change, chemical pressure—that existing policy instruments have failed to address. We calibrate $\mu_b$ from the FBI trend slope, scaled to the SSP3×RCP6.0 GLOBIO 4 scenario for forward-looking projections.

Second, the cross-country evidence shows no positive association between per-hectare CAP spending and biodiversity outcomes, whether measured in levels, changes, or after absorbing country and time effects. This implies that the ecological effectiveness of conventional direct payments is close to zero: we calibrate $\kappa_{\text{DP}} = 0.03$ in (15) and conduct sensitivity analysis over $\kappa_{\text{DP}} \in [0.01, 0.10]$.

Third, the biodiversity decline operates through intensification within existing agricultural land rather than through land conversion (Table 4). This identifies the intensive margin of farm-level management practices—precisely the margin targeted by the biodiversity conditionality $\kappa_B$ in the BF instrument—as the relevant channel for policy intervention.

Together, these findings pose a sharp policy question: if three decades of agricultural

Table 4: Cross-sectional validation: FBI, land use, and CAP spending

| | Dep. var.: $\ln(\text{FBI}_{it})$ | | | Dep. var.: land-use share | |
|---|---|---|---|---|---|
| | (1) | (2) | (3) | (4) ln(ag) | (5) ln(nat) |
| $t$ (trend) | $-0.0159^{***}$ | $-0.0158^{***}$ | $-0.0159^{***}$ | $+0.0034$ | $+0.0021$ |
| | (0.0010) | (0.0010) | (0.0010) | (0.0090) | (0.0078) |
| ln(ag share) | $-0.0526^{***}$ | | $-0.0128$ | | |
| | (0.0080) | | (0.0241) | | |
| ln(nat share) | | $+0.0800^{***}$ | $+0.0632^{*}$ | | |
| | | (0.0128) | (0.0374) | | |
| $\ln(\lambda_{\text{CAP}})$ | | | | $-0.1655^{***}$ | $-0.1017^{*}$ |
| | | | | (0.0432) | (0.0550) |
| $R^2$ | 0.456 | 0.460 | 0.460 | 0.032 | 0.016 |
| Observations | 436 | 436 | 436 | 300 | 300 |

*Notes:* Columns (1)–(3): pooled OLS of $\ln(\text{FBI}_{it})$ on land-use composition; columns (4)–(5): pooled OLS of land-use shares on CAP subsidy intensity $\lambda_{\text{CAP}}$ (EUR/ha). FBI data matched to nearest GLOBIO 4 land-use year (1992–2015). Robust (HC1) standard errors. No country fixed effects: within-country land-use variation is only 14% of between-country variation (6 observations per country over 23 years), making FE identification unreliable. $^{***}p < 0.01$, $^{**}p < 0.05$, $^{*}p < 0.1$.

subsidies totalling over EUR 1.5 trillion have produced no detectable improvement in farmland biodiversity, what alternative instrument design could succeed? The remainder of the paper proposes and evaluates one such design—a credit-based instrument that conditions financial incentives directly on biodiversity outcomes, rather than relying on unconditional income transfers.

# 4 Numerical methodology

The equilibrium is computed in four steps. First, the stopping problem for $V(s, b)$ is solved on the log-transformed grid $y = \ln s$ to obtain the free boundary $y^*(b)$. Second, the stationary Fokker–Planck equation (19) is solved on the domain $\Omega$ with absorbing, reflecting, and normalisation conditions to obtain $f(s, b)$. Third, aggregate output, biodiversity statistics, and policy costs are computed by integrating against $f$. Fourth, the planner optimisation (25) searches over $(\sigma_0, \phi, \kappa_B)$ using the equilibrium mapping $\pi \mapsto (y^*_\pi, f_\pi)$ under the DP-budget constraint and anti-arbitrage penalty, with $\gamma$ fixed.

The log-transform $y = \ln s$ converts the GBM diffusion to constant coefficients, improving numerical stability. The HJB variational inequality (17) is solved by policy iteration on a finite-difference grid: at each iteration, the optimal controls $(k^*, x^*)$ are updated from the current value function, and the resulting linear PDE is solved by sparse LU factorisation. Convergence is declared when the sup-norm change in $V$ falls below $10^{-6}$. The Fokker–Planck equation is discretised using upwind schemes for advection terms and central differences for diffusion, yielding a sparse linear system solved by direct factorisation. The resulting density $f$ is renormalised to unit mass. The planner's grid search over $(\sigma_0, \phi, \kappa_B)$ is refined by Powell's method starting from the grid optimum. The baseline results use a $100 \times 50$ grid ($N_y \times N_b$); we verify that doubling the resolution to $200 \times 100$ changes welfare and biodiversity moments by less than 1%.

# 5 Results

This section presents the quantitative findings. Section 5.1 compares blended finance with direct payments under two interest-rate regimes. Section 5.2 quantifies the welfare cost of the anti-arbitrage constraint. Section 5.3 evaluates a budget-neutral hybrid instrument that combines both policy channels. Section 5.4 examines regional heterogeneity across EU member states and NUTS2 regions.

## 5.1 DP and BF at equilibrium

Table 5 compares stationary equilibrium outcomes across three regimes—No Policy (NP), Direct Payments (DP), and optimised Blended Finance (BF*)—under two interest-rate environments: the pre-crisis era (2004–2011, $r_f = 3.07\%$) and the low-rate era (2015–2020, $r_f = 0.50\%$). The private credit schedule $\delta(s)$ is re-estimated for each period from eurozone FADN country averages, and the planner solves (25) subject to the inequality budget constraint $C_{\mathrm{BF}} \le C_{\mathrm{DP}}$.

Table 5: Equilibrium outcomes under No Policy, Direct Payments, and optimal Blended Finance across two interest-rate regimes.

| | **Pre-crisis (2004–2011)** | | | **Low-rate era (2015–2020)** | | |
|---|---|---|---|---|---|---|
| | NP | DP | BF* | NP | DP | BF* |
| *Panel A: Welfare* | | | | | | |
| $W$ | 42,790 | 41,823 | **47,495** | 47,585 | 47,734 | **56,546** |
| $\Delta W$ vs. DP (%) | +2.3 | — | +13.6 | −0.3 | — | +18.5 |
| *Panel B: Biodiversity* | | | | | | |
| $\mathbb{E}[b]$ | 0.397 | 0.402 | **0.482** | 0.387 | 0.400 | **0.510** |
| $\Delta\mathbb{E}[b]$ vs. DP (%) | −1.1 | — | +20.1 | −3.2 | — | +27.6 |
| Var($b$) | 0.035 | 0.034 | 0.039 | 0.036 | 0.035 | 0.045 |
| Net drift $\mathbb{E}[\mu_b + a]$ | −0.011 | −0.009 | −**0.002** | −0.011 | −0.009 | +**0.002** |
| $\mathbb{E}[a]$ (effort) | 0.000 | 0.002 | 0.009 | 0.000 | 0.002 | 0.013 |
| *Panel C: Output and farm structure* | | | | | | |
| $\mathbb{E}[q]$ (EUR) | 39,435 | 39,184 | 43,506 | 44,133 | 45,110 | **55,937** |
| Gini($q$) | 0.346 | 0.365 | 0.342 | 0.287 | 0.295 | 0.280 |
| *Panel D: Fiscal cost* | | | | | | |
| $C$ (EUR/farm) | 0 | 4,021 | **434** | 0 | 4,003 | **629** |
| $C_{\mathrm{BF}}/C_{\mathrm{DP}}$ (%) | — | — | 11 | — | — | 16 |
| $C_{\mathrm{prog}}$ (EUR) | 0 | 0 | 513 | 0 | 0 | 4,081 |
| Bio leverage $\mathbb{E}[a]/C$ | — | $4.2\times10^{-7}$ | $2.0\times10^{-5}$ | — | $4.2\times10^{-7}$ | $2.1\times10^{-5}$ |
| *Panel E: Optimal BF design* | | | | | | |
| $\sigma_0^*$ | — | — | 0.242 | — | — | 0.406 |
| $\phi^*$ | — | — | 0.001 | — | — | 0.008 |
| $\kappa_B^*$ | — | — | 0.032 | — | — | 0.027 |
| Participation (%) | — | — | 49 | — | — | 63 |

*Note:* $100 \times 50$ grid. NP sets $\lambda = 0$, $\kappa_{\mathrm{DP}} = 0$. BF* denotes the planner-optimised policy with the hard anti-arbitrage constraint $\delta_{\mathrm{BF,min}} \ge r_f$. Pre-crisis credit parameters: $r_f = 3.07\%$, $\rho_{ag} = 0.011$, $c_\delta = 1.84$, $\psi = 2.0$ (NLS on eurozone FADN). Low-rate era: $r_f = 0.50\%$ with baseline calibration (Table 1). In both periods the BF fiscal cost lies well below the DP envelope.

The central finding is that BF$^*$ welfare-dominates DP in both interest-rate environments. In the low-rate era, BF$^*$ raises welfare by 18.5% relative to DP, increases expected biodiversity by 28% ($\mathbb{E}[b] = 0.510$ versus 0.400), and reverses the net biodiversity drift from $-0.009$ under DP to $+0.002$ —the only regime in which expected biodiversity is non-declining. These gains are achieved at 84% lower fiscal cost ($C_{\text{BF}} = 629$ versus $C_{\text{DP}} = 4{,}003$ EUR per farm). In the pre-crisis era, BF$^*$ raises $\mathbb{E}[b]$ by 20% under tighter credit conditions, with an even lower fiscal cost ratio of 11%.

By contrast, DP is largely ineffective. Welfare under DP is essentially unchanged relative to NP ($\Delta W = -0.3\%$ in the low-rate era), and the net biodiversity drift remains negative ($-0.009$/yr) despite annual spending of 4,003 EUR per farm. This is consistent with the FBI panel analysis, which detects no structural break in farmland bird decline at the 2013 greening reform ($\kappa_{\text{DP}} = 0.03$).

BF$^*$ also raises expected farm output by 24% in the low-rate era ($\mathbb{E}[q] = 55{,}937$ versus 45,110 EUR) while reducing the output Gini coefficient from 0.295 to 0.280. The credit subsidy $\sigma_0\delta(s)$ delivers a larger absolute rate reduction to small farms ($\partial\Delta/\partial s < 0$), thereby functioning as a progressive instrument. By lowering the exit threshold $y^*(b)$, BF extends the survival of small farms and allows the positive size drift $\mu_s > 0$ to build their capital stock gradually.

