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Agri-environmental schemes for biodiversity and environmental protection: how were are not yet “hitting the right keys”

1. Introduction

Agri-Environment Schemes (AES) have been mainstreamed in agricultural policies across the globe as a means to financially incentivise farmers to undertake nature-protecting activities and to support rural development and mitigate environmental damage (Prager, 2015). At their core, they aim to compensate land managers for additional costs and income foregone incurred in abiding with higher environmental and ecological quality standards.

AES focusing in enhancing the environment and organic farming were formally introduced in the European Union (EU) in 1985 as part of the Agricultural Structures Regulation (Batary et al., 2015). Since they became compulsory for all European Union Member states in 1992 (ibid), AES have been progressively reinforced through the various reforms of the Common Agricultural Policy (CAP), especially that of 2009 (Wynne-Jones, 2013), and can now be considered to be very solidly established in European environmental policy. Over a quarter of all Utilised Agricultural Area (UAA) in 2013 in Europe was under some type of AES, while for 2020 the target was that 22.5% of all UAA (44 million hectares, Eurostat, 2017) is enrolled in AES. These figures vary greatly across countries, though. For example, Finland had more than 93.7% of its UAA enrolled in AES in 2013 while Greece had only 1.8%. Between 2007 and 2013 €23 billion were devoted to AES in the EU, rising to €25 billion for the current period of 2014-2020 (European Union, 2013). Acknowledging the need of countries to manage their agricultural expenditure according to their specific priorities, in the current CAP the EU also allowed up to 15% of total allocated funding to be transferred between pillars (European Union, 2013). This can push a country's AES expenditure up to 30% of the total CAP expenditure, the highest amount in EU history.

With each CAP reform came changes in the design and requirement of AES, with the most recent ones emerged as part of the Rural Development Plans (RDP) in 2013, (Regulation No 1305/2013 Article 28, Measure 10). The RDPs were primarily focused in the preservation of biodiversity and protection of natural resources (European Network for Rural Development, 2015). Under biodiversity safeguarding, RDPs included wetland protection, protection against invasive species, conservation of priority species and grassland restoration, amongst others. Preservation of natural resources refer to protection from soil erosion, improvement of water quality and reduction of fertilizer and pesticide use. Overall, 70 RDPs had a biodiversity-related objective, 67 had a landscape-preservation objective and 64 a water quality-enhancement objective. This reinforced policy focus on biodiversity protection awaken the academic interest and led to the publication of several studies that examined farmers' preferences for existing or hypothetical AES. These studies have focused on compensation offered to farmers participating in AES, the characteristics of the contracts and the effect that farmers' sociodemographic characteristic or the geographical characteristics of the area had on acceptability of the schemes (e.g. Schulz et al., 2014).

Despite their 35 years of history, AES in the European Union seem to have had limited success in providing environmental benefits (Uthes and Matzdorf, 2013). Reasons behind this limited success are varied and of different nature. While a large part of this would be related to ecological effectiveness (European Environment Agency, 2015; Hellerstein, 2017), some of this limited success has been explained on an economic basis. For example, the fixed nature of economic costs (such as the procurement of specialist equipment or advice before enrolling in a contract) has been found to act as a barrier to entry for small farmers (Ducos et al., 2009), while implementation and administration costs and payment uncertainty after enrolling in a scheme have also been shown to impede adoption (Bartowski et al., 2021). Such economic barriers also revolve around the changing or abandoning of past land use activities which results in higher opportunity costs for land managers (Schou et al., 2020). Other reasons seem to be related to wrongful applications, by which countries may have implemented AES that are best suited for contexts with different climate than their own (Batary et al., 2015), limiting AES success-on-the-ground. Other reasons are related to farmers' preferences. Dessart et al., (2019) suggest that dispositional (e.g. environmentalism, risk perceptions, etc.), social (e.g. inter-personal relationships) and cognitive (e.g. knowledge and competences) motivations affect farmers' adoption of environmental schemes. Farmers have also been found to prefer simple but sometimes less effective AES that result in limited biodiversity support (Dicks et al., 2013; Villanueva et al., 2015). Participating in low-risk, low-result schemes might not be enough to produce necessary changes in the agricultural landscape, (Dobbs and Pretty, 2008). For example, simple schemes in the Netherlands do not seem to have achieved high levels of biodiversity conservation or restoration (Kleijn et al., 2004) while conservation statuses of protected habitats show declining biodiversity rates in areas under AES (European Environment Agency, 2015). Also, biodiversity-enhancing AES have been found to increase adoption and knowledge of good farming practices (Herzon and Mikk, 2007; Okumah et al. 2018) which should be an incentive for farmers to enrol to biodiversity-supporting AES, nevertheless, land managers have been found to be reluctant to change their pre-existing farming practices (Fleury et al., 2015; Matzdorf and Lorenz, 2010). Finally, sociodemographic characteristics such as farm size, farmer age and previously received training also affect adoption of AES (Villanueva at al., 2017).

At the heart of some of these reasons are the factors that make farmers more or less inclined to adopt the schemes and accept compensation for the delivery of conservation or environmental features. There is a body of empirical studies that have addressed this matter in specific locations or across countries by surveying land managers preferences for enrolling in such AES schemes, based on the neoclassical economics notion of willingness to accept (WTA) monetary compensation for participation (e.g. Villanueva et al., 2015; Hasler et al., 2019). However, there has not yet been a systematic inspection of this evidence which can draw an overall picture to inform further policy design into more effective and successful accomplishment of AES' environmental goals. In this paper we present such a review in the form of a systematic review and meta-analysis of primary valuation studies addressing farmers' WTA participation in AES on the basis of biodiversity or environmental features. While qualitative reviews exist (e.g. Lastra-Bravo et al., 2015), to our knowledge this is the first systematic quantitative analysis. We focus both on farmers and foresters as they are both eligible to enrol in AES and, we look at AES contractual characteristics such as length of contracts offered, scientific support to land managers and monitoring of results, as well as the socio-demographic characteristics of land managers surveyed, climatic conditions and how widespread is the use of AES in agriculture in the country of the study. We first draw a descriptive picture of this evidence and we then look for the effect that these factors have in farmers' WTA monetary compensation from such schemes using a meta-regression function.

