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ARTICLE

Comparing effectiveness and return on investment of action- and results-based agri-environmental payments in Switzerland

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Abstract

Agri-environmental schemes are an important policy tool to foster agricultural sustainability. We assess the effectiveness and return on investment of two different schemes designed to encourage biodiversity conservation in Switzerland: payments for actions and payments for results. Empirically, we exploit a major policy reform that created a natural experiment by abruptly and unevenly increasing both payments across farmers. Using difference in differences, we estimate the effect of the policy reform on farmers for whom only the action- or the results-based payments increased, as well as on those for whom both increased, compared to farmers for whom neither increased. Our findings are fourfold: First, higher payments increased the biodiversity conservation area. A payment raise by 1% increased conservation areas on average by 0.6% in the action based, and by 1% in the results-based scheme. Second, the combination of both schemes increased average effectiveness but also windfall gains. Third, using a benefit transfer approach, we estimate a positive return of investment for all payment increases. Finally, the estimated return on investment for the results-based payments is higher than for the action-based payments.

David Wuepper and Robert Huber have contributed equally.

KEYWORDS
additionality, agri-environmental schemes, action based, biodiversity conservation, results based, return on investment, treatment effects, windfall effects

JEL CLASSIFICATION
Q18, Q57, Q58

1 | INTRODUCTION

A shift to more sustainable farming practices is an essential goal of agricultural policy worldwide. Agri-environmental schemes[1] are one policy tool to achieve this by incentivizing farmers to adopt sustainable farming practices, such as extensive grassland management or creating wildflower verges. The European Union and the US each spend about 4 and 6 billion euros annually on agri-environmental schemes, respectively (Science for Environment Policy, 2017; USDA, 2019). With the European Green Deal and the EU Farm to Fork strategy they will further gain importance (e.g. Pe'er et al., 2019; de Schutter et al., 2020; Pe'er et al., 2020; Scown et al., 2020). Most payments are action based, that is, they prescribe a certain farming practice, but they are not linked to achieving any environmental outcomes (Hanley et al., 2012). Action-based payments have been criticized for their possibly low cost effectiveness, and results-based payments have been proposed as a possible improvement (Burton & Schwarz 2013, 2013; Börner et al. 2017). There is, however, almost no empirical evidence on the relative performance of action- and results-based schemes.

Here, we assess the effectiveness and return on investment of action- and results-based agri-environmental payments for biodiversity conservation, both individually and also in combination. An abrupt reform of an ambitious agri-environmental program in Switzerland serves as our point of departure. In 2014, existing action- and results-based agri-environmental payments aimed to promote farmland biodiversity were considerably increased and augmented, but not all and not everywhere (Metz et al. 2020). Individual payments increased by up to 100% per hectare, depending both on the payment type and the location of the farms. Thus, our data include farmers for whom only action-based payments increased, farmers for whom only results-based payments increased, farmers for whom both increased (our three treatment groups), and farmers for whom no payments increase (our control group). Using panel data for all Swiss farms and a difference-in-differences design, we estimate how much additional biodiversity conservation area has been created by the payment increases. In addition, we estimate windfall effects, that is, how much of the payment increases only generated extra income without achieving additional biodiversity conservation. Finally, we use a benefit transfer approach to estimate returns on investments.

In theory, given perfect information about farm types and under identical ecosystem assets, action- and results-based payments could be equally effective (White & Hanley 2016). However, in real world settings, hidden information on farmers' provision costs and heterogeneous ecological potential may lead to adverse selection and moral hazard (Hanley et al. 2012). In the case of action-based payments (e.g., payments for low intensity grassland use), this implies that farmers may choose to be paid for practices they would have implemented anyway, and thus the resulting additionality is low (Claassen et al. 2018, Wunder et al. 2018). Furthermore, farmers might be less prone to comply with the practices if compliance costs are high (Hart & Latacz-Lohmann 2005). In contrast, the advantage of results-based payment schemes is that they do not reward farmers for a

[1]Definitions: Agri-environmental measure: A voluntary contract that pays farmers in exchange for a practice or a result. Agri-environmental scheme: A set of agri-environmental measures that have the same design characteristic, for example, action- or results-based. Agri-environmental program: A combination of agri-environmental schemes with the same objective, for example, support biodiversity conservation.

specific management action but for achieving measurable improvements, for example, payments depend on whether certain indicator species—often rare plants—are found (Burton & Schwarz 2013, 2013; Wunder et al. 2018). Generally, the relative performance of results-based schemes increases with rising uncertainty regarding the link between actions and outcomes (Gibbons et al. 2011, Börner et al. 2017). Farmers can use their experience and local knowledge to decide what they consider to be the most effective ways to achieve targeted outcomes (Burton & Schwarz 2013, 2013). This can counteract selection bias and moral hazard (Börner et al. 2017). Overall, the theoretical literature suggests that results-based measures might be more cost effective than action-based measures.

However, action-based schemes also have their advantages (e.g. Burton & Schwarz 2013, 2013). First, they are easier to implement, and monitoring is less costly, as they do not need complex indicators or field surveillances. Second, acceptance by farmers is often higher as they are perceived to be more fair, in the sense that all farmers are offered the same payment for a given action and that the payments do not change because of factors outside of their control, such as climatic variations (Hanley et al. 2012). Thus, regulations that focus on voluntary results-based payments could well discourage participants (Zabel & Roe 2009), and the desired biodiversity goals may be missed entirely if, for example, this is linked to a certain minimum habitat area (Ohl et al. 2008, Hanley et al. 2012).

Other theoretical studies also proposed that a combination of action- and results-based schemes could be more cost effective than either approach alone, because this might balance the farmers' risk considerations and the stochastic nature of the environment (Zabel & Roe 2009, Derissen & Quaas 2013). However, combining schemes also entails a challenge because addressing the same goal with multiple instruments may lead to inefficiencies (Tinbergen 1968) and increase farmers' windfall gains (Huber et al. 2017, Bartolini et al. 2021). Overall, the assessment of the relative cost-effectiveness of action- and results-based payments, and their combination, remains an important empirical question (Börner et al. 2017). A few previous studies have estimated the effect of single agri-environmental schemes and estimated their effect on the intended actions and outcomes, often followed by an analysis of their return on investment (e.g. Chabé-Ferret & Subervie 2013, Laukkanen & Nauges 2014, Claassen et al. 2018, Bertoni et al. 2020). However, a direct comparison of investments in schemes of varying designs (and combinations thereof) has not yet been carried out.

An optimal comparison of action- and results-based payments would analyze how each affected a direct measure of biodiversity. Such data are currently not available. However, Swiss farmers can receive both action- and results-based payments for the same plot of land, for example, they can be paid for extensive grassland management and then receive additional payments if experts find a certain number of rare indicator species on this land (which are region specific and thus take into account underlying environmental characteristics). Crucially, the receipt of results-based payments then implies that an area is of higher biological value than the land for which only the action-based payment is received. This allows us to analyze changes in the extent of extensive and species-rich grassland and using a benefit transfer to compute cost benefit ratios for the investments into the schemes.