The biodiversity effort elicited per euro of fiscal cost —the bio leverage ratio $\mathbb{E}[a]/C$— is $2.1 \times 10^{-5}$ under BF$^*$ versus $4.2 \times 10^{-7}$ under DP, a factor of approximately 50. The pre-crisis ratio is comparable (48:1). This efficiency gap arises because the government subsidises only the *spread* between private and subsidised rates, not the full capital cost; the farmer's co-investment through the unsubsidised portion of capital leverages each euro of public spending.

Finally, the optimal BF$^*$ design satisfies the budget constraint with substantial slack: the BF fiscal cost lies well below the DP envelope. The force that limits the subsidy share $\sigma_0$ is the convex programme cost $c_{\text{prog}}\sigma_0^{\gamma_{\text{prog}}}$, not the budget constraint itself.

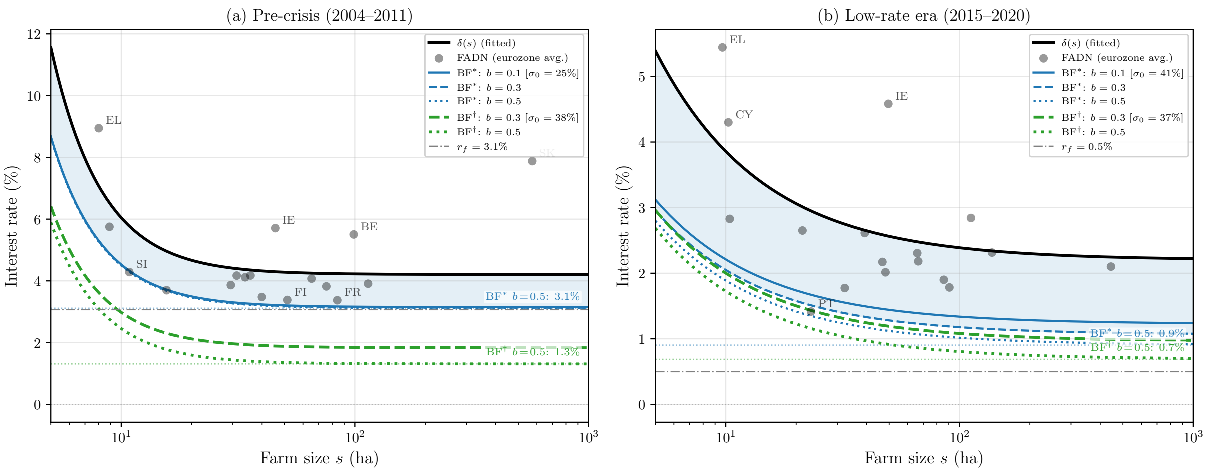


Figure 1: Private credit schedule $\delta(s)$ and subsidised BF rate $\delta_{\text{BF}}(s, b)$ under both interest-rate regimes. Left: pre-crisis (2004–2011); right: low-rate era (2015–2020). Each panel shows the unsubsidised schedule (solid black), the BF$^*$ rate at $b = 0.5$ (blue), and the BF$^\dagger$ rate at $b = 0.5$ (red dashed). FADN country-average borrowing rates (circles) are overlaid for comparison.

The welfare dominance of BF over DP rests on three reinforcing channels.

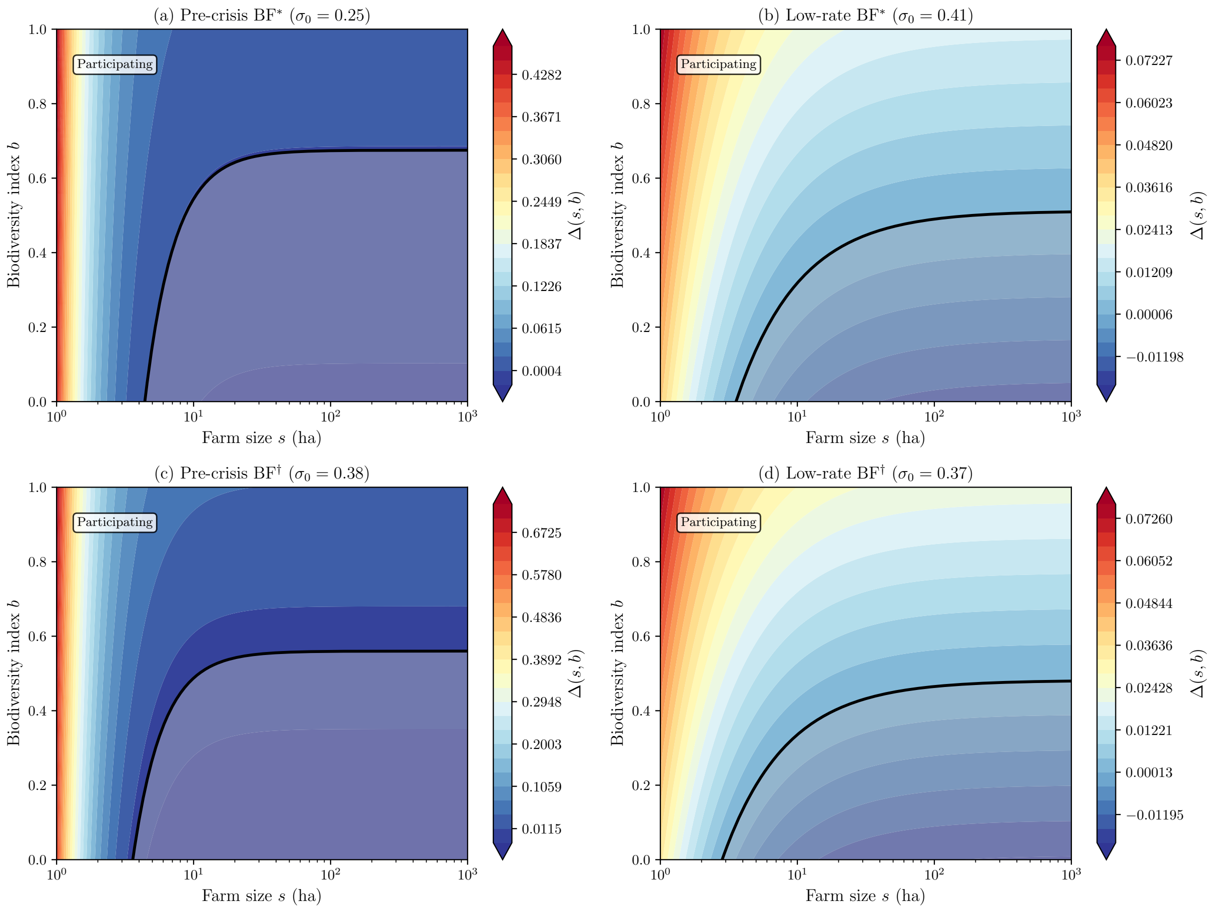


Figure 2: Net gain from BF participation $\Delta(s, b)$ and participation frontier. Top row: BF* (constrained); bottom row: BF† (unconstrained). Left: pre-crisis; right: low-rate era. The solid black contour marks $\Delta = 0$: farms above the line participate, those below do not. The progressive structure ($\partial\Delta/\partial s < 0$) is visible: small farms gain more from the subsidy than large farms at any biodiversity level.

The first is the *credit channel*. DP operates through a lump-sum transfer $\lambda s$ that enters the farmer's revenue without conditioning on biodiversity outcomes. The biodiversity effort induced per euro of transfer is governed by $\kappa_{\text{DP}} = 0.03$, which the FBI panel analysis reveals to be near zero. BF, by contrast, conditions the cost of capital on biodiversity: the subsidised rate $\delta_{\text{BF}}(s, b) = \delta(s)(1 - \sigma_0) - \phi b$ generates a continuous incentive gradient $\partial\delta_{\text{BF}}/\partial b = -\phi < 0$, so that improving biodiversity lowers borrowing costs at the margin. Combined with the conditionality parameter $\kappa_B$, this produces farm-level effort approximately seven times higher than under DP ($\mathbb{E}[a] = 0.013$ versus 0.002 in the low-rate era). The credit channel thus operates as a price instrument targeting the intensive margin of biodiversity investment, whereas DP functions as an unconditional income transfer with negligible behavioural response. Figure 1 illustrates this mechanism. The private schedule $\delta(s)$ is steeply convex in farm size, declining from roughly 12% for the smallest holdings to 4% for the largest—a gradient that reflects the well-documented size-dependent risk premium in agricultural lending. The BF$^*$ subsidised rates (blue band) compress this spread from above: at median biodiversity ($b = 0.5$), the rate floors at 3.1% in the pre-crisis era and 0.9% in the low-rate era, bounded below by the respective $r_f$. Two features are economically significant. First, the absolute rate reduction is largest for small farms ($\approx 8$ pp at $s = 5$ ha versus $\approx 1$ pp at $s = 500$ ha), so the uniform proportional subsidy $\sigma_0$ operates as a *de facto* progressive transfer through the concavity of the credit schedule. Second, the vertical gap between BF$^*$ and BF$^\dagger$ is substantially wider in the pre-crisis era, where the higher $r_f$ leaves room for a large biodiversity premium $\varphi b$ below the anti-arbitrage floor; in the low-rate era the two schedules nearly converge, foreshadowing the modest welfare gap documented in Section 5.2.

The second is the *progressive structure* of the subsidy. The private credit schedule $\delta(s) = r_f + \rho_{ag} + c_\delta s^{-\psi}$ is steeply decreasing in farm size: small farms face rates three to five times higher than large farms. The proportional subsidy $\sigma_0\delta(s)$ therefore delivers a larger absolute rate reduction to small farms, relaxing the credit constraint precisely where it is most binding. This lowers the exit threshold $y^*(b)$, extending the survival of small farms and allowing the positive size drift ($\mu_s > 0$) to increase their capital stock gradually. The lower Gini coefficient under BF$^*$ (0.280 versus 0.295 in the low-rate era) reflects this extensive-margin effect. Figure 3 provides direct evidence of the capital mobilisation induced by BF. The left column displays the optimal subsidised capital share $x^*(s, b)$: small farms channel nearly all investment through the subsidised instrument ($x^* \to 1$ as $s \to 0$), because the proportional rate reduction is largest where the private rate is highest; large farms allocate only 30–50% to the subsidised channel, funding the remainder at market rates. The right column reports the ratio of total capital under BF to its no-policy level, $k^*_{\text{BF}}/k^*_{\text{NP}}$. This ratio exceeds unity throughout the participating region, reaching 1.5–2.75× for the smallest farms. The credit subsidy therefore does not merely redirect existing investment from private to subsidised channels; it mobilises *additional* capital formation, particularly among credit-constrained small farms that would otherwise under-invest relative to the social optimum. This crowding-in of private investment is the micro-level source of the high bio leverage ratio reported in Table 5.