We focus on European studies as it allows deriving multi-country comparisons which share a common framing, i.e. that of the CAP. Having a narrow scope also allows for higher confidence when choosing a measurable indicator variable in meta-analyses. It should be noticed, however, that similar AES policies have been implemented in other contexts. For example, the Australian Environmental Stewardship Program uses market-based incentives for farmers to achieve restoration and rehabilitation of biodiversity since mid-2007 (Ansell et al., 2016). Also, in the United States, monetary incentives have been given to farmers since 1985 to retire land from production for environmental conservation policies (Stubbs, 2014). Understanding whether the current design of AES and compensation offered to farmers is enough for them to support such schemes should inform the future design of AES, especially in light of the renewal of CAP in 2021 and face to the new challenges brought by Brexit (European Commission, 2018). Despite this European focus, results are expected to be of broader relevance since the notion of AES is widespread across the world (either under this or other framings (Schomers and Matzdorf, 2013)), with biodiversity-oriented AES being the most common globally (Ansell et al., 2015). Furthermore, AES are key in local actions of land managers that either directly or indirectly align with most land-related Sustainable Development Goals (SDGs) (Mann et al., 2018), as they focus on the strengthening of rural farming (SDG2), promote sustainable means of food production that improve resource efficiency (SDG12) and nature and biodiversity protection (SDG15). Therefore results from this study should also be useful in designing strategies to contribute to achieving these goals.

2. Methodology

Primary valuation studies of farmers and foresters' preferences to accept compensation for participating in AES have so far mainly focused on results-oriented schemes (e.g. Villanueva et al., 2015; Birge et al., 2017). In these, land managers receive compensation for delivering certain environmental services and benefits to the public (Burton and Schwarz, 2013), relating to early notions of agriculture multi-functionality (Huang et al., 2015) and more recently to provision of public goods and ecosystem services from agriculture (Schaller et al., 2018). The studies focus on contractual aspects such as whether offering monitoring of results or scientific and farm advisor support affects their willingness to participate in the scheme (e.g. Espinosa-Goded et al., 2010, Hasler et al., 2019); or whether length of contract duration has any effect on acceptability of the scheme (e.g. Santos et al., 2015). The AES studied in this literature are a mix of elements from actual existing AES contracts and from hypothetical schemes that researchers believe would be more appropriate or applicable to the needs of land managers.

A common feature in these studies is the way they conceptualize the environmental features of the AES in the form of set-aside land for environmental purposes. While the framing of such set-aside land varies across studies, i.e. by presenting it as afforestation (for recreation purposes or biodiversity increase), biodiversity offsetting, ecologically focus areas (EFA), buffer or riparian strips/zones, or protection from grazing, in all of them land managers are offered some contractual options to mitigate the loss of income occurred from setting the land aside. Some studies focus solely on such set-aside options (e.g. Villanueva et al., 2015, Santos et al., 2015), while others included it amongst other features such as environmentally friendly land management practice (e.g. use of biological fertilizer in Latacz-Lohmann and Breustedt, 2019). The loss of income from set-aside land impacts the economic welfare of land managers and compensation needs to be offered to incentivise them to enrol in such schemes. This compensation is what these studies measure using the notion of WTA, obtained via surveys using so-called stated preferences techniques (e.g. Chèze et al., 2017). WTA in neoclassical economics is the monetary amount that an individual is willing to receive as compensation for a certain loss of welfare that would restore welfare to its previous level

(Hanemann, 1991). In the context of this paper this refers to the loss of farm or land rent income due to taking land out of production to deliver biodiversity and environmental benefits, as well as transaction and implementation costs the land manager incurs as part of that process. In other terms, land managers are awarded compensation for delivering a public good (biodiversity enhancement or the protection of environmental features) while initially incurring some loss of private income for not producing other privately-sold goods (such as agricultural product or timber). The underlining assumption, rooted in welfare economics theory, is that survey respondents (in this case, land managers) are going to behave rationally, maximizing their utility and without displaying strategic behaviour when stating their WTA (del Saz-Salazar et al., 2009). It should be noted, thought, that land managers' welfare can be affected by non-monetary aspects such as social dimensions (inclusion/exclusion from a group of fellow land managers if you participate or not in a scheme) and benefiting from maintaining a certain environmental status of a public good (Kuhfuss et al., 2016; Skuras and Tyllianakis, 2018).

Focusing on surveys asking land managers for their WTA to have land taken out of production for to support biodiversity or environmental features allows us to collect a relatively uniform measure of economic welfare in the same format across studies (Euros per land manager, per hectare of land, per year) and to look for factors that may influence WTA. The use of welfare measures, such as WTA, has been shown to qualify as effect sizes for meta-analysis (Bateman and Jones, 2003).

2.1. Literature search

Searches were carried out in both ISI Web of Science and Google Scholar. While Google Scholar has received criticism (Boeker et al., 2013) it was considered appropriate in this case since it was anticipated that the literature would be small and hence the search attempted to also gather grey literature. However, searches in both databases produced the same results. A first search using as primary keywords the terms "Agri-Environment Scheme" and "Willingness to Accept" produced 111 documents. Additional searches in all search fields in

ISI with the two aforementioned keywords and a combination of each of the following secondary terms [“farmer(s)” OR, “land manager(s)” OR, “biodiversity” OR, “set-aside” OR, “ecologically-focus area” OR “afforestation”] were also conducted. During this search we explicitly excluded papers that referred to the schemes exclusively as payments for ecosystem services (PES) and did not (also) explicitly use the terminology “agri-environment schemes”. The reason for doing so is because the very definition of PES is currently being contested and it is notoriously fuzzy (Martin-Ortega and Waylen, 2018; Wunder, 2015). By doing so, we remain aligned to how the original studies defined the schemes they were analysing. This search resulted in 79 studies.