Our dataset is derived from a comprehensive farm census survey of all Swiss farms for the period 2010 to 2017 ($N = 386{,}711$). Our empirical framework are different versions of difference-in-differences designs, combined with matching, including recent methodological advances such as coarsened exact matching (Bertoni et al. 2020), doubly robust difference in differences (Ho et al. 2007, Sant'Anna & Zhao 2020), fully flexible difference in differences (Mora & Reggio 2019), as well as the honest approach to parallel trends (Rambachan & Roth 2019). Our analysis of the return on investment is based on a benefit transfer approach proposed in a recent meta-analysis on willingness to pay for ecosystem services by Huber and Finger (2020), which focused precisely on the subjects of our study.

We find that that a 1% rise in payments led to an average increase in 0.6% and 1% in areas enrolled in the action- and results-based scheme, respectively. In our treatment group in which both payments increased, the average elasticity of supplying conservation areas amounts to 0.9 for the action- and 1.2% for the results-based scheme. At the same time, average windfall effects also increased in this treatment group. The payment effectiveness is, on average, higher in action-based payment, that is, the governmental needs to spend less to get an additional hectare of conservation area. However, we also find that the benefits are higher than the governmental spending for both schemes, resulting in a positive return on investment. The return on investment is on average three times higher for the results-based scheme because our benefit transfer estimates suggest that conservation areas under the results-based scheme are considerably more valuable.

The next section provides a brief policy background to our analysis, which is followed by a description of our data and details of our empirical strategy. The following section presents our main results, which are discussed in the subsequent section, followed by a conclusions.

2 | BACKGROUND

Swiss agriculture is characterized by small-scale farming systems embedded in intensively used multifunctional landscapes, some of which harbor biodiversity hot spots, for example, in mountain areas (Huber, Snell et al. 2017). At the same time, Switzerland numbers among the countries providing the highest support for agriculture worldwide and faces multiple pressures on the environment and especially biodiversity, for example, through intensive use of pesticides (Möhring et al. 2020) and large nitrogen surpluses (Wuepper et al. 2020).

In 1999, as a reaction to these environmental pressures, Switzerland introduced an agri-environmental program to maintain and support agricultural biodiversity (Huber & Finger 2019). Initially, this program only included an action-based scheme, that is, farmers received a payment for managing land according to specified practices that support biodiversity conservation. For example, farmers received payments for a low-intensity use of hay meadows. This meant that they were not allowed to cut the grass before June 15 or to use fertilizer on grassland (Herzog et al. 2005). It is important to note that the measures in this program were also part of cross-compliance regulations, which obliged farmers to participate in the scheme with at least 7% of their total land or risk losing the eligibility for receiving direct payments. This implies that basically all Swiss farmers must enroll land in the action-based, agri-environmental scheme. To encourage the enrollment of areas with higher biological value, a result-based payment was introduced in 2003. The results-based scheme is paid in addition to the action-based payment, that is, these payments are additive on the same plot of land. To get these additional results-based payments, farmers have to request a botanical assessment of their grasslands and if a minimum number of predefined rare species from the red list can be found (see e.g., Mack, et al. 2020 for details), they receive an amount that depends on where this land is located, for example, in lowlands or mountain regions. As a rule, an expert visits the farm before the first cut in June to assess the botanical quality by searching for species from a list that allows the "regional potential" of the corresponding field to be defined. This means that the list of species indicating the botanical quality of the field is adjusted to account for the exposition and altitude in the corresponding region of Switzerland. The expert must follow a strict protocol (where to start searching, how to consider edges, etc.) to prevent bias in the assessment of the quality. It should be noted that the final assessment results in a categorical measurement, that is, either the plot of land is eligible for the results-based payment or not, even though the biological quality is a continuous variable and there might be variability of biological quality also within areas eligible only for the action-based or both payments. For example, if the threshold for indicator plants is six, extensive grassland is categorized to have the same biological value whether it has none, or five indicator plants, and above the threshold, the same is true for species-rich grassland, whether it has seven or ten indicator plants.

After the first assessment, the evaluation is repeated in the course of the regular farm visits by the authority that controls compliance with Swiss regulations, that is, this also applies to the action-based scheme. These farm visits follow a risk-based approach, which means that farms that changed production or failed to comply with certain regulations in an earlier control are visited more often. Each farm in Switzerland must be controlled at least every 4 years by the respective authority.

Empirical research shows that the program achieved to avoid further biodiversity loss in the Swiss agricultural landscapes. Action-based payments supported, for example, the abundance of birds and vascular plants (Herzog et al. 2005, Kampmann et al. 2008, Albrecht et al. 2010, Zingg et al. 2018, Zingg et al. 2019). However, the total enrolled area and its botanical quality remained below what was aimed for (e.g., Birrer et al. 2007, Zellweger-Fischer et al. 2011). This motivated a major policy reform in 2013 to improve the program by investing considerably more money (by 140 million CHF, which was a plus of 60%) and changing all the individual payments, already starting in the next year (2014; see Metz et al. 2020 for details).

These adjustments were made on the basis of both farm location and service provided. For example, results-based payments for the detection of indicator plants on extensive grassland rose by 500 CHF for the valley and the hill zones as well as for two of the four Swiss mountain zones and by 300 CHF in the other mountain zones. Some payments were increased uniformly across zones but not evenly across the service provided, that is, the payments for hedges and other structures were increased by 500 CHF per hectare in all zones, whereas payments for wildflower verges and biodiversity strips were increased by 1000 CHF per hectare in all zones. On the other hand, some payments did not rise at all in some zones. Importantly, farmers received the higher payments also for the areas that were already in the program, and they could enroll additional areas into both schemes. This means, potentially some farmers only received more money without making any changes ("windfall gains").

In our dataset, we have farmers for whom only the action-based payments rose after 2013 but not the results-based payments, there are farmers for whom only results-based payments rose but not the action-based payments, there are farmers for whom both payments rose, and there are farmers for whom neither rose. With these different treatment groups and the control group, we are able to estimate—using difference in differences—how much the increased payments incentivized farmers to expand their biodiversity conservation, how much they increased windfall, and their net welfare effect.

A peculiarity of the reform was that the vote on the reform was held in June 2013 and the changes in the direct payment system came into effect in January 2014. Thus, farmers could not anticipate the design of the new system and adjust beforehand (OECD 2017).

3 | DATA

Our analysis is based on census data from the Swiss Federal Office for Agriculture and consists of comprehensive information on all Swiss farms in the period between 2010 and 2017. For each farm and year, it contains: (i) the area (in ha) for the farming practices that support biodiversity conservation; (ii) the amount of payments (in CHF) for these practices; (iii) structural variables such as farm size, land use, type, intensity, and labor; and (iv) all other direct payments to Swiss farmers. All area-related variables are shown in Table 1, all other variables are shown in Table 2. The changes in agri-environmental payments in the three treatment groups are shown in Figure 1.