The third is *fiscal efficiency*. BF$^*$ achieves higher biodiversity and output while spending 84% less public money than DP ($C_{\text{BF}} = 629$ versus $C_{\text{DP}} = 4{,}003$ EUR/farm in the low-rate era). The government subsidises only the spread between private and subsidised rates ($\sigma_0\delta(s) \cdot x^* \cdot k^*$), not the full capital cost. The farmer's own co-investment—through $(1 - x^*)k^*$ at the private rate and $x^*k^*$ at the reduced rate—leverages each euro of public spending. The bio leverage ratio of approximately 50:1 quantifies this multiplier: the credit channel mobilises private investment in biodiversity that the transfer channel leaves untapped.

These three channels interact with the participation decision. In the low-rate era, 63% of the farm population participates in BF$^*$. Non-participants are predominantly large farms with low biodiversity for which the conditionality cost $\kappa_B(1-b)$ exceeds the credit benefit $\sigma_0\delta(s)+\phi b$. This self-selection is welfare-improving: farms that opt out are precisely those where the subsidy would generate the least biodiversity gain per euro of expenditure. Figure 2 maps the participation surplus $\Delta(s,b)$ across the full state space. The frontier (black contour at $\Delta=0$) slopes downward: small farms participate at virtually any biodiversity level, because their high private borrowing cost makes even a modest proportional subsidy highly valuable; large farms participate only above a biodiversity threshold ($b \geq 0.5$ under BF$^*$ in the low-rate era), where the conditionality reward $\kappa_B b$ outweighs the foregone flexibility of private capital. The colour gradient confirms that the participation surplus is monotonically decreasing in $s$—the distributional signature of a progressive instrument. Under BF$^\dagger$ in the pre-crisis era (panel c), the relaxed constraint pushes $\Delta > 0$ over nearly the entire state space, consistent with the higher participation rate (56% versus 49% under BF$^*$). The self-targeting property has an important practical implication: the government need not observe farm-level credit constraints or biodiversity stocks to direct spending toward the most responsive farms—voluntary participation achieves this allocation endogenously.

The aggregate distributional consequences of these farm-level responses are visible in Figures 4 and 5. Under NP and DP, the stationary joint density $f(s,b)$ concentrates at low biodiversity ($b \approx 0.3$–$0.4$), and the two densities are visually indistinguishable—a striking confirmation that existing direct payments produce no discernible shift in the cross-sectional distribution of farmland biodiversity. Under BF$^*$, the density mass shifts upward to $b \approx 0.5$–$0.6$, and under BF$^\dagger$ in the pre-crisis era the mode reaches $b \approx 0.7$. The marginal density $\bar{f}(b)$ (Figure 5) makes the welfare ranking transparent: the BF distribution lies uniformly to the right of the DP distribution, with a rightward shift of approximately 0.11 units in expected biodiversity. In the low-rate era, the BF$^*$ and BF$^\dagger$ stationary distributions are nearly indistinguishable—both in their joint densities and in their marginal biodiversity profiles ($\mathbb{E}[b]=0.510$ versus 0.515)—confirming that the anti-arbitrage constraint imposes a negligible cost when $r_f$ is already close to zero and the subsidy space between the two constraint floors is narrow. In the pre-crisis era, by contrast, the wider gap between $r_f$ and zero gives the unconstrained planner room to push borrowing costs further below the risk-free rate, generating a visibly broader and more rightward-shifted distribution under BF$^\dagger$ ($\mathbb{E}[b]=0.558$ versus 0.485). This period-dependent divergence underscores that the economic value of relaxing the anti-arbitrage constraint is itself a function of the monetary policy environment.

## 5.2 The cost of anti-arbitrage discipline

A central design feature of BF$^*$ is the hard anti-arbitrage constraint $\delta_{\mathrm{BF,min}} \geq r_f$, which prevents the subsidised rate from falling below the risk-free rate. While this constraint eliminates carry-trade incentives and simplifies prudential supervision, it restricts the planner's ability to reward high-biodiversity farms with deeply discounted credit. To quantify the welfare cost of this restriction, we compare BF$^*$ with a relaxed variant BF$^\dagger$ that replaces the hard constraint with a soft convex penalty on rate violations, retaining only a non-negativity floor $\delta_{\mathrm{BF}}(s,b) \geq 0$ and the budget constraint $C_{\mathrm{BF}} \leq C_{\mathrm{DP}}$.

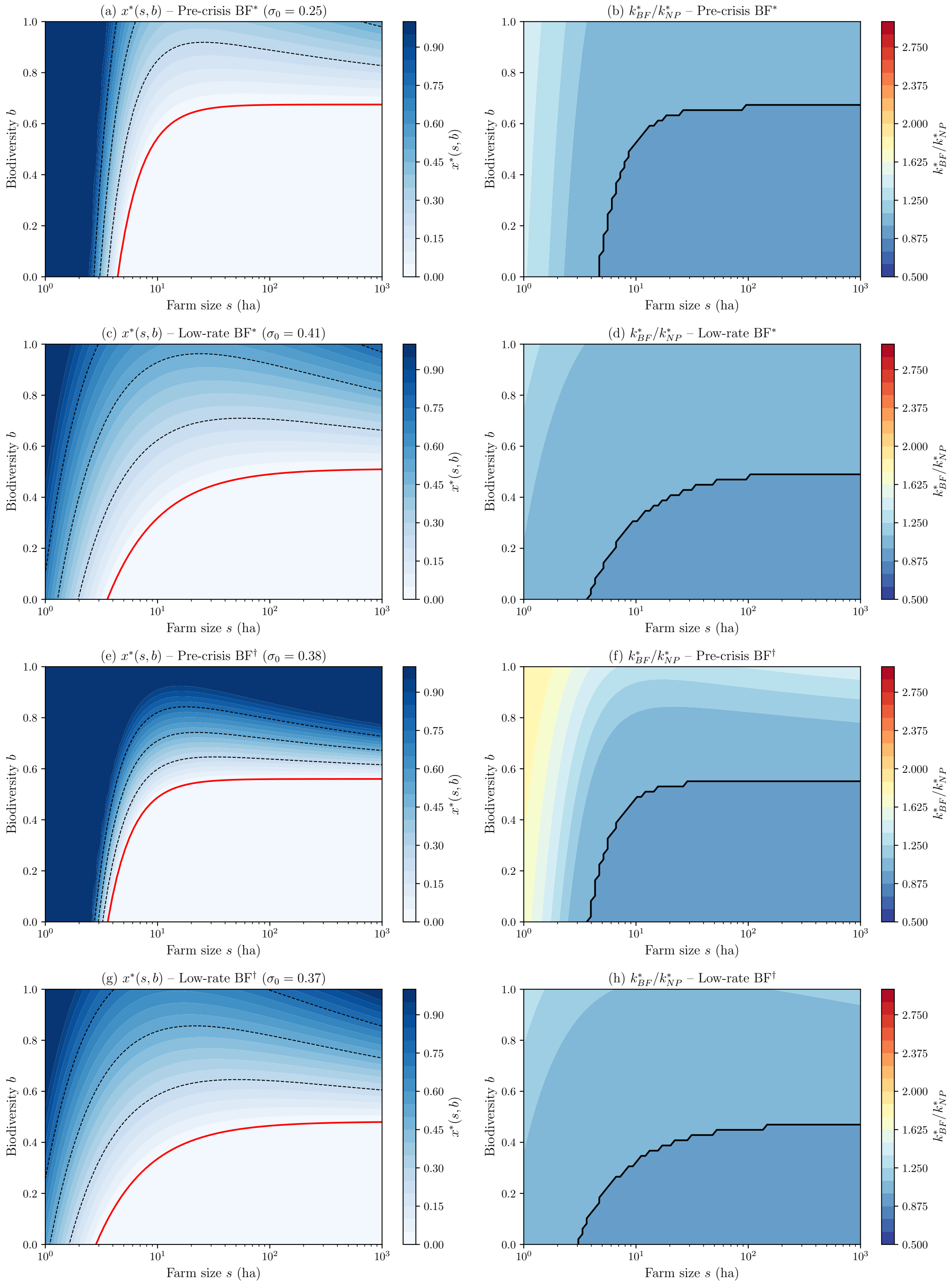


Figure 3: Optimal farm-level controls in the $(s, b)$ plane. Each row corresponds to one regime–period combination (BF* pre-crisis, BF* low-rate, BF† pre-crisis, BF† low-rate). Left: subsidised capital share $x^*(s, b)$; right: capital-to-size ratio $k^*/s$. White curve: exit threshold $y^*(b)$.