These 79 studies were filtered based on whether they provided key statistical information (sample size and measures of statistical variability such as standard errors or deviation or P-values) and what was considered as the effect size (mean WTA over the sample size). Studies that did not include one of these estimates were excluded from the analysis as not meeting sufficient quality criteria (e.g. Liznin et al., 2015) and this resulted in 27 studies being retained. The search was initially not geographically restricted. However, the vast majority of studies identified were carried out in Europe, probably due to the effect of AES being prominent in EU under CAP (only five studies reporting on Australia, Taiwan, Kenya and Uganda appeared in the list). The meta-analysis regression was subsequently undertaken only for the EU studies (excluding non-EU countries such as Switzerland and Norway), given the majority of the studies took place there and the contract design was more similar across those surveys. This led to a total final number of 20 primary studies. When studies reported several estimates over different subsamples, these were considered as separate WTA observations and reported accordingly (e.g. Hasler et al., 2019). This led to a total final number of 26 distinct WTA observations out of the 20 studies.

In the literature both negative and positive WTA estimates exist for biodiversity or environmental features enhancing contracts. This requires some attention since it has critical implications for the regression of these values. Positive WTA corresponds to the welfare measure as explained earlier, i.e. the amount of money that the land manager would accept to maintain their welfare level in compensation for the loss of income due to lack of production in the set-aside land. Negative WTA cannot, however, be interpreted instead as willing-to-pay (WTP) in this particular context. While WTP is another measure of welfare change in neoclassical economics terms, it is incongruent when referring to farmer's enrolment in AES, as it would mean that land managers are actually willing to pay to incur a cost or forego some income to set-aside land from production (or to put in economic terms, confound consumer surplus and equivalent surplus (Hanley et al., 2009)). This would mean that implementing the contract has no net costs to the land managers (i.e. there are no trade-offs). Therefore, negative WTA should be interpreted instead as an indication that the suggested options to respondents in the stated preference survey were not appealing enough or that respondents' interests were fundamentally against engaging in the delivery of biodiversity enhancing and protecting environmental features (what in environmental economics is referred to as a 'protest' response (Hanley et al., 2009)). This can be partially explained by the range of compensation land managers are offered in primary valuation studies, with small ranges increasing non-participation or protest responses (Villanueva et al., 2017). For the purpose of the present systematic review, we keep both type of studies, i.e. the ones reporting positive WTA as well as negative WTA, but we distinguish between them in the analysis. This leads to two regression models: one with the full set of WTA estimates (N = 26) and one only for the positive and statistically significant WTA estimates (N =13, labelled 'best set' of studies). We then discuss the implications of the differences amongst these two models.

Income is expected to negatively impact WTA, as dictated by the theory of diminishing marginal utility (Groothius et al., 1998). Information on income is traditionally expected to be collected in stated preference surveys but past meta-analyses have shown it is not always the case (e.g. Tyllianakis and Skuras, 2016). To have a uniform income measure and overcome potential missing information from studies on farmers' income, we use income data for the European level provided in the European Commission's database Farm Accountancy Data

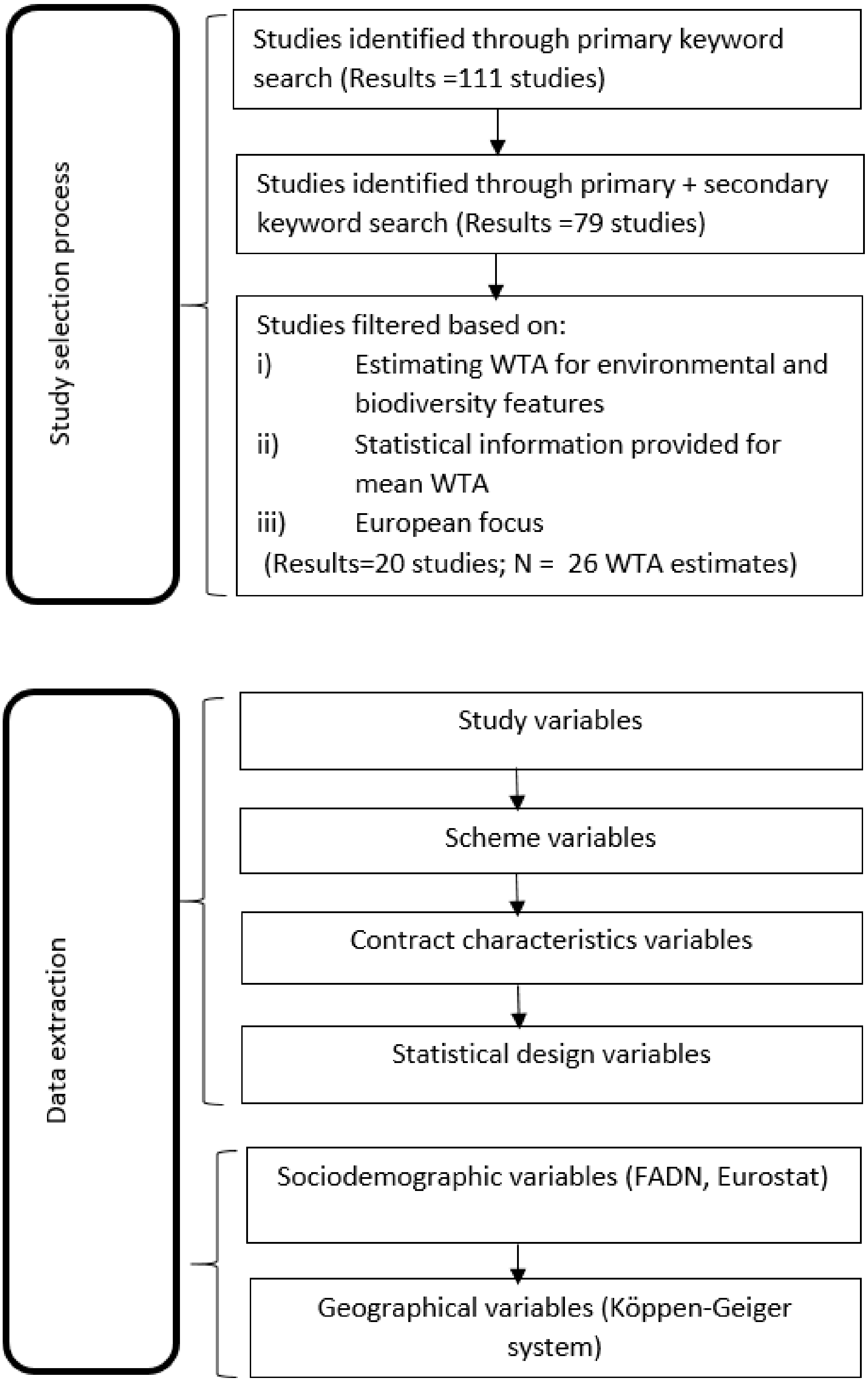
Network (FADN) (EU Open Portal Data). As the latest FADN data available end in 2017 the income estimates for studies carried out in 2018 and 2019 were adjusted for inflation on 2019 prices[1].