Figure 2 shows the temporal development of the extent of extensive and species-rich grassland, as well as biodiversity measures on cropland (e.g., wildflower verges), all in hectares, in our three treatment groups (defined by which payments increased after the reform in 2013), compared to a common control group (for which no payments increased). It can be seen that the level of biodiversity conservation areas was almost always higher in any treatment than the control group, regularly also growth paths differed already before the reform, but around the reform, there are also visible changes in some growth paths.

TABLE 1 Outcome variables: Biodiversity conservation practices

Variable	Description	Mean (*SD*)
Extensive grassland	Extensively managed, mown only once per year, after date of flowering, and no fertilization	3.00 (4.09) ha
Species-rich grassland	grassland with at least six indicator plants, identified by experts	1.02 (2.40) ha
Biodiversity measures on cropland	Wildflower verges (wildflower strips on arable land for at least two vegetation periods), biodiversity strips (strips of wildflowers along crops for at least two vegetation periods of which only on half is mown each vegetation period), biodiversity reserves (strips of native wild herbs alongside cereals, rapeseed, sunflowers and legumes for at least two vegetation periods), permanent wildflower fallows, rotational wild herbs fallows	0.06 (0.42) ha

Note: We have three outcome variables of interest: (1) area under extensive grassland, which is targeted with the action-based scheme; (2) species-rich grassland, which is targeted with the results-based scheme; (3) biodiversity enhancing practices on cropland, which are again targeted with the action-based scheme.

4 | EMPIRICAL ANALYSIS

In this section, we present (a) our empirical identification strategy, including robustness tests, and (b) how we computed the returns on investment.

4.1 | Identification strategy

Our identification strategy involves estimating different versions of difference in differences (combined with matching) between each of the three separate groups of farmers who, in 2014, either received higher action-based payments, higher results-based payments, or both, and a control group of farmers who received no increase in agri-environmental payments at all. This control group mostly consists of farmers who either provide services for which payments were not increased at all or not in the specific growing zone in which they are located. We generally estimate average treatment effects on the treated (ATT). For the comparisons of action- and results-based payments (and their combination), we additionally assume that the treatment groups respond similar to the same price change.

The main identification issue we encounter is selection bias because we cannot assume that the farmers who choose to enroll in voluntary AES are comparable to those who do not, that is, that they share a common counterfactual. The difference-in-differences design, often combined with matching techniques, is a methodological option to identify treatment effects in this setting (Chabé-Ferret & Subervie 2013, Chabé-Ferret 2015, Bertoni et al. 2020). In its simplest form, the assumption that participating and non-participating farmers are fully comparable is relaxed, and it is only assumed that the outcomes of interest would have changed in parallel over time without treatment (the "parallel trends" assumption). If fulfilled, this means that post-treatment deviations from parallel trends can be interpreted as treatment effect. However, this assumption is still strong (Rambachan & Roth 2019, Marcus & Sant'Anna 2021), and below we explain four approaches how to address this challenge.

First, the plausibility of parallel trends can be assessed by comparing the balance of observable characteristics between the treatment and control group and by testing whether, before treatment, trends in the treatment and control group were parallel (Angrist & Pischke 2008). The standard difference-in-differences is then estimated as:

TABLE 2 Control variables: Payments and structural characteristics of Swiss farms

Variable	Unit	Description	Mean (*SD*)
Direct payments*			
Food supply	CHF	Payments for ensuring food supply: (a) basic payment and (b) payments for arable land	32,314 (23,583)
Difficult production conditions	CHF	Payments for production under difficult conditions (mountainous and steep slopes)	8793 (11,361)
Grass-based milk and meat	CHF	Payments for grass-based milk and meat production (restricted maize and concentrate feed in cattle feeding)	1117 (2175)
Extensive crop		Extensive crop production (no or reduced pesticide use allowed)	7114 (8254)
Organic agriculture	CHF	Payments for organic production	795 (3453)
Landscape maintenance	CHF	Payments for cultural landscapes on hillsides and mountain regions	1148 (2578)
Agglomeration	CHF	Bonus payment for connecting conservation areas	1476 (2400)
Farm structures			
Livestock units	Unit	Standardized units of livestock	26 (27)
Farm area	Unit	All land per farm (crop rotation and grassland)	21 (16)
Crop area	Ha	Crop rotational area per farm	5.5 (10.3)
Grass area	Ha	Grassland area per farm	9.7 (9.6)
Labor	Standard labor unit	Standardized units of labor	1.8 (1.7)
Age	Years	Age of farmer	47 (10)
Production zone	Category	Lowlands, hilly region, mountain areas	
Farm types**	Category	Farm typology, that is, dairy farms, crop farm	

Note: * for details see e.g., Huber et al. (2017) ** farm typology based on Mack et al. (2020).

$$E[Y_{it}|D_i = D] = \delta + \sum_{\tau=2}^{T} \delta_\tau I_t^\tau + \gamma D + \gamma_T \times I_t^T \times D \quad (1)$$

where Y_{it} is an outcome (e.g., hectares of extensive or species-rich grassland), D identifies the treatment groups (e.g., those with an increase in action-, results-based, or both payments) with $D \in \{0,1\}, I_t^T$ is a dummy for the post-treatment period, and γ_T identifies our treatment effect of interest. Moreover, I_t^τ is a dummy for period τ, which means it takes value one if $t = \tau$ and zero otherwise, and δ and γ are regression coefficients to be estimated.

Different solutions are available to deal with suspicions that parallel trends might be violated, for example, due to covariate imbalance and/or deviating pretreatment trends. First, pretreatment control variables can be used either directly as covariates or, more commonly, via a range of matching techniques, such as Propensity Score, Mahalanobis, Nearest Neighbor, or Coarsened Exact Matching (Chabé-Ferret & Subervie 2013, Bertoni et al. 2020, Sant'Anna & Zhao 2020). We always allowed the same farmers to be matched multiple times (one-to-many matching) in order not to lose any observations. For the nearest neighbor matching, the distance metric between the farmers was their propensity score, based on simple logistic regressions.

Secondly, matching and difference-in-differences can also be combined to yield a "doubly robust" estimator that only requires one of the two techniques to be valid in order to give an unbiased ATT estimate (Ho et al. 2007, Sant'Anna & Zhao 2020). Formally, the estimator for repeated cross-sectional data and propensity score matching as matching algorithm is (Sant'Anna & Zhao 2020):

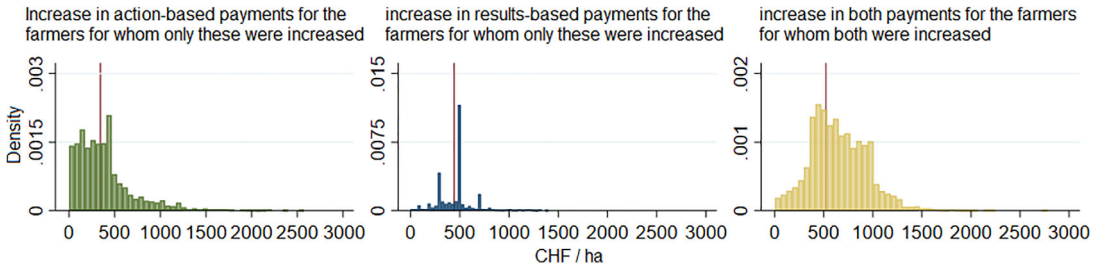

FIGURE 1 Payment increases in treatment groups. The reform increased payments in the biodiversity program with heterogeneous increases across measures and growing zones (see Appendix, Figure S1 for a map of these growing zones and Figure S2 for maps of the spatial distribution of the three treatment groups and the control group). The figure shows the average payment increases per farm after the reform in Swiss Francs (roughly equivalent to USD and EUR) in the three treatment groups, containing 33%, 6%, and 47% of the farmers, respectively. The remaining farmers are in the control group (14%)