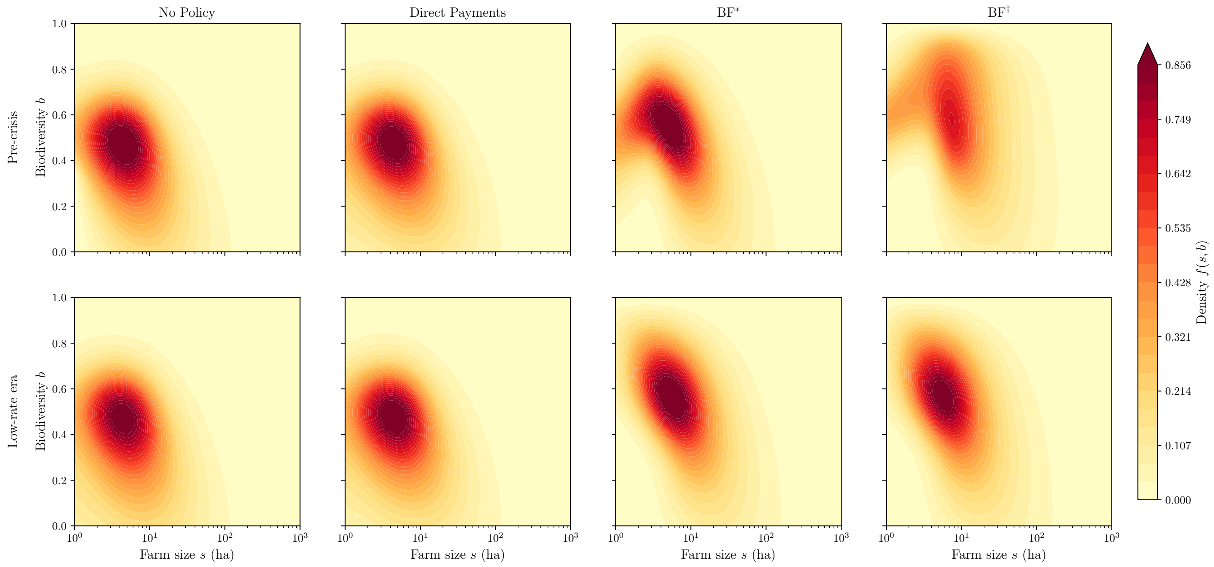


Figure 4: Stationary joint density $f(s, b)$ from the Fokker–Planck equation. Top row: pre-crisis; bottom row: low-rate era. Columns: NP, DP, BF*, BF†. Under NP and DP the mass concentrates at low $b$ (biodiversity declining). Under BF the distribution shifts rightward toward higher $b$, with BF† showing the strongest shift in the pre-crisis period where the subsidy space is largest.

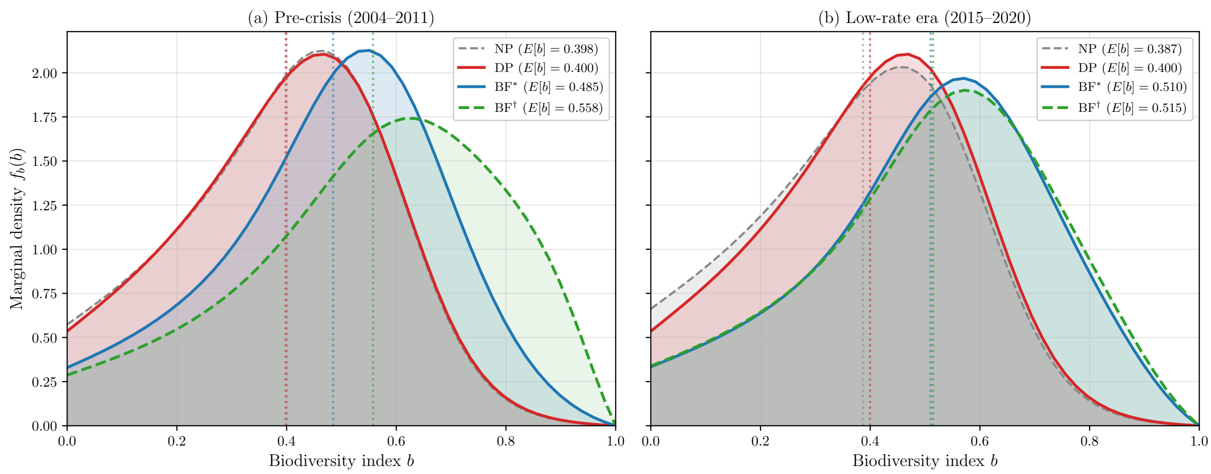


Figure 5: Marginal biodiversity distribution $\bar{f}(b) = \int f(s, b)\,\mathrm{d}s$ under each policy regime. Left: pre-crisis; right: low-rate era. Under NP and DP the distribution concentrates at low $b$; BF* shifts the mode substantially to the right. BF† achieves a further rightward shift in the pre-crisis era (where high $r_f$ leaves room for a large biodiversity premium) but essentially coincides with BF* in the low-rate era (where the $\delta_{\mathrm{BF}} \geq 0$ floor binds tightly).

Table 6: Cost of anti-arbitrage discipline: BF$^*$ (hard constraint) vs. BF$^\dagger$ (soft penalty).

| | **Pre-crisis (2004–2011)** | | **Low-rate era (2015–2020)** | |
|---|---|---|---|---|
| | BF$^*$ | BF$^\dagger$ | BF$^*$ | BF$^\dagger$ |
| *Panel A: Welfare* | | | | |
| $W$ | 47,495 | **48,097** | 56,546 | **58,412** |
| $\Delta W$ vs. DP (%) | +13.6 | **+15.0** | +18.5 | +**22.4** |
| *Panel B: Biodiversity* | | | | |
| $\mathbb{E}[b]$ | 0.482 | **0.499** | 0.510 | **0.516** |
| Net drift $\mathbb{E}[\mu_b + a]$ | −0.002 | **0.000** | +0.002 | +**0.004** |
| $\mathbb{E}[a]$ (effort) | 0.009 | 0.010 | 0.013 | 0.014 |
| *Panel C: Output* | | | | |
| $\mathbb{E}[q]$ (EUR) | 43,311 | **44,614** | 55,937 | **57,680** |
| *Panel D: Fiscal cost* | | | | |
| $C$ (EUR/farm) | 434 | 658 | 629 | 772 |
| $C_{\text{BF}}/C_{\text{DP}}$ (%) | 11 | 16 | 16 | 19 |
| $C_{\text{prog}} + C_{\text{rate}}$ (EUR) | 513 | 1,120 | 4,081 | 3,967 |
| *Panel E: Optimal design* | | | | |
| $\sigma_0^*$ | 0.242 | 0.293 | 0.406 | 0.385 |
| $\varphi^*$ | 0.0008 | 0.0001 | 0.0080 | 0.0134 |
| $\kappa_B^*$ | 0.032 | 0.031 | 0.027 | 0.029 |
| Participation (%) | 49 | 56 | 63 | 63 |
| $\delta_{\text{BF,min}} - r_f$ (pp) | 0.0 | −0.1 | 0.0 | −0.5 |

*Note:* BF$^*$ imposes the hard anti-arbitrage constraint $\delta_{\text{BF,min}} \geq r_f$ ($c_{\text{rate}} = 0$). BF$^\dagger$ replaces this with a soft penalty and retains only the floor $\delta_{\text{BF}} \geq 0$. $\delta_{\text{BF,min}} - r_f$ is the spread of the minimum subsidised rate over the risk-free rate; negative values indicate the subsidised rate dips below $r_f$. $100 \times 50$ grid.

Table 6 shows that the welfare cost of anti-arbitrage discipline is moderate in both interest-rate environments. In the pre-crisis period, relaxing the hard constraint raises the welfare gain from +13.6% to +15.0% relative to DP. The planner responds by increasing $\sigma_0$ from 0.24 to 0.29, pushing the minimum subsidised rate 0.1 percentage points below $r_f$ and raising $\mathbb{E}[b]$ from 0.482 to 0.499. In the low-rate era, BF$^\dagger$ delivers a +22.4% welfare gain versus +18.5% for BF$^*$; the biodiversity premium $\varphi$ rises from 0.008 to 0.013, and $\mathbb{E}[b]$ increases from 0.510 to 0.516. In both cases the non-negativity floor $\delta_{\text{BF}} \geq 0$ binds under BF$^\dagger$, confirming that the planner would prefer negative subsidised rates if permitted—an economically unappealing outcome that validates the anti-arbitrage rationale.

The key insight is that even under the conservative hard constraint, BF$^*$ delivers welfare gains of 14–19% over DP. The additional 1–4 percentage points available under BF$^\dagger$ represent an upper bound on what could be gained by relaxing prudential safeguards—a modest premium relative to the regulatory and reputational risks of allowing subsidised rates to fall below the risk-free benchmark. Figure 6 visualises the planner's problem by plotting welfare $W(\sigma_0, \varphi)$ at the optimal $\kappa_B$. The surface is smooth and strictly concave, yielding a well-defined interior optimum in all four regime–period combinations—the planner faces no multiplicity of local optima. The BF$^*$ optimum (white star) sits on or immediately adjacent to the anti-arbitrage boundary (red dashed), confirming that this constraint binds: absent the prudential floor, the planner would choose a larger subsidy or biodiversity premium. By contrast, the budget constraint $C_{\text{BF}} = C_{\text{DP}}$ (orange dashed) lies well above the optimum in all panels—the fiscal envelope is never binding, and the force that limits the policy is the anti-arbitrage condition together with the convex programme cost, not the available budget. This has a direct policy implication: the government can implement BF with substantial fiscal headroom, leaving resources available for complementary instruments (Section 5.3). The welfare-improving region ($W > W_{\text{DP}}$, above the white contour) spans a wide range of $(\sigma_0, \varphi)$ combinations, suggesting that even a moderately miscalibrated BF policy would still welfare-dominate DP—a practical robustness property for instrument design under parameter uncertainty. Finally, in the low-rate BF$^\dagger$ panel (d), the anti-arbitrage and non-negativity boundaries nearly coincide, providing a geometric explanation for why BF$^* \approx$ BF$^\dagger$ when $r_f$ is low: there is simply no room between the two constraint surfaces for the planner to exploit.

## 5.3 Fiscal savings reallocation

Because BF$^*$ achieves its biodiversity and output gains at a fraction of the DP fiscal cost ($C_{\text{BF}} \approx 630$ versus $C_{\text{DP}} \approx 4{,}000$ EUR/farm in the low-rate era), a natural policy question is whether the fiscal savings could fund additional direct payments, combining BF credit incentives with supplementary DP spending.

We evaluate a budget-neutral hybrid instrument, denoted BFDP, in which the BF$^*$ optimal parameters $(\sigma_0^*, \varphi^*, \kappa_B^*)$ are maintained and an additional per-hectare payment $\lambda$ is calibrated so that total fiscal cost equals the DP baseline: $C_{\text{BFDP}} = C_{\text{DP}}$. This represents a pure reallocation of existing DP budgets, with no additional public spending. We vary the direct-payment biodiversity intensity over $\kappa_{\text{DP}} \in \{0.01, 0.03, 0.05, 0.10\}$ to span the plausible range of ecological effectiveness of generic DP transfers.