The extent of land under production that a manager has is expected to influence WTA as such land is the only one that can be enrolled in AES (European Commission, 2020). As with income, only few primary studies were found to collect such information (e.g., McGurk et al., 2020). As an approximation of land size, farm size in the FADN data on Utilised Agricultural Land (UAA) at the country level were used. UAA estimates were assumed to be identical with 2017 estimates for studies conducted after that date. Finally, as the distribution of studies was geographically diverse and AES have been found to be influenced by climate in Europe (Eurostat, 2017), the climate where the study was carried out was also recorded by classifying countries under the Köppen-Geiger classification system (Kottek et al., 2006).

The full literature search and information extrapolation process can be seen in Figure 1, and the full list of studies is found in Appendix 1.

Figure 1. Flow diagram of the study selection process and information collected

[1] It could be argued that income needs to be corrected by purchasing power across the different countries in the database, however, this was not feasible as the FADN database does not provide estimates in such terms. Past studies (e.g. Tyllianakis and Skuras, 2016) that have used several income estimates as explanatory variables in meta regressions have not found differences between using different income measures as determinants of the effect size (i.e. WTA in our case). The only significant differences in results in such analyses was when using the stated income from participants in primary surveys, compared to income estimates from public or European databases. As the primary studies in our meta-analysis did not report income from the studies (only 3 of them did) we did not use such an indicator as regressors.



2.2. Evidence overview and model specification

The analysis of the evidence occurs it two phases: first we provide an overview of the evidence based on simple descriptive statistics. This includes evidence across the following categories (see also Figure 1):

i. Study variables (referring to year of study, sample size, type of stated preference method and type of land managers targeted)
ii. Scheme variables (referring to whether schemes were described explicitly as AES to survey respondents and whether the schemes targeted explicitly biodiversity or whether they targeted environmental features more generally)
iii. Contract characteristics variables (referring to the contract options offered to land managers having lengthier contracts or the requirement of monitoring of results)
iv. Statistical design variables (referring to statistical information regarding WTA, such us positive or negative WTA)
v. Sociodemographic variables (e.g. farm size, age and income)
vi. Climatic variables (referring to the climate of the area where a study took place)

Secondly, we developed a meta-analytical model, using mean WTA to participate in the AES as the dependent variable. Using stated preferences measures such as WTA in a meta-analysis is common in the environmental economic literature (Tyllianakis and Skuras, 2016; Penn and Hu, 2021). Mean WTA of farmers to participate in AES for protecting biodiversity or enhancing of environmental land features within the framework defining AES can be considered a consistent and comparable effect size across studies as it requires a certain portion of land to be set-aside for such purposes, across all studies. As studies are collected from various countries from studies offering different contract options to farmers, as well as the fact that we include both farming and agroforestry focused studies, heterogeneity between studies is expected. Therefore, the heterogeneity of WTA across studies (*i)* can be expressed through θ, the grand population mean, the distance from the grand mean u_i and the sampling error component ε_i:

$$WTA_i = \theta + u_i + \varepsilon_i \quad (1)$$

with $u_i \sim N(0, \tau^2)$ and $\varepsilon_i \sim N(0, \sigma_i^2)$ while τ^2 denotes the true variation between studies. In order to further examine the drivers of heterogeneity between studies the use of meta-regression is advised (Tyllianakis and Skuras, 2016). We also account for possible heterogeneity from having multiple WTA estimates from single studies.

As such, WTA follows a normal distribution around a linear predictor $WTA|\theta_i \sim N\left(\theta_i, \sigma_i^2\right)$ where $\theta_i \sim N(x_i\beta, \tau^2)$ and Eq.(1) becomes:

$$WTA_i = x_i\beta + u_i + \varepsilon_i \tag{2}$$

with $u_i \sim N(0, \tau^2)$ and $\varepsilon_i \sim N(0, \sigma_i^2)$ and x is a vector of covariates that can explain WTA.

By depicting income and WTA in a logarithmic form as part of the socio-economic characteristics in Eq.(2) we are able to estimate also income's elasticity for the WTA for biodiversity and environmental areas' features, i.e. how much does WTA increases or decreases by a 1% decrease or increase of the farmer's income (Villanueva et al., 2015). This serves as an indication of how sensitive WTA is to land manager's income. If the elasticity of WTA is greater than 1 then more wealthy land managers would require higher compensation than less wealthy farmers to participate in biodiversity or environmental features-protecting AES, having interesting implications in terms of policy effects.

3. Results

3.1. Overview of existing evidence

The descriptive statistics from the observations from the studies can be found in Table 1, following the structure presented in Figure 1.

Variable name	Description	Frequencies (N=26)	Mean (Max, min and st. deviation) N=26
Study variables			
Study year	Year that the data were collected	-	Mean: 2014 Min-max: 2008-2019, s.d (3.78)
Sample size	The sample size recorded for each study area (i.e. number of land managers interviewed)	-	Mean: 310.68 Min-max: 27 – 1027 s.d. (263.94)
Choice Experiment	Categorical variable; 1 if the study used the Choice Experiment method to derive values; 0 if it used the Contingent Valuation method	0=1 1=25	-
Land management type	Categorical variable; 1 if the study focused on farmers; 0 if it focused on foresters	0=3 1=23	-
Scheme variables			
Scheme presented as AES	Categorical variable; 1 if the schemes were explicitly referred to as AES to land managers;0 otherwise	0=5 1=21	-
Scheme focused on biodiversity	Categorical variable; 1 if the study's focus was on biodiversity improvement schemes; 0 otherwise	0=15 1=11	-
Scheme focused on environmental features	Categorical variable; 1 if the study's focus was on enhancing environmental features; 0 otherwise	0=6 1=20	-
Contract characteristics variables			
CAP specific	Categorical variable; 1 if the study's goal was to inform CAP; 0 otherwise	0=14 1=12	-
Support offered	Categorical variable; 1 if the study's schemes offered support to farmers; 0 otherwise	0=18 1=8	-
Long contracts	Categorical variable; 1 if the study's schemes offered contract periods longer than 5 years; 0 otherwise	0=13 1=13	-
Monitoring	Categorical variable; 1 if the study's schemes offered monitoring of results; 0 otherwise	0=20 1=6	-
Statistical design variables			
Negative WTA	Categorical variable; 1 if the WTA had a negative sign; 0 otherwise	0=13 1=13	-