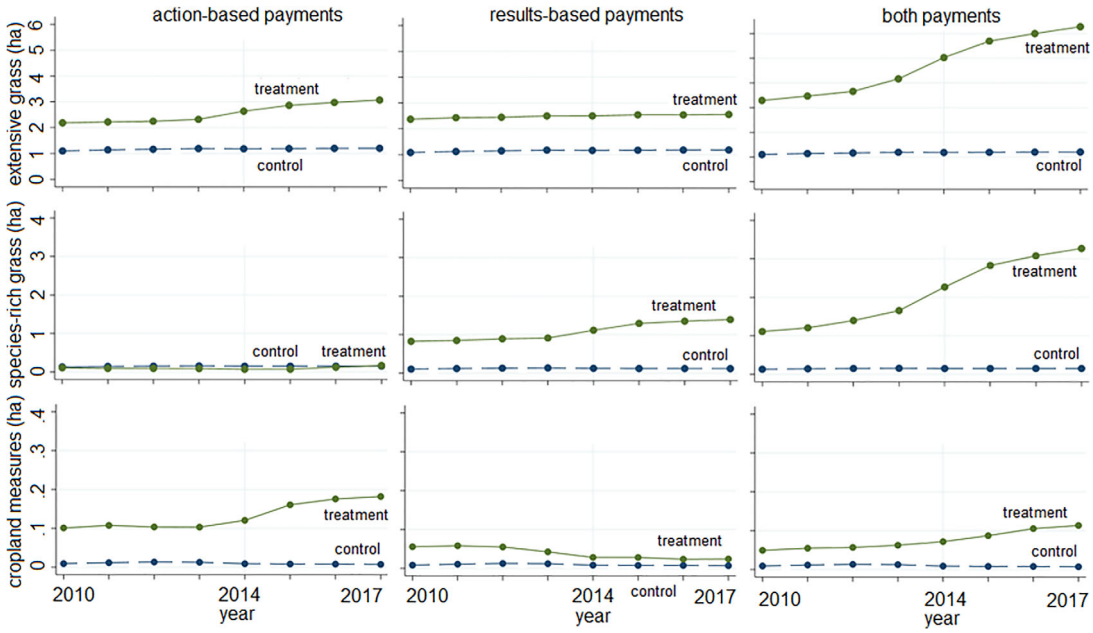

FIGURE 2 Observed dynamics in treatment and control groups. There are three treatments (increased action-based payments, increased results-based payments, both increased), three outcomes (extensive grassland, species-rich grassland, biodiversity measures on cropland), and one common control group (no increase in any agri-environmental payments)

$$\gamma_T = \frac{1}{\mathbb{E}[D]}\mathbb{E}\left[\frac{D-p(X)}{1-p(X)}\frac{T-\lambda}{\lambda(1-\lambda)}Y\right] \tag{2}$$

where $p(X)=\mathbb{P}(\mathrm{D}=1|X)$ is the propensity score for each farmer, and $\lambda=\mathbb{P}(\mathrm{T}=1)$.

We use two kinds of covariates for matching farms. The first set includes all the various payments received by Swiss farmers. A particularity of the Swiss system is that all payments are additive, so receipt of one payment has no effect on eligibility for other payments. As these payments constitute a large share of Swiss farm income, they are important farm characteristics. Our second set of covariates includes structural characteristics of the farms, such as size (on average, 21 hectares, with 6 hectares of cropland and 15 hectares of permanent grassland, 27 standardized livestock units, and

two standardized labor units), the age of the manager (on average 50 years), and the farm's location (canton and growing zone, i.e., lowlands, hilly regions, mountainous areas) as well as farm types (e.g., dairy).

A third approach is to explicitly correct the differences in group dynamics by fitting group specific trends (Chabé-Ferret & Subervie 2013, Mora & Reggio 2019). The trend that should be fitted, for example, linear, quadratic, or another, can be selected by "exclusion" (Mora & Reggio 2019). The fundamental idea is that in the absence of group specific trends, we assume not only (conditionally) parallel trends but also (conditionally) parallel changes in trends, and changes in changes in trends, and so on. Invalid assumptions can be eliminated by estimating all possible specifications based on these assumptions and statistically testing whether they produce the same estimates. For example, the assumption of parallel trends can be viewed as invalid if estimates differ significantly when assuming parallel changes in trends. Mora and Reggio (2019) then recommend fitting group specific trends that cannot be rejected. The inclusion of linear trends result in estimate (4) instead of (2):

$$E[Y_{it}|D_i = D] = \delta + \sum_{\tau=2}^{T} \delta_\tau I_t^\tau + \gamma D + \gamma_T \times I_t^T \times D + \gamma_1 \times t \times D \tag{3}$$

The inclusion of higher order polynomials results in estimate (4) instead of (3):

$$E[Y_{it}|D_i = D] = \delta + \sum_{\tau=2}^{T} \delta_\tau I_t^\tau + \gamma D + \gamma_T \times I_t^T \times D + \sum_{r=1}^{R} \gamma_r \times t^r \times D \tag{4}$$

Here, $\gamma_1 \times t$ and $\sum_{r=1}^{R} \gamma_r \times t$ capture differences in pretreatment dynamics that persist over time. The fully flexible model incorporating all these dynamics then estimates (Mora & Reggio 2019):

$$E[Y_{it}|D_i = D] = \delta + \sum_{\tau=2}^{T} \delta_\tau I_t^\tau + \gamma D + \sum_{\tau=2}^{T} \gamma_\tau \times I_t^\tau \times D \tag{5}$$

The standard difference-in-differences set-up is a special case of this, with $\gamma_\tau = 0$ for all $\tau < T$.

The fourth approach is a partial identification approach, proposed by Rambachan and Roth (2019). The fundamental idea is to simulate how estimates are affected by deviations from the parallel trends assumption. Only approximately parallel trends can be assumed, rather than perfectly parallel trends, and it can be analyzed what is the largest deviation from parallel trends that would still not qualitatively change the estimates. It is easiest to illustrate this graphically. Figure 3 shows the assumption of parallel trends as a perfect linear line and how adding/subtracting M relaxes this assumption while still allowing identification of confidence sets (Rambachan & Roth 2019).

Our main results from combining the different matching and difference-in-differences approaches are shown below in Section 5, and a comprehensive overview of all estimates is shown in the Supplementary Materials (Figures S3–S5 and Tables S1–S10).