Table 7 yields three findings. At the baseline DP effectiveness ($\kappa_{\text{DP}} = 0.03$), the budget-neutral BFDP achieves welfare gains comparable to pure BF$^*$ (+17.3% versus +18.4% in the low-rate era) while raising expected biodiversity by 29% relative to DP. The fiscal savings released by the credit channel thus fund meaningful complementary

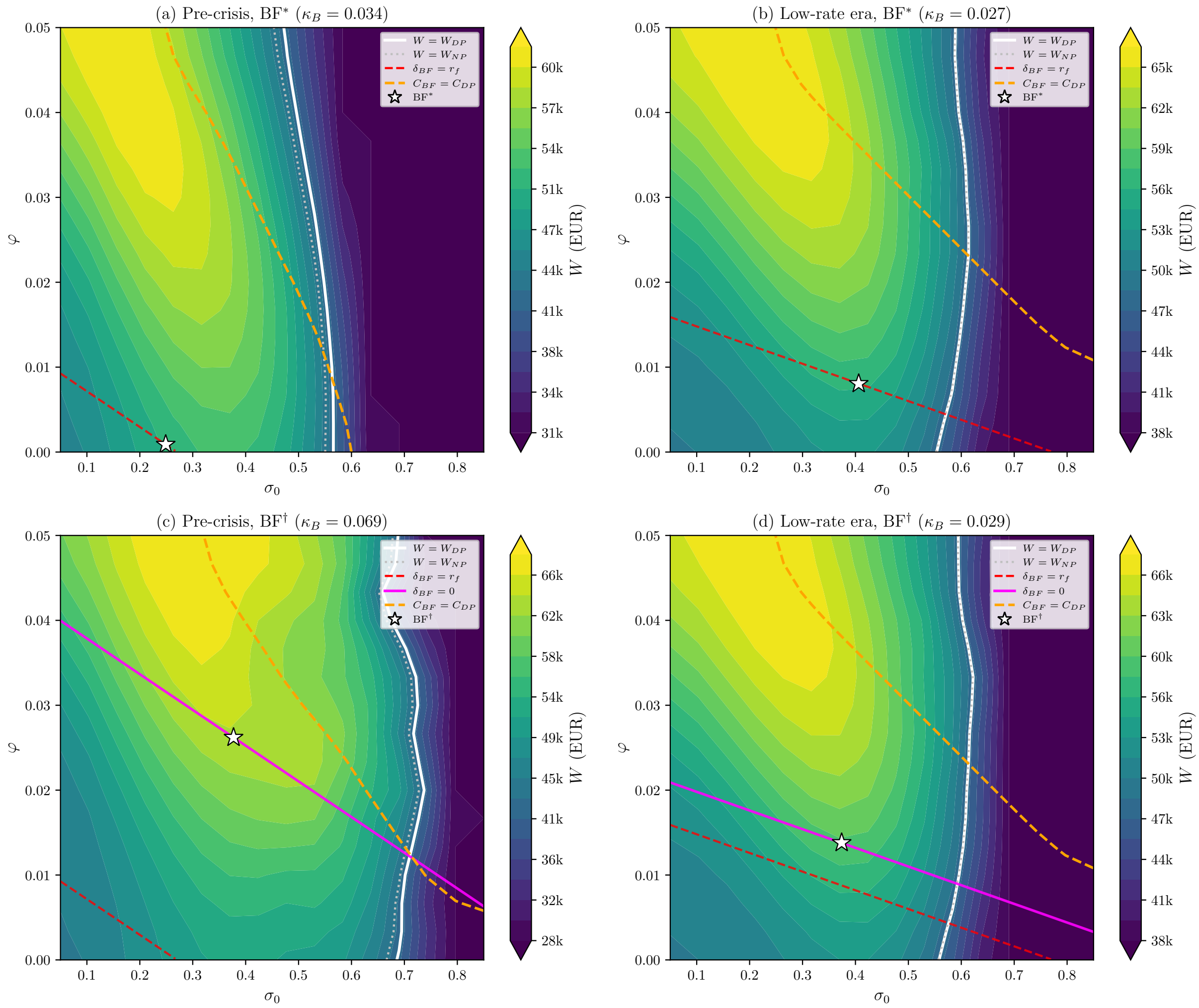


Figure 6: Welfare landscape $W(\sigma_0, \varphi)$ at the optimal $\kappa_B$. Top row: BF* (hard anti-arbitrage: $\delta_{\text{BF,min}} \geq r_f$); bottom row: BF† (soft penalty, $\delta_{\text{BF}} \geq 0$ floor). Left: pre-crisis; right: low-rate era. White contour: $W = W_{\text{DP}}$; grey dotted: $W = W_{\text{NP}}$; red dashed: anti-arbitrage boundary $\delta_{\text{BF}} = r_f$; magenta solid (BF† panels): non-negativity floor $\delta_{\text{BF}} = 0$; orange dashed: budget constraint $C_{\text{BF}} = C_{\text{DP}}$; white star: optimum. In the low-rate era (d), the two constraint boundaries nearly coincide, explaining why BF† ≈ BF*. In the pre-crisis era (c), the wider gap permits a substantial biodiversity premium.

Table 7: Fiscal reallocation: budget-neutral BF+DP versus pure DP. BFDP maintains $\text{BF}^*$ parameters and adds DP payments such that $C_{\text{BFDP}} = C_{\text{DP}}$. $\Delta W$ and $\Delta\mathbb{E}[b]$ are reported relative to pure DP.

| | $W$ | $\Delta W$ | $\mathbb{E}[b]$ | $\Delta b$ | $\mathbb{E}[a]$ | $C$ |
|---|---|---|---|---|---|---|
| *Low-rate era (2015–2020)* | | | | | | |
| DP | 47,734 | — | 0.400 | — | 0.002 | 4,003 |
| $\text{BF}^*$ | 56,525 | +18.4% | 0.510 | +27.5% | 0.013 | 625 |
| BFDP ($\kappa_{\text{DP}}=0.01$) | 55,596 | +16.5% | 0.512 | +28.0% | 0.013 | 3,999 |
| BFDP ($\kappa_{\text{DP}}=0.03$) | 55,986 | +17.3% | 0.516 | +29.1% | 0.014 | 4,002 |
| BFDP ($\kappa_{\text{DP}}=0.05$) | 56,329 | +18.0% | 0.521 | +30.3% | 0.014 | 4,007 |
| BFDP ($\kappa_{\text{DP}}=0.10$) | 56,986 | +19.4% | 0.532 | +33.2% | 0.016 | 4,000 |
| *Pre-crisis (2004–2011)* | | | | | | |
| DP | 41,624 | — | 0.400 | — | 0.002 | 4,003 |
| $\text{BF}^*$ | 47,283 | +13.6% | 0.481 | +20.4% | 0.009 | 438 |
| BFDP ($\kappa_{\text{DP}}=0.01$) | 46,150 | +10.9% | 0.480 | +20.2% | 0.009 | 4,002 |
| BFDP ($\kappa_{\text{DP}}=0.03$) | 46,752 | +12.3% | 0.488 | +22.1% | 0.009 | 4,003 |
| BFDP ($\kappa_{\text{DP}}=0.05$) | 47,213 | +13.4% | 0.495 | +23.7% | 0.010 | 3,999 |
| BFDP ($\kappa_{\text{DP}}=0.10$) | 48,126 | +15.6% | 0.512 | +28.0% | 0.011 | 4,004 |

conservation spending without sacrificing the core welfare advantage.

Welfare and biodiversity gains are monotonically increasing in $\kappa_{\text{DP}}$. When direct payments are ecologically effective ($\kappa_{\text{DP}} = 0.10$), the combined instrument dominates both pure alternatives: $\Delta W = +19.4\%$ and $\Delta\mathbb{E}[b] = +33.2\%$ in the low-rate era. The two channels are complementary: BF provides incentives through the cost of capital, while DP supplements them with direct transfers conditional on biodiversity outcomes.

Even when DP spending is ecologically ineffective ($\kappa_{\text{DP}} = 0.01$), the BFDP hybrid never underperforms pure $\text{BF}^*$ by more than two percentage points of welfare, because the credit channel provides the bulk of the biodiversity improvement irrespective of the DP supplement. The pre-crisis period exhibits qualitatively similar patterns, with somewhat larger gains at high $\kappa_{\text{DP}}$ because the DP supplement is amplified by the improved biodiversity baseline that the BF channel generates.

## 5.4 Regional heterogeneity

The baseline results treat the EU as a representative region using continent-wide calibrated parameters. In practice, the biodiversity drift $\mu_b$ varies substantially across space: under the SSP3×RCP6.0 scenario, GLOBIO4 projections imply NUTS2-level annual drifts ranging from $-0.009$ (intensive arable regions in France and the Netherlands) to $+0.001$ (extensifying regions in Romania). This subsection examines whether blended finance retains its advantage under such heterogeneous ecological pressures.

### 5.4.1 Four representative cases

We select four NUTS2 regions at the 5th, 50th, 95th percentiles and maximum of the $\mu_b$ distribution across 232 EU27 regions. For each, we rescale the GLOBIO4 MSA drift to FBI units (scale factor $\approx 9.3$, preserving the spatial ranking) and re-solve the planner problem, holding all other parameters at their FBI-NUTS2 calibrated values.

Table 8: Blended finance effectiveness across representative NUTS2 regions (SSP3×RCP6.0).

| Case | NUTS2 | $\mu_b$ | $W_{\text{DP}}$ | $W_{\text{BF}^*}$ | $\Delta W^*$ | $\mathbb{E}[b]_{\text{DP}}$ | $\mathbb{E}[b]_{\text{BF}^*}$ | $\sigma_0^*$ | $\phi^*$ | $\kappa_B^*$ | $C_{\text{BF}}/C_{\text{DP}}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Severe loss | FRJ2 | −0.024 | 41,232 | 44,760 | +8.6% | 0.335 | 0.421 | 0.376 | 0.009 | 0.021 | 24% |
| Median loss | ES23 | −0.011 | 47,755 | 56,688 | +18.7% | 0.400 | 0.508 | 0.390 | 0.008 | 0.025 | 15% |
| Near-zero | PT18 | −0.002 | 53,664 | 64,424 | +20.1% | 0.520 | 0.585 | 0.366 | 0.009 | 0.032 | 10% |
| Positive | RO42 | +0.007 | 55,275 | 63,044 | +14.1% | 0.634 | 0.663 | 0.304 | 0.010 | 0.034 | 13% |

*Notes:* Four NUTS2 regions at the 5th, 50th, 95th percentiles and maximum of the $\mu_b$ distribution (232 EU27 regions). $\mu_b$ from GLOBIO4 MSA (2015–2050) rescaled to FBI units (×9.3); all other parameters at FBI-NUTS2 defaults (Table 1). BF*: $\delta_{\text{BF}} \geq r_f$, $C_{\text{BF}} \leq C_{\text{DP}}$.