WTA significant	Categorical variable; 1 if the WTA was statistically significant; 0 if insignificant or not reported	0=5 1=21	-
Sociodemographic variables			
Age	Mean age of the land manager (9 observations)	-	Mean: 50 Min-Max: 43 -62, s.d. (6.62)
Gross farm income	Gross farm income in the country of the study in Euros (in 2019 values)	-	Mean: 86873 Min-max: 16466-220783 s.d. (59669)
Net farm income	Net farm income in the country of the study in Euros (in 2019 values)	-	Mean: 203334 Min-max: -53517 – 63917 s.d. (26756)
Utilised Agricultural Area	Utilised Agricultural Area according to FADN, in hectares	-	83.67 s.d. (95.63)
Climatic variables			
Mediterranean	Categorical variable; 1 if the area's climate corresponds to Mediterranean as per Köppen-Geiger's classification, 0 otherwise	0=18 1=8	-
Temperate	Categorical variable; 1 if the area's climate corresponds to Temperate without dry season and warm summer as per Köppen-Geiger's classification, 0 otherwise	0=21 1=5	-
Continental	Categorical variable; 1 if the area's climate corresponds to Temperate continental climate as per Köppen-Geiger's classification, 0 otherwise	0=3 1=23	-
Cold	Categorical variable; 1 if the area's climate corresponds to Cold, without dry season and with cold summer as per Köppen-Geiger's classification, 0 otherwise	0=23 1=3	-

Table 1: Descriptive statistics of evidence collected from the 26 studies

Results show that WTA studies on AES in Europe cover a period from 2007 to 2019, which also coincides with the introduction and evolution of AES in the farming and forestry fields as it has been established by the previous two CAP periods (European Parliament, 2020).

Twenty-three studies surveyed farmers (including sheep and beef farmers and crop farmers) whereas only three studies surveyed forest owners. Only five studies referred to the AES also in terms of PES and the majority (21 studies) referred explicitly to AES only and did not confound them with other types of framings. With regards to the methods used to measure WTA, 25 studies used the choice experiment (CE) method and only one study used the contingent valuation (CV) method (see Adamowicz et al., (1998) for a description of the differences between the two).

Forty-four percent of the studies had biodiversity increase or protection as an explicit goal of the AES, while considerably more (77%) had AES focusing on some type of protection or enhancement of environmental features (i.e. not explicitly focusing on biodiversity). In more than half of the studies (65%), WTA estimates focused explicitly on some type of set aside land, either generically (e.g. Villamayor-Tomas et al., 2019) or as Ecological Focus Areas (e.g. Villanueva et al., 2015). The rest of the studies included set-aside as a by-product of environmentally-friendly ways of land management through, e.g. adopting nitrogen-fixing crops (e.g. Espinosa-Goded et al., 2010), vulnerable ecosystem protection (e.g. Czajkowski et al., 2019) and afforestation (e.g. Tyrväinen et al., 2020). The majority of studies (69%) presented contracts to land managers that also included more features apart from the set aside land, such as fertilizer use and flexibility over the area enrolled in the scheme (e.g. Beharry-Borg et al., 2009; Espinosa-Goded et al., 2010).

Thirteen studies in our review reported negative WTA which, as previously explained, is to be interpreted as either an indication that the offer presented in the valuation survey was not appealing enough to the respondents or as representing a form of protest against the valuation exercise. Either case calls for caution in the interpretation of the results derived from these studies for informing policy design, as it will be discussed. Overall, thirteen studies (50% of studies) reported positive mean WTA while WTA being statistically significant which were then used here as a "best-set" of variables in the regression.

The studies also focused on a variety of contract options and three types of options were of particular interest: monitoring, contract length and scientific support. Monitoring refers to contracts were monitoring of results was a contract option for land managers (e.g. Espinosa-Goded et al., 2010) and six studies (23%) of the surveys offered that. Contract length refers to whether the schemes offered contracts to land managers that would be longer than 5 years and thirteen studies offered that. As with monitoring, the 5-year period is common contract option is such surveys (e.g. Christensen et al., 2011). Lengthier contracts might increase land managers' participation as they offer a more secure financial future for the farmers while giving enough time for biodiversity-related benefits and enhancement of environmental feature to occur. Such benefits are well-known to require long periods of time to occur (e.g. Vaissière et al., 2018). Finally, scientific support refers to whether contracts offered to land managers included any type of scientific and farm advisory support and 8 studies in our sample offered such a contract feature. Such support might make land managers more inclined to enrol as contract requirements might feel more feasible (Emery and Franks, 2012).

More than half (58%) of the studies did not report farmers' farm sizes while only 11% reported some measure of farm income. Serious lack of reporting in the literature was observed with respect to sociodemographic variables. Only 35% of the studies reported the mean age of their sample.

Eight studies took place in Mediterranean climates under the Köppen-Geiger classification system (Spain, south of France and Portugal), five studies were undertaken in areas with warm summers and wet winters (France apart from the south and UK) while three studies were conducted in cold climates (Sweden and Finland). The rest of studies (23) were conducted in continental climates including Germany, Denmark and the UK among others.