4.2 | Economic valuation of environmental outcomes from differently designed schemes

A benefit transfer approach is used to assess the societal benefits of the different estimated treatment effects (see Rolfe et al. 2015). We proceed by describing our approach in three consecutive steps: (1) definition of the context for ecosystem service valuation (Johnston & Wainger 2015); (2) estimation of willingness to pay for ecosystem services using a meta-regression analysis function transfer

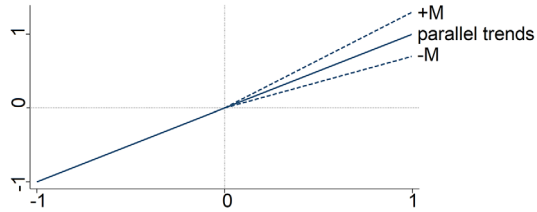

FIGURE 3 Relaxing the perfect parallel trends assumption. The linear line represent the parallel trends assumption, M shows assumed deviations

(Richardson et al. 2015), and (3) aggregation of estimates to quantify the empirically identified changes in ecosystem service values from areas in the action-based and results-based categories, respectively.

The context of our benefit transfer approach is that we combine our analysis of changes in biodiversity conservation practices with willingness to pay (WTP) estimates for ecosystem services from extensively managed grasslands. We use the recent meta-regression study by Huber and Finger (2020), in which the authors analyze the variation in reported willingness to pay for ecosystem services provided by grasslands as estimated in 79 individual European studies. The study uses a set of methodological and case study related variables to identify the effects of changes in grassland use intensities on the WTP. Another approach would have been to ask people directly to value the two schemes. This would have allowed to consider actual preferences of the citizens for action- and results-based payments, and make our valuation more consistent. In contrast, we here must assume that the public values the outcomes of the schemes rather than the schemes itself. The drawback of this approach is that the valuation of the two schemes is indirect via their impact on landscape and biodiversity. The economic valuation of these impacts come with many methodological challenges (Johnston et al. 2017). The advantage of our benefit transfer approach is that the valuation of the schemes is based on multiple studies, which still include context specific variables. Thus, given the context of our analysis, this meta-study is well suited to transfer the results from the regression analysis. We exploit two specific characteristics of the meta-study that are optimal for our purpose of valuing the benefits achieved with the different treatments.

First, we can use these estimates to predict the WTP for a shift from intensively to extensively managed grassland, which is the most important category in the biodiversity conservation program in Switzerland under the action-based scheme. This switch usually includes the appearance of flowers in biodiversity conservation practice (e.g. Lindemann-Matthies et al. 2011, Rewitzer et al. 2017). Thus, we also apply the estimated values from this meta-study to the other conservation practices, such as the biodiversity measures implemented on cropland, that is, wildflower verges and strips.

Second, we make use of the framing effect reported in the studies. Huber and Finger (2019) show that WTP estimates for extensively managed grassland are higher when the corresponding change is framed in the context of biodiversity conservation. More specifically, public willingness to pay increased by 18 Euros per person and year if the underlying valuation study framed the shift from intensive to extensively managed grassland in the context of biodiversity conservation rather than landscape maintenance.

This framing effect allows us to differentiate the estimated benefits from areas with extensive management but expected lower ecological quality (areas under the action-based scheme) from areas with observed ecological quality (species richness of areas under the results-based scheme). The assumption is that action-based measures create benefits by maintaining a diverse landscape and some biodiversity (i.e., with lower WTP), whereas the areas under results-based measures provide additional benefits from higher biodiversity conservation (i.e., with higher WTP). Furthermore, Huber and Finger (2019) show that the WTP for grassland ecosystem services is higher in mountain

regions or uplands compared to areas in the lowlands. As a specific example, in mountain areas, the willingness to pay per person and year for shifting from intensively to extensively managed grassland is 55 Euros higher than the same shift in the lowlands. This can also be applied in our benefit transfer approach to estimate the value of conservation areas under the action-based and results-based scheme. Thus, we use these two characteristics (framing and region) of the meta-regression model for our benefit function, which has the following linear form:

$$\widehat{WTP}_{ij} = \widehat{\beta}_0 + \widehat{\theta}_j x_j + \widehat{\vartheta}_i y_i + \sum_{g=1}^{G} \widehat{\mu}_g f_g + \sum_{k=1}^{K} \widehat{\beta}_k z_k \quad (6)$$

where WTP_{ij} represents the transferred willingness to pay estimates for ecosystem services from the action-based scheme and the results-based schemes in mountain and lowland regions, respectively. The first two terms after the intercept $\widehat{\beta}_0$ describe the framing variables in the meta-regression with x_j as a dummy variable for the framing, i.e., (i_1 = landscape maintenance/action-based and i_2 = biodiversity/results-based) and y_j as a dummy variable for region (j_1 = lowland and j_2 = mountains). The last two terms represent case study and methodological related variables in the meta-regression model where f_g is a set of variables g characterizing specificities of the case study (e.g., number and type of participants in the survey, such as general public vs. local inhabitants), and z_k is a vector containing k methodological variables characterizing the valuation studies (e.g., dichotomous choice vs. multinomial choice setting). The parameters $\widehat{\beta}_0, \widehat{\theta}_j, \widehat{\vartheta}_i, \widehat{\beta}_k, \widehat{\mu}_g$ were generated from the meta-regression model by Huber and Finger (2019) using an MM-estimator, a robust regression technique (Finger, 2010) and multiway clustering, which permits the adjustment of standard errors for variations of estimates in the same research article (Cameron et al., 2011).

We then estimate the $\widehat{WTP}_{ij}$ using the following specifications for the transfer function: (1) the case study specific variables f_g were either set to the mean value of the underlying data or adjusted to Swiss specific parameters for the year (i.e., GDP in the year of the study), (2) real expenditure per capita in purchasing power parity and the agri-environmental program (see Table S11 in the Supplementary material). Because methodological variables z_k in the meta-regression cannot be reasonably adjusted in a benefit transfer approach, we use these methodological parameters to calculate a range of predicted WTP estimates, whereby we distinguish between studies that used discrete choice methods and contingent valuation studies. This allows us to interpret our results independently of methodological uncertainties from underlying WTP estimates. In the main body of the text, however, we focus on the values from discrete choice methods only. The reason is that our estimates from discrete choice experiments are lower compared to those values from contingent valuation studies. This allows us to interpret our results as lower bounds for the calculations of return on investments. Results from different specifications of the WTP estimations are presented in the supplementary material (Table S12). Based on these assumptions, we estimate that the average willingness to pay for ecosystem services ranges between 15–75 for action-based payments and 35–95 Euros per person and year for results-based payments in the lowlands and mountain regions, respectively.