Table 8 confirms that BF* dominates DP in all four cases, with welfare gains ranging from +8.6% under severe loss (FRJ2, $\mu_b = -0.024$) to +20.1% under near-zero drift (PT18, $\mu_b = -0.002$). Expected biodiversity improves uniformly. The pattern of gains varies with local pressure: under severe loss the planner deploys a higher subsidy ($\sigma_0^* = 0.38$) but achieves a smaller welfare gain, because the strong negative drift is harder to offset. Near-zero and positive-drift regions exhibit larger gains with lower subsidy rates, reflecting the complementarity between natural biodiversity recovery and policy-induced effort. The fiscal cost ratio $C_{\text{BF}}/C_{\text{DP}}$ ranges from 10% to 24%, confirming that BF remains substantially cheaper than DP across heterogeneous ecological conditions.

### 5.4.2 Country-level optimisation

We extend the analysis to all 27 EU member states. For each country, we compute the pixel-weighted average of NUTS2 regional drifts $\mu_b$, rescaled from GLOBIO4 MSA to FBI units (×9.3), and optimise the BF* policy $(\sigma_0^*, \varphi^*, \kappa_B^*)$ under the budget constraint $C_{\text{BF}} \leq C_{\text{DP}}$.

Table 9 reports the results. BF* raises expected biodiversity in all 27 countries, with $\Delta\mathbb{E}[b]$ ranging from +0.06 (France, $\mu_b = -0.027$; Romania, $\mu_b \approx 0$) to +0.11 (Austria, Czechia, Poland, $\mu_b \approx -0.012$). Two patterns are noteworthy. The biodiversity gain is largest at moderate decline rates ($|\mu_b| \approx 0.010$–$0.013$) and tapers at both extremes: under severe decline the negative drift is difficult to offset, while under near-zero decline the baseline already preserves biodiversity, leaving less room for improvement. The optimal policy parameters are remarkably stable across the moderate-decline group: $\sigma_0^* \approx 0.39$, $\varphi^* \approx 0.008$, $\kappa_B^* \approx 0.027$. This robustness suggests that a single EU-wide BF policy calibrated to the median country would capture most of the potential gains.

Figure 7 maps the per-NUTS2 biodiversity gain $\Delta\mathbb{E}[b]$ when each country applies its own optimised BF* policy. Improvements are widespread: 227 out of 232 regions experience higher expected biodiversity under BF* than under DP. The largest gains ($\Delta\mathbb{E}[b] > 0.08$) concentrate in Central and Northern Europe, where moderate biodiversity decline meets strong credit-channel responsiveness. The five French regions with the most severe projected decline ($\mu_b < -0.04$) are the only areas where the BF* biodiversity gain is near zero, a finding we investigate further below.

Table 9: Country-level BF* optimisation (SSP3×RCP6.0).

| Country | Code | $\mu_b$ | $\sigma_0^*$ | $\phi^*$ | $\kappa_B^*$ | $\mathbb{E}[b]_{\mathrm{DP}}$ | $\mathbb{E}[b]_{\mathrm{BF}^*}$ | $\Delta\mathbb{E}[b]$ |
|---|---|---|---|---|---|---|---|---|
| Austria | AT | −0.012 | 0.411 | 0.0079 | 0.026 | 0.387 | 0.499 | +0.113 |
| Belgium | BE | −0.019 | 0.406 | 0.0081 | 0.024 | 0.353 | 0.459 | +0.105 |
| Bulgaria | BG | −0.006 | 0.391 | 0.0084 | 0.030 | 0.461 | 0.551 | +0.090 |
| Croatia | HR | −0.003 | 0.364 | 0.0089 | 0.031 | 0.504 | 0.573 | +0.069 |
| Cyprus | CY | −0.008 | 0.388 | 0.0085 | 0.027 | 0.430 | 0.531 | +0.101 |
| Czechia | CZ | −0.011 | 0.404 | 0.0081 | 0.026 | 0.400 | 0.511 | +0.111 |
| Denmark | DK | −0.006 | 0.367 | 0.0089 | 0.027 | 0.452 | 0.544 | +0.092 |
| Estonia | EE | −0.013 | 0.393 | 0.0082 | 0.025 | 0.384 | 0.492 | +0.108 |
| Finland | FI | −0.007 | 0.394 | 0.0083 | 0.029 | 0.436 | 0.536 | +0.099 |
| France | FR | −0.027 | 0.307 | 0.0102 | 0.018 | 0.330 | 0.390 | +0.061 |
| Germany | DE | −0.010 | 0.400 | 0.0082 | 0.027 | 0.411 | 0.519 | +0.108 |
| Greece | EL | −0.016 | 0.399 | 0.0082 | 0.025 | 0.365 | 0.473 | +0.108 |
| Hungary | HU | −0.004 | 0.386 | 0.0084 | 0.031 | 0.477 | 0.560 | +0.083 |
| Ireland | IE | −0.014 | 0.369 | 0.0089 | 0.024 | 0.376 | 0.478 | +0.102 |
| Italy | IT | −0.010 | 0.395 | 0.0082 | 0.026 | 0.409 | 0.516 | +0.107 |
| Latvia | LV | −0.006 | 0.366 | 0.0089 | 0.027 | 0.455 | 0.545 | +0.090 |
| Lithuania | LT | −0.008 | 0.360 | 0.0091 | 0.026 | 0.428 | 0.524 | +0.097 |
| Luxembourg | LU | −0.017 | 0.408 | 0.0080 | 0.025 | 0.361 | 0.470 | +0.109 |
| Malta | MT | −0.006 | 0.391 | 0.0084 | 0.029 | 0.458 | 0.550 | +0.092 |
| Netherlands | NL | −0.011 | 0.391 | 0.0083 | 0.026 | 0.400 | 0.508 | +0.107 |
| Poland | PL | −0.013 | 0.398 | 0.0082 | 0.026 | 0.382 | 0.491 | +0.109 |
| Portugal | PT | −0.007 | 0.380 | 0.0084 | 0.029 | 0.447 | 0.539 | +0.093 |
| Romania | RO | −0.000 | 0.349 | 0.0093 | 0.033 | 0.539 | 0.598 | +0.059 |
| Slovakia | SK | −0.005 | 0.334 | 0.0096 | 0.029 | 0.464 | 0.542 | +0.078 |
| Slovenia | SI | −0.009 | 0.399 | 0.0082 | 0.028 | 0.420 | 0.524 | +0.104 |
| Spain | ES | −0.008 | 0.390 | 0.0084 | 0.027 | 0.427 | 0.529 | +0.102 |
| Sweden | SE | −0.006 | 0.386 | 0.0085 | 0.030 | 0.459 | 0.549 | +0.090 |

*Notes:* Country-level $\mu_b$ is the area-weighted (by pixel count) average of NUTS2 regional drifts, rescaled from GLOBIO4 MSA to FBI units (×9.3). All other parameters at FBI-NUTS2 calibrated defaults. BF* policy $(\sigma_0^*, \phi^*, \kappa_B^*)$ is optimised at the country level under the budget constraint $C_{\mathrm{BF}} \leq C_{\mathrm{DP}}$. $\Delta\mathbb{E}[b] = \mathbb{E}[b]_{\mathrm{BF}^*} - \mathbb{E}[b]_{\mathrm{DP}}$.

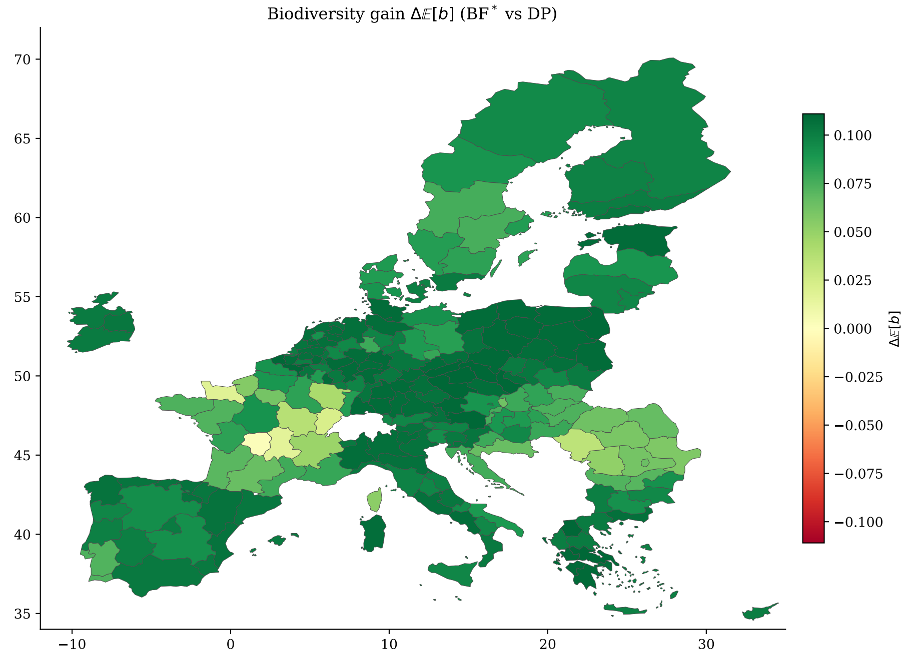


Figure 7: Biodiversity gain $\Delta\mathbb{E}[b]$ (BF* vs. DP) by NUTS2 region under country-level optimal policy (SSP3×RCP6.0). Each region is evaluated under its country's optimal BF* parameters.

### 5.4.3 The limits of blended finance under severe decline: evidence from France

France provides a natural laboratory for assessing BF effectiveness under extreme ecological pressure. With a country-level drift of $\mu_b = -0.027$—approximately 2.5 times the EU median—it exhibits the most severe projected biodiversity decline among the 27 member states. Under BF* alone, the country-level biodiversity gain is $\Delta\mathbb{E}[b] = +0.061$, roughly half the gain achieved in moderate-decline countries such as Austria (+0.113) or Czechia (+0.111).