3.2. Mean willingness to accept AES for biodiversity or environmental features

Table 2 shows the differences in average WTA value changes when weighting of studies according to the inverse variance method is applied. Initially we estimate a fixed-effects model using Eq. (1) for the pooled set of studies. Then a random-effects model following DerSimonian and Laird (1986) was fitted to the same data, estimated in *R* (R Core Team, 2013) with the *meta* command from Schwarzer (2007) on the mean WTA values of all 26 studies. The fixed-effects (FE) and random-effects (RE) meta-analysis results show large differences. In both cases the mean WTA is negative, but in the FE it is much smaller (-13.94 Euros/ha/year in the FE vs. -0.86 Euros/ha/year in the RE). This difference is statistically significant at the 1% level, demonstrating between-study heterogeneity. The high I^2 (100%) shows the large variation between studies might be due to real variations in WTA, also justifying the use of a random-effects meta-analysis. In order to explain the variation between studies, we conducted a series of subgroup analyses and tested whether their differences were statistically significant. First, we tested whether WTA was different between studies explicitly using the AES framing and those that referred to AES and PES framing interchangeably. The results were statistically insignificant. Then we tested for differences between studies that explicitly used the CAP as their reference for designing the proposed AES and those which did not explicit use the CAP framing. No statistical significant differences were found between these studies either. Offering support to farmers as part of their contract was significantly different between groups, with the studies that offered it having a WTA of -215.84 Euros [CI:-360;-71.30] compared to those that did not (WTA=178 Euros/ha/year [CI:137.1; 220.8]). Studies offering contracts longer than 5 years (WTA= -217.19 Euros/ha/year) [CI:350.46; -83.91]) also were statistically different from those that did not (WTA=221.61 Euros/ha/year [CI: 177.01; 266.22]). WTA of studies with schemes offering land managers monitoring of results also were statistically different from those that did not. The forest plot of all studies by year of survey implementation is presented in Figure 2, in the Appendix.

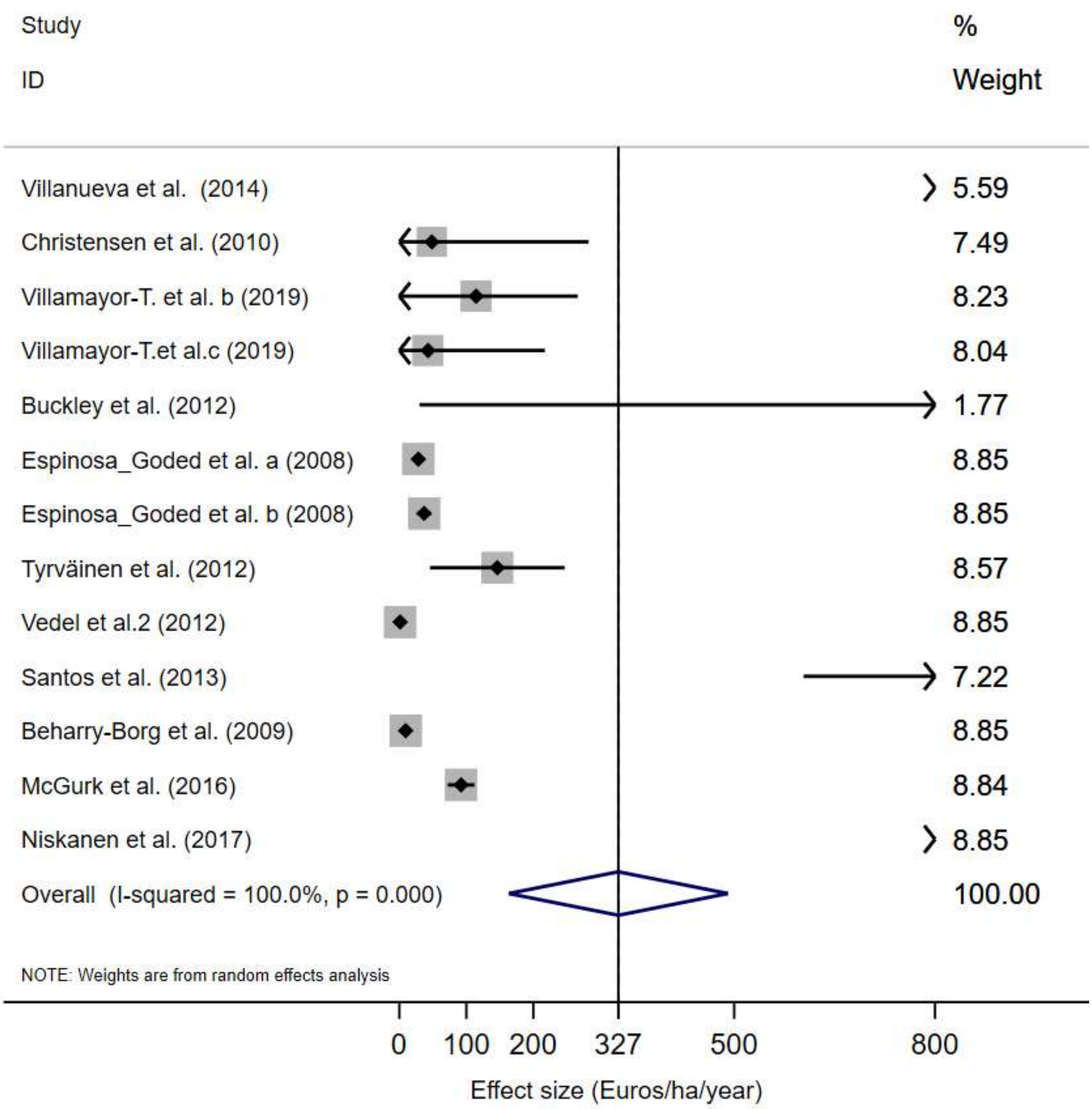
We also examined how estimates change based when reducing them to the “best set of studies” as defined earlier, i.e. when restricting it to studies which have a positive and statistically significant mean WTA, in a RE model. The result is a positive mean WTA of 327.02 Euros/ha/year for delivering biodiversity or environmental features. It is worth noticing the dramatic effect that including negative WTA values has on the average regressed value, which moves from -0.86 to 327.02 Euros/ha/year in the RE models (see Table 2).

The forest plot for this “best-set” of studies by year of study is presented in Figure 3. This shows how earlier studies were closer to the mean estimate than most of the later studies.