The third and final step in our benefit transfer approach is to specify the exact changes in ecosystem services from the identified change in action- and results-based areas, whereby baseline values must be quantified for ecosystem services from these areas. We aggregate the WTP estimates over the population in Switzerland (∼8.6 million persons) and divide it by the existing surface areas under the action-based and results-based schemes in the lowlands and mountain regions, respectively. In doing so, our baseline values reflect the current state of conservation areas under the action-based scheme in the lowlands (∼48,000 ha) and mountain regions (∼64,000 ha) as well as conservation areas under the results-based scheme in lowlands (∼20,000 ha) and mountain regions (∼54,000 ha). This allows us to derive the WTP per ha, which we can then combine with the treatment effects from our empirical analysis. Our final estimates of the economic value of one hectare of

additional conservation areas under the action- and results-based scheme amount to 2342 and 5580 CHF, respectively. This difference in value between conservation areas under the action- and results-based scheme emerges from three underlying sources: (i) the meta-regression estimates, (ii) the shares of the two conservation practices in the lowlands and the mountain regions, and (iii) the total extent of these areas. Please note that our approach for the aggregation assumes a constant benefit from additional biodiversity conservation areas. Although this is a strong assumption in theory, we have the advantage of being in a position to assess small, observable changes with respect to the baseline of existing biodiversity conservation areas in the lowlands and mountain regions (i.e., changes in areas and not scheme participation per se).

5 | RESULTS

5.1 | Treatment effect estimates

Before we estimated the difference in differences, we matched farmers based on their observable characteristics. We compared a range of matching techniques and assessed them based on their achieved covariate balance. All matching algorithms achieved a reasonable balance between the farmers who received either higher action- or results-based payments with the control group of farmers who received no increase in payments (see Appendix, Figure S1). However, many algorithms had trouble with balancing the group of farmers who received both higher action- and results-based payments with the control group of farmers who received no increase in payments. An algorithm that always achieved a good balance for all groups is coarsened exact matching (Figure 4).

Figure 5 shows the estimated average treatment effects on the treated for our three treatments (increased action-based payments, increased results-based payments, and both payments increased) and outcomes (extensive grassland, species-rich grassland, and cropland measures). For each combination, three estimates are shown: the estimate from standard difference in differences after matching (STD), the estimate from doubly robust difference in differences (Sant'Anna & Zhao 2020) (DR DID), and the estimate from fully flexible difference in differences after matching (Mora & Reggio 2019) (FF DID).In the majority of cases, the estimates of STD, DR DID, and FF DID are the same, but there are three cases with small differences: The treatment effect of having both payments increased on the area of extensive grassland is 0.91 ha when we use STD, 0.81 ha when we use DR DID, and 0.56 ha when we use FF DID. Similarly, for the treatment effect of having both payments increased on the area of species-rich grassland, the estimates are 0.88, 0.82, and 0.62 ha, respectively. Finally, for the increase in results-based payments and the area of cropland measures, STD and DR

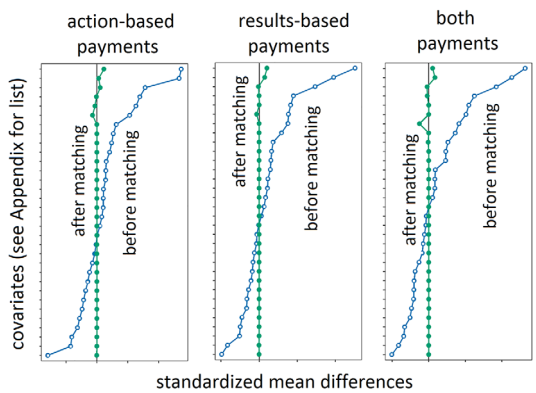

FIGURE 4 Covariance balance plots. Blue, hollow dots = before matching, green, full dots = after matching

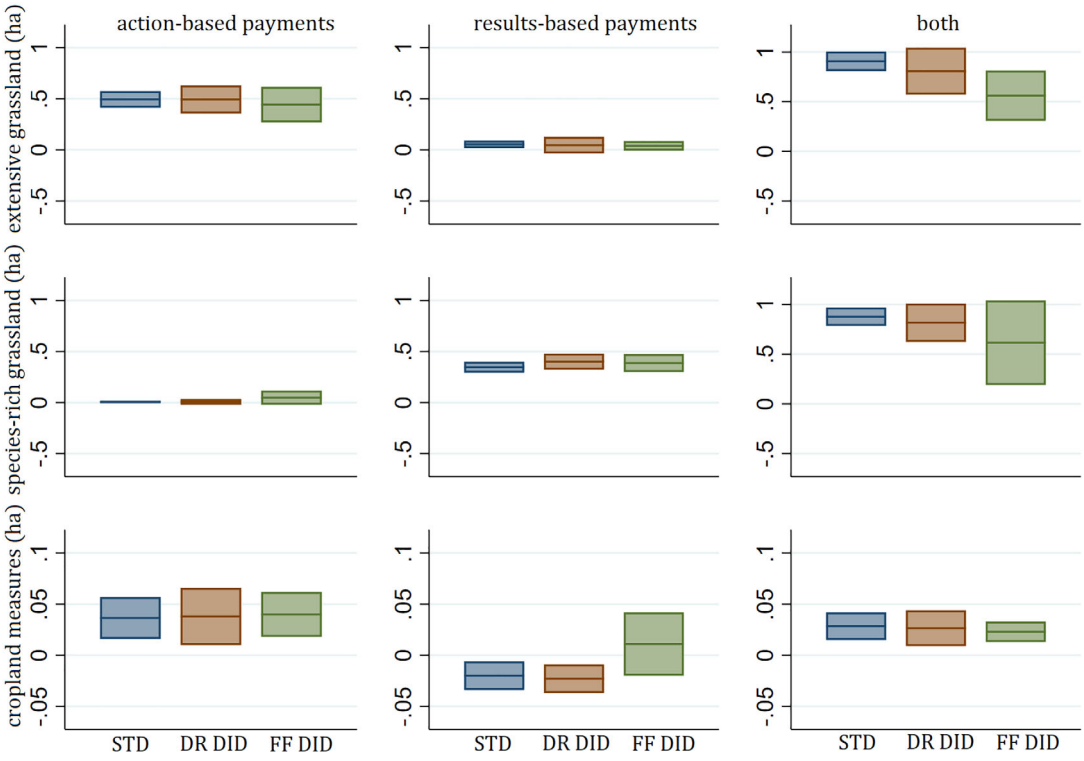

FIGURE 5 Estimates. Bars show 95% confidence intervals and their center the point estimates. STD, Standard Difference in -Differences after matching; DR DID, Doubly Robust Difference in Differences; FF DID, Fully Flexible Difference in Differences after matching. Estimates are ATT. For a table with all estimates, including the estimated standard errors, see Appendix, Table S1. For an analysis of heterogeneous treatment effects, see Figure S5 in the Appendix

DID each estimate a negative treatment of 0.02 ha, whereas FF DID indicates no effect (coefficient + 0.01). Comparing these estimates with the raw data (e.g., Figure 2), it seems that FF DID best corrects for diverging pretreatment trends in these cases. Thus, we rely on FF DID for our main estimates.

Our main estimates suggest that the increase in action-based payments increased extensive grasslands by 0.44 ha, and cropland measures by 0.04 ha, per treated farm and year. The increase in results-based payments increased species-rich grassland by 0.38 ha, per treated farm and year. Finally, the increase in both payments increased extensive grasslands by 0.56 ha, species-rich grasslands by 0.62, and cropland measures by 0.02 ha, per treated farm and year. Figure 6 shows the estimated counterfactuals together with the observed changes in the three treatment groups and the common control group.