The 22 French NUTS2 regions span extreme within-country heterogeneity: $\mu_b$ ranges from $-0.081$ (Picardy, FRI2) to $+0.002$ (Île-de-France, FR10). The five regions with $\mu_b < -0.04$ experience almost no welfare improvement from BF* ($\Delta W$ between $-7\%$ and $-0.2\%$), while southern and western regions with moderate decline achieve gains of $+10\%$ to $+18\%$.

Three mechanisms account for the reduced effectiveness of BF under severe biodiversity loss. First, even a substantial farm-level effort $\mathbb{E}[a] \approx 0.012$ leaves the net drift at $\mu_b + \mathbb{E}[a] \approx -0.015$—still strongly negative. The biodiversity SDE $\mathrm{d}b = [\mu_b + a]\,\mathrm{d}t + \sigma_b\,\mathrm{d}\hat{W}$ implies that the exogenous decline dominates the endogenous response. Second, recognising this, the planner optimally sets a lower subsidy ($\sigma_0^* = 0.31$) because the marginal biodiversity return per euro of credit subsidy is diminished when the exogenous decline is large. Third, the BF channel operates indirectly through credit conditions—lower spreads raise farm investment, part of which spills over into biodiversity effort—and this indirect mechanism cannot overcome a sufficiently strong ecological headwind.

A natural question is whether combining both instruments—the credit-channel incentive of BF with the direct biodiversity payment of DP—can improve outcomes in high-pressure regions. We apply the budget-neutral BFDP policy of Section 5.3 to France, holding total fiscal cost constant at $C_{\mathrm{DP}}$ and splitting the budget between BF subsidies and DP payments.

Under BFDP with $\kappa_{\mathrm{DP}} = 0.10$, France achieves $\Delta\mathbb{E}[b] = +0.074$ (versus +0.061 under BF* alone) and the welfare gain rises from +6.0% to +8.9%. Allowing a higher credit subsidy $\sigma_0 = 0.40$ within the BFDP framework yields $\Delta\mathbb{E}[b] = +0.110$ and a net drift of $-0.007$, nearly doubling the biodiversity gain of BF* alone and bringing France close to the levels achieved by moderate-decline countries under BF*. The intuition is that BF and DP are complementary under severe decline: DP provides a direct biodiversity floor that partially offsets the strong negative drift, while BF redirects credit-channel incentives to elicit additional farm-level effort. Neither instrument alone matches the combined effect.

This finding carries a clear policy implication. In regions where biodiversity pressure is moderate ($|\mu_b| \leq 0.015$), BF alone delivers large gains at a fraction of the DP cost. In regions with severe ecological decline, however, a combination of BF and targeted DP is needed to achieve comparable biodiversity outcomes.

# 6 Robustness

## 6.1 Subsidy pass-through friction

A maintained assumption in the baseline model is that the full proportional subsidy $\sigma_0$ is transmitted to the farmer. In practice, financial intermediaries may capture part of the subsidy through imperfect pass-through. We introduce a leakage parameter $\tau \geq 0$ such that the farmer's effective subsidy rate is

$$\sigma_{\mathrm{eff}}(\sigma_0) \;=\; \sigma_0\,(1 - \tau\,\sigma_0), \tag{27}$$

while the government still disburses at the full rate $\sigma_0$. The wedge $\tau\,\sigma_0^2\,\delta(s)$ per unit of subsidised capital is captured by intermediaries and constitutes a pure deadweight loss.

The quadratic form in (27) is motivated by two channels. As the subsidy grows larger, intermediaries face stronger incentives to extract rents via origination fees, unfavourable terms, or delayed disbursement—Lucca et al. (2019) estimate that roughly 40% of federal student-loan subsidies are captured by lending institutions through higher tuition, a pattern that generalises to credit subsidies more broadly. In addition, bureaucratic and verification costs grow with programme scale, creating frictional losses that increase with $\sigma_0$. These frictions are well documented: Agarwal et al. (2018) find incomplete pass-through of credit expansions to borrowers; Barnwal (2024) documents leakage in public transfer programmes; Muralidharan et al. (2016) measure transfer leakage in developing-country programmes; and Philippon (2015) shows that the unit cost of financial intermediation has remained roughly constant at $\sim 2\%$ despite technological progress.

Under leakage, the subsidised rate (4) and the fiscal cost (20) are modified as follows. On the farmer side, the subsidised rate becomes $\delta_{\mathrm{BF}}(s, b) = \delta(s)(1 - \sigma_{\mathrm{eff}}) - \phi\, b$, so that all optimal controls ($x^*_{\mathrm{BF}}$, $k^*_{\mathrm{BF}}$) and the biodiversity drift $a_{\mathrm{BF}}$ respond to $\sigma_{\mathrm{eff}}$ rather than $\sigma_0$. On the government side, fiscal cost is computed at the full $\sigma_0$, reflecting that the government pays the full amount even though only a fraction $(1 - \tau\sigma_0)$ reaches the farmer. The anti-arbitrage penalty (23) evaluates the violation using $\sigma_{\mathrm{eff}}$, since only the effective subsidy determines the farmer-facing rate. Because $\sigma_{\mathrm{eff}}$ is concave in $\sigma_0$ and peaks at $\sigma_0 = 1/(2\tau)$, leakage creates an additional force toward an interior optimum even without programme costs: beyond $1/(2\tau)$, raising $\sigma_0$ reduces the effective subsidy while increasing fiscal cost.

The pass-through literature suggests $\tau \in [0.3, 0.6]$: Lucca et al. (2019) imply $\tau \approx 0.4$ for credit subsidies, and Barnwal (2024) documents leakage rates of 30–50% in public programmes. We report results for $\tau \in \{0, 0.3, 0.6\}$.

Table 10: Optimal BF policy under subsidy leakage ($\tau$).

| $\tau$ | $\sigma_0^*$ | $\sigma_{\text{eff}}^*$ | $\phi^*$ | $\kappa_B^*$ |
|---|---|---|---|---|
| 0 (baseline) | 0.350 | 0.350 | 0.015 | 0.030 |
| 0.3 | 0.350 | 0.313 | 0.015 | 0.030 |
| 0.6 | 0.350 | 0.277 | 0.015 | 0.030 |

*Note:* $\sigma_{\text{eff}}^* = \sigma_0^*(1 - \tau\sigma_0^*)$. $100 \times 50$ grid, inequality budget constraint ($C_{\text{BF}} \leq C_{\text{DP}}$).

Table 11: Equilibrium outcomes under subsidy leakage: NP, DP, and optimal BF.

| | **NP** | **DP** | **BF** ($\tau$=0) | **BF** ($\tau$=0.3) | **BF** ($\tau$=0.6) |
|---|---|---|---|---|---|
| *Biodiversity* | | | | | |
| $\mathbb{E}[b]$ | 0.387 | 0.400 | **0.509** | **0.497** | **0.486** |
| $\text{Var}(b)$ | 0.033 | 0.035 | 0.047 | 0.047 | 0.047 |
| $\mathbb{E}[a]$ (effort) | 0.000 | 0.002 | **0.013** | **0.012** | **0.010** |
| $\mathbb{E}[a]/\mathbb{E}[a]_{\text{DP}}$ | 0% | 100% | **787%** | **683%** | **583%** |
| *Output* | | | | | |
| $\mathbb{E}[q]$ (EUR) | 44,133 | 45,110 | **57,061** | 55,412 | 53,630 |
| *Fiscal cost* | | | | | |
| $C$ (EUR) | 0 | 4,003 | 687 | 572 | 467 |
| $C_{\text{BF}}/C_{\text{DP}}$ | — | 100% | **17%** | **14%** | **12%** |
| *Welfare* | | | | | |
| $W$ | 47,825 | 47,734 | **58,194** | **57,041** | **55,530** |
| $\Delta W$ vs. DP | +0.2% | — | **+21.9%** | **+19.5%** | **+16.3%** |

*Note:* $100 \times 50$ grid with inequality budget constraint ($C_{\text{BF}} \leq C_{\text{DP}}$) and hard anti-arbitrage constraint $\delta_{\text{BF,min}} \geq r_f$. NP = No Policy ($\lambda = 0$, $\kappa_{\text{DP}} = 0$). All parameters at low-rate era baseline (Table 1). BF policy parameters for each $\tau$ in Table 10.

Tables 10 and 11 establish that the dominance of BF over DP is robust to intermediary rent extraction across the entire empirically calibrated range. At moderate leakage ($\tau = 0.3$, where the farmer receives an effective subsidy of $\sigma_{\text{eff}}^* = 0.313$ rather than 0.350), BF still delivers +19.5% higher welfare and +24% higher expected biodiversity than DP. Even under severe capture ($\tau = 0.6$, where 21% of the credit subsidy is lost to intermediaries), BF achieves +16.3% higher welfare and raises $\mathbb{E}[b]$ from 0.400 to 0.486. Farm-level biodiversity effort remains six to eight times higher than under DP at all leakage levels.

The planner's optimal policy is invariant to $\tau$: $\sigma_0^* = 0.35$, $\varphi^* = 0.015$, and $\kappa_B^* = 0.030$ are selected at every leakage level, with $\sigma_{\text{eff}}^*$ declining mechanically. This invariance indicates that the credit-channel design is robust to intermediary capture—the planner does not need to restructure incentives as the effective subsidy erodes. The fiscal cost ratio $C_{\text{BF}}/C_{\text{DP}}$ ranges from 17% at baseline to 12% at $\tau = 0.6$: higher leakage narrows the effective subsidy, reducing both farmer-side benefits and government expenditure, but BF remains substantially cheaper than DP at every level.

# 7 Conclusion

This paper has developed a continuous-time heterogeneous-agent model to evaluate blended finance as an instrument for biodiversity conservation on agricultural land. The model features two endogenous state variables—farm size and a biodiversity index—and allows farms to choose capital, subsidised intensity, biodiversity effort, and exit timing. We calibrate the model to EU agricultural and ecological data and compare optimised blended finance (BF) with the existing Common Agricultural Policy direct payment (DP) system.

The quantitative analysis yields five main findings.