	Fixed-effects estimate (Pooled estimates, N=26)	Random-effects estimate (Pooled estimates, N=26)	Random-effects estimate (best set estimates, N=13)
Euros per hectare, per year for delivering biodiversity/environmental features	-13.94 [-13.95; -13.85]	-0.86 [-35.58; 33.86]	327.02 [163.36; 490.68]

Table 2: Mean land managers’ Willingness to Accept for Agri-Environment Schemes focusing on biodiversity or environmental features (values converted to 2019 Euros).

Figure 3: Mean effect from random-effects meta-analysis of Agri-Environment Schemes focusing on biodiversity or environmental features from “best set” of studies (13 studies with positive and statistically significant WTA)



3.3. Meta-regression results

In order to understand the factors determining heterogeneity on WTA across studies, meta-regressions were run in Stata15 using the *xtreg* command in order to include clustered standard errors and account for inter-dependencies of data from multiple studies, as advised in Harbord and Higgins (2008). The *metareg* command that assumes a cross-sectional data layout and normal standard errors was also used. For that, the evidence collected from the studies, and that has been presented in tabulated form in Table 1, was tested in a regression as independent variables. These were defined by the variables which during the subgroup analysis in section 3.2 showed statistically significant differences in mean WTA. We also included variables for key sociodemographic characteristics, as is common in the literature, as well as one related to the geo-climatic context of the country where the study took place. Each of the six categories of data detailed in Section 2.2. was represented in the models. The FADN measure of gross farm income was selected from the list of socio-economic variables while the variable indicating whether the study took place in a cold climate country according to the Köppen-Geiger classification was selected for the geographic variables. The meta-regressions were run for the pooled positive and negative WTA observations and for the best set of studies, with cross section and panel data layout models (Table 3), as is common in other meta-analyses of welfare measures within the field (Tyllianakis and Skuras, 2016; Jacobsen and Hanley, 2009).

	Pooled, cross section model (N=26)	**Pooled, random effects model (N=26)**	**Best set, cross section model (N=13)**	**Best set,random effects model (N=13)**
Dependent= WTA	*Coefficient (st. error in parentheses)*			
Constant	897.23*** (301.85)	545.61 (412.43)	439.07 (730.99)	459.19 (766.91)
Support offered	-332.06 (198.31)	93.37 (489.7)	-1152.98* (481.19)	-1188.10*** (0.14)
Long contracts	-473.22*** (183.78)	-802.50** (337.10)	-578.05 (713.38)	-668.84 (443.27)
Monitoring	-128.92 (209.22)	-497.79 (506.06)	69.45 (780.81)	93.36 ***(0.09)
Scheme focused on biodiversity	-585.91*** (244.85)	-314.94 (405.34)	162.22 (738.72)	-282.27 (765.80)
Scheme focused on environmental features	-196.34 (251.98)	190.50 (393.83)	901.99** (319.66)	710.08 (766.90)
Gross farm income	0.01 (0.00)	-0.01*** (0.00)	-0.01** (0.00)	-0.01*** (0.00)
Cold climate	43.55 (254.02)	156.31*** (4.92)	-280.81 (438.17)	-291.63 (443.27)

Adjusted R-squared	5%	38%	14%	68%

*Table 3: meta-regression results for land managers WTA for participating in AES delivering biodiversity or environmental features benefits, *** denoting significance at the 1% level, ** 5% level*

As expected, model fit is better for the best-set than the pooled models. The random effects models also show a better fit, with the best-set random effects model displaying a fairly good fit (68%) compared to the literature in this field (e.g. Soon and Ahmad, 2015; Tyllianakis and Skuras, 2016). This confirms that WTA estimates from multiple studies indeed impact the results and that including the negative WTA estimates very much affects the results. It should be noted, though, that the difference between the estimates of the best-set model that accounts for within-study heterogeneity with clustered standard errors and the model that does not, is very small, showing that the best-fit model for the best set does not suffer from non-independencies in the data.

In terms of the factors that affect WTA, there is great variability across the four models. The contract length variable is significant in the two pooled models. The sign is negative, which would have indicated that longer contracts lower the compensation that farmers require to enter the schemes. However, this variable is not significant in the best-set models. Support offered is not significant in the pooled models, but it is in the two best-set models. The sign is negative, indicating that farmers would be willing to accept less compensation to enrol if they obtain support. Monitoring is significant and positive in the random effects best set model. The positive sign indicates that if there is monitoring, farmers would require higher compensation. Effects of whether the schemes focus specifically on biodiversity or environmental features are quite inconsistent across the four models with diverging signs and statistical significance, and in any case, non-significant in the best set random effects model. The effect of the geo-climatic region is also inconsistent across all models and, in any case, not significant in the best set random effects model. Gross farm income is significant across all models except for the pooled cross-section one, but very small (estimate -0.01).

3.2. Sensitivity of WTA with respect to income

As income is showed to be significantly impacting WTA for enrolling in biodiversity or environmental features-protecting AES across models in Table 3, a simple model with both sides of Eq. (2) logarithmically transformed was estimated to measure the sensitivity of WTA with changes in income. Apart from the measure of gross farm income, the net farm income for the countries where the primary studies took place as provided by FADN was used to check for consistency. Naturally, such a model includes only studies with positive WTA given WTA values are log-transformed. The results are presented in Table 4 and show that elasticity's absolute value is higher than one and significant for the net farm income measure. Significant elasticity higher than one means that the proportion of WTA to the income measure increases as income increases, i.e. wealthier farmers benefit more from enrolling in AES.

Dependant = logWTA (N=13)	**Coefficient**	**St. error**
Log(gross farm income)	-2.74*	1.34
Constant	33.93*	16.40
Log(net farm income)	-5.13	3.05
Constant	57.05	33.73

*Table 4: Income elasticities of WTA, *denoting significance at the 10% level*

4. Discussion

The literature regarding land managers' WTA compensation to enrol in AES targeting biodiversity or environmental features in Europe is relatively recent and limited, but growing. Studies included in this work span over two different CAP periods where a move towards compulsory measures of greening and set-aside areas is evident and reflected in the academic effort, with over 70% of the studies having taken place in the last ten years.