5.2 | The role of the parallel trends assumption

We use the "honest approach to parallel trends" of Rambachan and Roth (2019) to understand the sensitivity of our results to deviations from parallel trends (Figure 7).

The plots show the estimates from standard difference in differences with matching in green (the estimate on the left in each plot) and the estimated effect range from increasing deviations from parallel trends in brown. In most cases, allowing for deviations from parallel trends only makes the estimates less precise. However, in the cases in which we saw some deviations between approaches in

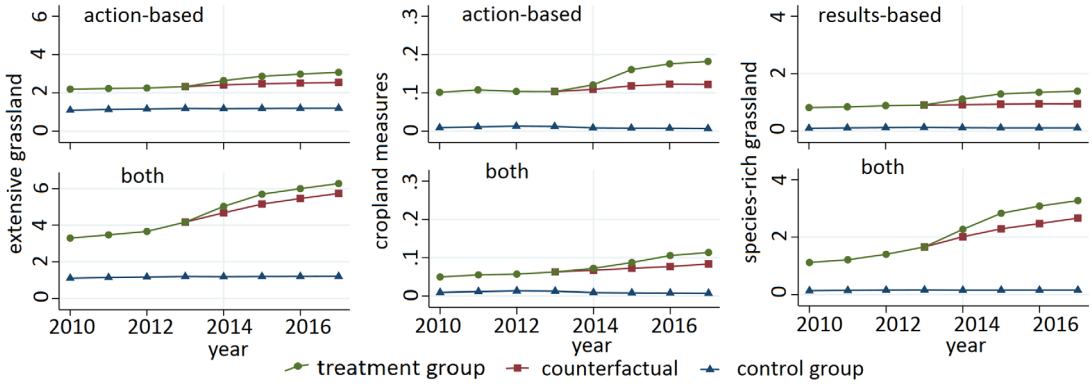

FIGURE 6 Counterfactuals. The figure shows the observed trends among treated (circles), controls (triangles) and estimated counterfactual trends for the treated (squares) in hectares per farm, for all treatments and outcomes that are statistically significant at the 5% level

the main-results section (see Figure 5), we see comparatively large changes in the estimates as a result of deviations from parallel trends. This is the likely explanation why STD and DR DID give slightly different results in these cases than FF DID, which produces the most conservative estimates because it controls for group-specific, pretreatment trends.

5.3 | Elasticities and the return on investment of action- and results-based payments

We estimate that the rise in payments effectively increased the areas under the two agri-environmental schemes. For the treatment group for which only the action-based payments increased, we find an average elasticity of supply of 0.6 (Table 3). This implies that a 1% increase in the payment led to an average increase of the amount of extensively managed grassland and biodiversity measures on cropland by 0.6%. The average elasticity is slightly higher in the treatment group for which both payments increased (0.9). For the results-based payment, we find an average elasticity of 1 in the treatment group for which only the results-based payments increased. Also here, the average elasticity is slightly higher in the treatment group for which both payments increased (1.2). In both cases, however, there is an overlap of our estimates, and we do not find significant differences in the elasticity between the treatment groups.

The average effectiveness varies between the two payment schemes. In the group with higher action-based payments, our results suggest that the government must spend 713 CHF to achieve one hectare more of extensively managed grassland or biodiversity measures on cropland. To preserve on hectare more of species rich grassland, the government must spend, on average, 1102 CHF. This is in line with the theoretical considerations that action-based payments should be less costly. However, the payment effectiveness is higher for both schemes in the group for which both payments increased. An additional hectare of biodiversity conservation area is bought for 327 and 535 CHF in the action- and results-based scheme, respectively. As in the case of the elasticities, however, the overlapping confidence intervals in our estimation imply that these differences between the action- and results-based payments are not statistically significant. This is important because some of the theoretical literature suggests that action-based schemes could be implemented at lower costs for the government than results-based schemes. Our empirical findings do not support this claim because we find no significant difference in the effectiveness of action- and results-based payments.

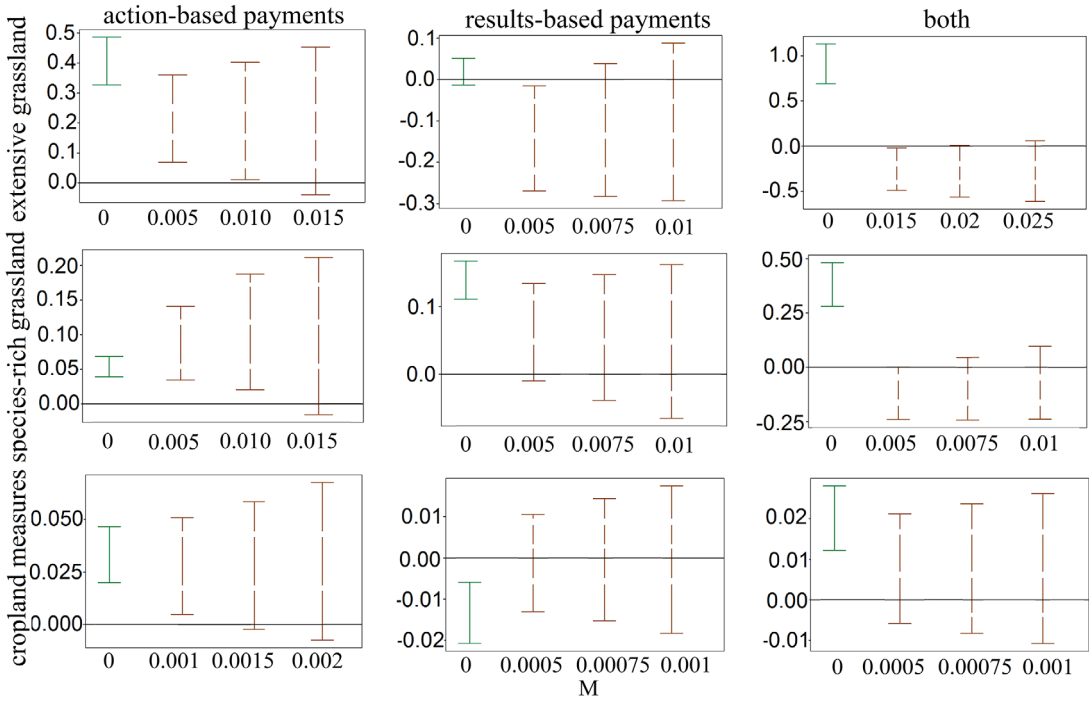

FIGURE 7 Estimates' sensitivity to deviations from parallel trends. The green, full bars show the estimated ATT from a combination of Mahalanobis matching and difference in differences based on the assumption of perfect parallel trends. Brown, dashed bars show how deviations by ±M from perfect parallel trends would change the estimates

Although the government needs to spend more for increasing the area of species rich grassland, the average windfall effects are higher for the areas under the action-based scheme. An increase in action-based payments created windfall effects between 700 and 800 CHF per farm, whereas the results-based scheme led to average windfalls of 400–470 CHF.