First, optimised BF welfare-dominates DP in both interest-rate environments we consider. In the low-rate era, BF raises expected biodiversity by 28% and is the only policy regime that reverses the net biodiversity decline, turning a negative drift of $-0.009$ per year under DP into a positive drift of $+0.002$. These gains are achieved at 84% lower fiscal cost than DP, with a biodiversity leverage ratio approximately 50 times higher. The mechanism is that BF operates through the cost of capital rather than through unconditional transfers: the subsidised rate $\delta_{\text{BF}}(s, b) = \delta(s)(1 - \sigma_0) - \phi b$ creates a continuous incentive gradient in biodiversity, eliciting seven times more farm-level conservation effort than DP.

Second, the instrument is automatically progressive. Because the proportional subsidy $\sigma_0\delta(s)$ scales with the private borrowing cost—which is steeply decreasing in farm size—small, credit-constrained farms receive the largest absolute rate reduction. This lowers exit thresholds, extends the survival of small farms, and compresses the output distribution, reducing the Gini coefficient from 0.295 under DP to 0.280 under BF.

Third, BF dominates DP across all 27 EU member states, with expected biodiversity gains ranging from +0.06 in France (severe decline) and Romania (near-zero decline) to +0.11 in Austria, Czechia, and Poland (moderate decline). The optimal policy parameters are remarkably stable across the moderate-decline group, suggesting that a single EU-wide BF design would capture most of the potential gains. However, in regions with severe biodiversity pressure—illustrated by the French case study, where the country-level drift is 2.5 times the EU median—BF alone cannot fully offset the exogenous ecological decline. Combining BF with targeted direct payments in a budget-neutral hybrid (BFDP) nearly doubles the biodiversity gain, revealing a policy complementarity between the credit and transfer channels under extreme conditions.

Fourth, the welfare cost of imposing the anti-arbitrage constraint $\delta_{\text{BF,min}} \geq r_f$—which prevents the subsidised rate from falling below the risk-free rate and eliminates carry-trade incentives—is moderate: 1–4 percentage points of welfare relative to a relaxed design. This is a modest premium for the regulatory simplicity and market discipline that the constraint provides.

Fifth, the dominance of BF over DP is robust to subsidy pass-through frictions. Even when 60% of the credit subsidy is captured by financial intermediaries ($\tau = 0.6$), BF delivers 16% higher welfare and 22% higher expected biodiversity than DP. The planner's optimal policy parameters are invariant to $\tau$, indicating that the credit-channel design does not need to be restructured in the presence of intermediary rent extraction.

Several limitations suggest directions for future research. The model assumes a stationary environment and does not capture transition dynamics—how the farm distribution adjusts from the current state to the long-run equilibrium under a new policy. Extending the Fokker–Planck framework to a time-dependent setting would allow the analysis of phase-in paths, adjustment costs, and the political economy of gradual reform. The biodiversity index is scalar, whereas real ecological outcomes are multidimensional; incorporating species-specific or habitat-specific indicators would refine the welfare analysis. The model treats the biodiversity monitoring technology as given; in practice, the cost and accuracy of ecological monitoring are themselves policy-relevant and could be endogenised. Finally, general equilibrium effects—changes in land prices, output prices, and credit markets in response to a large-scale BF programme—are absent from the partial equilibrium framework; embedding the farm-level problem in a spatial or sectoral equilibrium model would be a natural extension.

Despite these caveats, the results suggest that redirecting a fraction of existing CAP spending from unconditional per-hectare transfers toward a biodiversity-conditional credit instrument could substantially improve ecological outcomes on European farmland. The credit channel mobilises private co-investment, aligns farm-level incentives with biodiversity goals through the cost of capital, and achieves broad participation through voluntary self-selection—all at a fraction of the fiscal cost of the current direct payment system. Whether such a reform is politically feasible is, of course, a question that lies beyond the scope of a structural model; but the quantitative magnitudes documented here—a 50-fold improvement in biodiversity leverage and an 84% reduction in fiscal cost—suggest that the potential welfare gains are large enough to warrant serious policy consideration.

# References


Adamopoulos, T., Brandt, L., Leight, J., and Restuccia, D. (2022). Misallocation, selection, and productivity: A quantitative analysis with panel data from China. *Econometrica*, 90(3):1261–1282.

Adamopoulos, T. and Restuccia, D. (2014). The size distribution of farms and international productivity differences. *American Economic Review*, 104(6):1667–1697.

Agarwal, S., Chomsisengphet, S., Mahoney, N., and Stroebel, J. (2018). Do banks pass through credit expansions to consumers who want to borrow? *The Quarterly Journal of Economics*, 133(1):129–190.

Baker, M., Bergstresser, D., Serafeim, G., and Wurgler, J. (2022). The pricing and ownership of US green bonds. *Annual Review of Financial Economics*, 14:415–437.

Bareille, F. and Largier, B. (2026). A meta-analysis on the productive value of crop biodiversity. *American Journal of Agricultural Economics*.

Barnwal, P. (2024). Curbing leakage in public programs: Evidence from India's direct benefit transfer policy. *American Economic Review*, 114(12):3812–3846.

Chabé-Ferret, S. and Subervie, J. (2013). How much green for the buck? Estimating additional and windfall effects of French agro-environmental schemes by DID-matching. *Journal of Environmental Economics and Management*, 65(1):12–27.

Featherstone, A. M. and Moss, C. B. (1994). Measuring economies of scale and scope in agricultural banking. *American Journal of Agricultural Economics*, 76(3):655–661.

Flammer, C. (2021). Corporate green bonds. *Journal of Financial Economics*, 142(2):499–516.

Flammer, C., Giroux, T., and Heal, G. (2024). Blended finance. Technical report, National Bureau of Economic Research.

Gloy, B. A., Gunderson, M. A., and LaDue, E. L. (2005). The costs and returns of agricultural credit delivery. *American Journal of Agricultural Economics*, 87(3):703–716.

Gregory, R. D., Vorisek, P., Noble, D. G., Van Strien, A., Klvanova, A., Eaton, M., Gmelig Meyling, A. W., Joys, A., Foppen, R. P., and Burfield, I. J. (2019). An analysis of trends, uncertainty and species selection shows contrasting trends of widespread forest and farmland birds in Europe. *Ecological Indicators*, 103:284–295.

Hansen, L. P. and Scheinkman, J. A. (2025). Emission prices, biomass, and biodiversity in tropical forests. In *AEA Papers and Proceedings*, volume 115, pages 391–396. American Economic Association 2014 Broadway, Suite 305, Nashville, TN 37203.

Heckman, J. J. and Smith, J. A. (2004). The determinants of participation in a social program: Evidence from a prototypical job training program. *Journal of Labor Economics*, 22(2):243–298.

Hopenhayn, H. A. (1992). Entry, exit, and firm dynamics in long run equilibrium. *Econometrica: Journal of the Econometric Society*, pages 1127–1150.

IPBES (2019). Global assessment report on biodiversity and ecosystem services. Technical report, Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany.

Karpf, A. and Mandel, A. (2018). The changing value of the 'green' label on the US municipal bond market. *Nature Climate Change*, 8(2):161–165.

Liang, J., Crowther, T. W., Picard, N., Wiser, S., Zhou, M., Alberti, G., Schulze, E.-D., McGuire, A. D., Bozzato, F., Pretzsch, H., et al. (2016). Positive biodiversity–productivity relationship predominant in global forests. *Science*, 354(6309):aaf8957.

Lucca, D. O., Nadauld, T., and Shen, K. (2019). Credit supply and the rise in college tuition: Evidence from the expansion in federal student aid programs. *The Review of Financial Studies*, 32(2):423–466.

Miao, J. (2005). Optimal capital structure and industry dynamics. *The Journal of finance*, 60(6):2621–2659.

Moretti, E. and Benzaquen, M. (2024). Mitigating farmland biodiversity loss: A bio-economic model of land consolidation and pesticide use. *arXiv preprint arXiv:2407.19749*.

Mundlak, Y. (1978). On the pooling of time series and cross section data. *Econometrica*, 46(1):69–85.

Muralidharan, K., Niehaus, P., and Sukhtankar, S. (2016). Building state capacity: Evidence from biometric smartcards in India. *American Economic Review*, 106(10):2895–2929.

Newbold, T., Hudson, L. N., Hill, S. L., Contu, S., Lysenko, I., Senior, R. A., Börger, L., Bennett, D. J., Choimes, A., Collen, B., et al. (2015). Global effects of land use on local terrestrial biodiversity. *Nature*, 520(7545):45–50.

OECD (2018). Making blended finance work for the Sustainable Development Goals. Technical report, OECD Publishing, Paris.

Pe'er, G., Dicks, L. V., Visconti, P., Arlettaz, R., Báldi, A., Benton, T. G., Collins, S., Dieterich, M., Gregory, R. D., Hartig, F., et al. (2014). EU agricultural reform fails on biodiversity. *Science*, 344(6188):1090–1092.

Petersen, M. A. and Rajan, R. G. (1994). The benefits of lending relationships: Evidence from small business data. *The Journal of Finance*, 49(1):3–37.

Philippon, T. (2015). Has the US finance industry become less efficient? On the theory and measurement of financial intermediation. *American Economic Review*, 105(4):1408–1438.

Schipper, A. M., Hilbers, J. P., Meijer, J. R., Antão, L. H., Benítez-López, A., de Jonge, M. M. J., Huijbregts, M. A. J., Alkemade, R., et al. (2020). Projecting terrestrial biodiversity intactness with GLOBIO 4. *Global Change Biology*, 26(2):760–771.

Stiglitz, J. E. and Weiss, A. (1981). Credit rationing in markets with imperfect information. *American Economic Review*, 71(3):393–410.

TEEB (2010). *The Economics of Ecosystems and Biodiversity: Mainstreaming the Economics of Nature*. UNEP.

Tilman, D., Clark, M., Williams, D. R., Kimmel, K., Polasky, S., and Packer, C. (2017). Future threats to biodiversity and pathways to their prevention. *Nature*, 546(7656):73–81.

Wuepper, D. and Huber, R. (2022). Comparing effectiveness and return on investment of action- and results-based agri-environmental payments in Switzerland. *American Journal of Agricultural Economics*, 104(5):1585–1604.

Zerbib, D. O. (2019). The effect of pro-environmental preferences on bond prices: Evidence from green bonds. *Journal of Banking & Finance*, 98:39–60.