The average return on investment from the two payments schemes is above one, and thus the increase in both payments is estimated to have increased overall welfare. This suggests that the increase in payments in both schemes is economically meaningful. In addition, our benefit transfer approach suggests that the value of species rich grassland (5580 CHF/ha) is much higher than the value of merely extensively managed grassland, that is, the area enrolled in the action-based scheme (2342 CHF/ha). Thus, the estimated return on investment of the results-based payments is three times the estimated return of the action-based payments. The scale of the benefit from species rich grasslands compared to less intensively managed grassland implies a high likelihood that the return on investment is indeed higher for the results-based compared to action-based payments. Because the requirements for receiving results-based payments consider the regional environmental characteristics of the area, our analysis suggest that this difference results from the payment and not regional differences in biodiversity potentials (see regional distribution of payment increases in Figure S2).

6 | DISCUSSION AND CONCLUSION

Agri-environmental schemes are an increasingly important policy tool to develop more sustainable farming systems worldwide. The existing theoretical literature suggests that results-based schemes could be more cost effective than action-based schemes. Although there is empirical evidence on the

TABLE 3 Estimated returns on investment

Treatment group			Increase in action-based payments	Increase in results-based payments	Increase in both payments	
			6127	4789	13,588	
Number of treated farms	Formula	Unit	Action based	Results based	Action based	Results based
Average treatment effect on the treated farmers (95% CI)	γ_T	ha	0.48 *(0.29–0.68)*	0.40 *(0.3–0.48)*	0.58 *(0.32–0.85)*	0.62 *(0.19–1.04)*
Counterfactual, that is, estimated area without payment increase	Y_{0i}	ha	2.39	0.91	3.74	1.40
Average change in payments per ha	Δp	CHF per ha	342	441	189	332
Initial payment per ha	P^0	CHF per ha	1055	1005	1045	879
Elasticity	$\frac{(\gamma_T \times P^0)}{(Y_{0i} \times \Delta p)}$	Ratio	0.6 *(0.4–0.9)*	1.0 *(0.8–1.2)*	0.9 *(0.5–1.3)*	1.2 *(0.4–2)*
Effectiveness, that is, average payment for one ha more	$\frac{p}{\gamma_T}$	CHF per ha	713 *(504–1193)*	1102 *(1475–2321)*	327 *(386–590)*	535 *(515–1747)*
Average windfall	$\Delta p \times Y_{0i}$	CHF per farm	819	400	708	464
Average change in total payments per farm	$C = \Delta p \times Y_{1i}$	CHF per farm	1113	613	1210	1086
Benefit	$B = \widehat{WTP}_{CE} * \gamma_T$	CHF per farm	1124 *(672–1590)*	2232 *(1668–2650)*	1358 *(752–1984)*	3460 *(1060–5803)*
Benefit–cost	$\frac{B}{C}$	Ratio	1.0 *(0.6–1.4)*	3.6 *(2.7–4.3)*	1.1 *(0.6–1.6)*	3.2 *(1–5.3)*

Note: γ_T, Y_{0i} are econometric estimates, $P^0, \Delta p, Y_{1i}$ can be observed in our data. $\widehat{WTP}$ is derived from benefit transfer approach. Our estimates imply a value of 2342 and 5580 CHF per additional ha under the action-based and the results-based scheme, respectively. Numbers in brackets for elasticity, effectiveness, benefit, and benefit–cost show the range of our calculations using the 95% confidence intervals (CI) from our main estimation keeping the average change in payments per ha constant. Sensitivity analysis of benefit–cost ratios are provided in the Appendix.

effectiveness and return on investment of individual agri-environmental instruments, we currently lack empirical evidence on the comparative performance of action- and results-based payments, individually and in combination. Here, we take advantage of a sudden, major reform of the Swiss biodiversity payment system in 2014. As a result of this reform, many farmers received higher action- or results-based payments, or both, although some did neither. This quasi-experiment allows us to estimate average treatment effects on the treated for each payment increase using difference-in-difference approaches. We moreover combined our econometric analysis with a benefit transfer approach to calculate return on investments for the differently designed agri-environmental schemes and their combination.

We find that—on average—all payments improved the intended outcomes; that is, extensively managed grasslands and biodiversity conservation areas on croplands were increased by higher action-based payments, and species rich grasslands were increased by higher results-based payment. A 1% raise in the payment led to an average increase of these areas between 0.6% to 1.2%. Although, on average, the government thus has to spend less to increase the biodiversity areas under the action-based scheme, the average windfall effects are lower in the results-based scheme. Overall, our results suggest that there is no significant difference in the effectiveness of the two schemes with respect to their intended outcomes and that the combination of differently designed agri-environmental schemes must not necessarily diminish the performance of the individual schemes.

Based on our benefit transfer approach, we then find positive return on investments for all parts of the policy reform. Given our result that species rich grassland is more valuable to the public than the extensively managed grassland and cropland measures delivered with action-based scheme, our findings suggest that the estimated return of investment of the results-based payments is threefold compared to the action-based payments. This reflects the higher societal value of species rich grasslands. Overall, thus, our findings are consistent with the theoretical claims that the results-based payments are more cost effective than the action-based payments.

A limitation of our study is that our outcome variables are changes in areas with categorically different biological value, and we do not have direct biodiversity measures. We must acknowledge that using area changes is only a proxy of actual biodiversity outcomes. Another limitation is the uncertainty of the estimated benefits for the achieved outcomes. We use a benefit transfer from a recent meta-analysis on the public's willingness to pay for the ecosystem services provided. Although our contribution explicitly accounts for the methodological uncertainty arising from the underlying willingness to pay studies, these estimates are more uncertain than our estimated treatment effects. A misspecification of the beneficiaries, for example, that is, the population over which WTP estimates are aggregated, can have a much larger effect than a misspecification of the value itself (Johnston & Wainger 2015). Thus, the available economic valuation data still contain uncertainties, and our approach could be further improved by more specific valuations, for example, by directly focusing on the schemes in question rather than the general valuation of landscapes and biodiversity. The combination of such data with measurement of the effective changes in flora and fauna species would be a crucial next step but an expensive one that can likely only be implemented at a more modest scale.

Our empirical findings have important implications for the optimization of agri-environmental programs and making agricultural policy more sustainable (Pe'er et al. 2019, Pe'er et al. 2020). First, we conclude that raising the budget for existing agri-environmental measures effectively leads to an increase of the areas allocated to biodiversity conservation and that the resulting benefits are higher than the corresponding costs in both schemes. To further improve their performance, however, it might be advisable to focus on the windfall gains they create and to mitigate them, for example, by using an auction mechanism instead of a uniform raise in payments (e.g., Banerjee 2018). Second, although on a per-hectare level, action- and results-based payments by and large performed comparable in our analysis, the average return on investment is much higher for results-based payments. Thus, our findings support the theoretical claim that, if the program's budget allows, results-based payments should be prioritized.

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SUPPORTING INFORMATION

Additional supporting information may be found in the online version of the article at the publisher's website.

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