Studies focusing on environmental features are more prominent in the AES literature than explicit biodiversity protection. This is also aligned with what found on studies in non-EU contexts, such as agri-environment programmes focusing on biodiversity in Australia (e.g. Salt, 2016; Ansell et al., 2016) and land conservation programmes the US (Hellerstein, 2017). In the EU-context, this can be attributed to the fact that several of the studies focused on

Ecological Focus Areas, which have become mandatory under the current CAP for arable farmers with land over fifteen hectares (Zinngrebe et al., 2017), while biodiversity benefits are less clearly spelled out as they are assumed to be co-benefits from ecologically focus areas (European Commission, 2017). Of the various scheme characteristics that might increase likelihood of land manager participation (i.e. monitoring, offering scientific support and lengthier contracts) none appeared in the majority of the studies in the literature (at most in 48% in the case of having contracts longer than 5 years). Such a finding is a clear testimony of the variety of AES contracts offered in Europe under CAP in its evolving formulation, as well as a lack of consistency in the support provided to land managers in the delivery benefits related to biodiversity or environmental features (Proctor et al., 2012).

From the general overview of the evidence, two findings are most striking. On the one hand, is the really scarce reporting on key characteristics such as farm size and, more importantly, farm income in the studies. While understandable (since it is always hard to get land managers to report their income), this on itself is problematic, since it restricts significantly the policy messages that can be derived from the evidence across the board.

On the other hand, is the fact that 48% of the existing studies report negative WTA for enrolling in such AES. As mentioned, negative WTA is to be considered in this context as land managers not willing to trade-off land productivity for monetary compensation for the delivery of biodiversity/environmental features or that they 'protest' to such trade-offs as presented in the existing studies (Czajkowski et al., 2019). The emphasis on *as presented in the existing studies* is an important one here that relates to a complex picture requiring careful discussion. At a first glance, this would suggest a widespread reluctance from the part of land managers to accepting the terms of compensation, either because they do not have an interest in delivering biodiversity or environmental features protection or because they are not willing to incur into the trade-off (as also suggested by the negative mean WTA in Table 2 for the pooled model). However, an in-depth look into the broader results challenges this conclusion, or in any case, makes it more nuanced. Firstly, as the results of the meta-regression models in

Table 3 show, the constant in the pooled model is highly positive and significant. Statistically, a significant model constant in this context means that land managers are, in principle, willing to accept a change from the status quo (Borenstein et al., 2015). This would therefore suggest that they are open for compensation to enrol in AES schemes, just possibly not in trade-off of the features offered to them (as per the studies). This suggest that the issue is not necessarily a *general* lack of willingness by land managers to receive compensation for biodiversity/environmental features, but rather the effect of protest/lack of preference for the *specific ways* in which these features are delivered to them in the studied AES schemes. This also resonates with findings from the quantitative study by Czajkowski et al., (2019) and qualitative studies (e.g., Uthes and Matzdorf, 2013). It seems, therefore, that land managers would be generally willing to trade-off compensation for the delivery of biodiversity or ecological features but that we (academia and/or policy) are not yet *hitting the right key* on how to best match their preferences for it.

A closer look at the factors influencing WTA (Table 3) also deepens in this complex picture. Firstly, as noted, there are great levels of inconsistency and variability depending on the evidence that is included in the regressions (pooled vs. best set), providing in the overall a weak evidence base. If not scrutinized carefully it can lead to potentially biased policy recommendations (considering the large effects on impacts and signs that the negative WTA estimates have in the pooled model). Focusing primarily in the best-set random effects model (i.e. the most robust evidence, albeit reduced), it would seem that offering support to farmers as part of their contract may reduce the amount of compensation that they are willing to accept for enrolling. This is consistent with findings from qualitative farmer surveys (e.g. Emery and Franks, 2012) where this feature was explicitly requested by farmers for future AES. However, the actual model estimate for this variable (-1188.10) makes the policy translation of this result implausible. This estimate would indicate that farmers are willing to lower their compensation by over one thousand euros per hectare per year for having this feature in their contract. This is higher than the highest compensation in EU-funded AES (when Natura 2000 areas are included in the land under contract, where payment can rise up to 900 Euros/hectare/year - otherwise highest levels are of 450 Euros (European Network for Rural Development, 2015)). Something similar happens with the (statistically significant) estimate for the variable on whether the scheme includes specific environmental features in the best set non-clustered model (901.99). Having monitoring as contractual feature in the AES, on the other hand, does yield statistically significant and plausible results in the best-set random effects model (positive estimate of 93.36). It makes sense to think that farmers would be requiring higher compensation if they are going to be monitored in their compliance with the scheme (e.g. Vedel et al., 2015), as a sort of compensation for being "policed". Re-imagining ways of monitoring compliance in ways that farmers are less put-off by it (e.g. using awareness-focused participatory approaches (Okumah et al., 2021) or offering payment based on modelled results instead of surveyed or sampled results (Bartkowski et al., 2021)) may provide some interesting avenues moving forward.

Significant negative effect of income in WTA (Table 3) is in line with what has been reported by some of the general environmental literature (e.g. del Saz-Salazar et al. 2012), but the effect is so small (1%), that it does not seem of particular policy relevance in this context. More interesting, however, are the policy implications of income's elasticity in WTA as they reveal the effect that the policy (AES delivering biodiversity and environmental features protection) has on income. Our findings (Table 4) would indicate that wealthier farmers stand to gain more than less wealthy farmers from enrolling in AES focusing on biodiversity and environmental features protection. This is to some extent miss-aligned with environmental and policy goals for the new CAP (European Union, 2013) and the requirements for achieving SDGs (e.g. SDG2, Griggs et al., 2017), which place a renewed emphasis in the promotion of rural development and landscape-wide approaches for a more effective delivery of biodiversity and environmental protection. If such AES are to be successful in achieving those goals, then new mechanisms for encouraging land managers with smaller holdings (who also tend to have lower income) to enrol are needed. If these schemes also promoted collaboration of adjacent small land holdings via collective action (Vanni, 2013), e.g. by providing incentive such as agglomeration bonuses (Sheremet et al., 2018), then more effective delivery of environmental protection is likely to be achieved, replicating what happens in larger holdings (e.g. Dallimer et. 2010; Schou et al., 2020). A final note on the average WTA values. The analysis of the best-set of studies shows an average of 327 Euros/hectare/year (Table 2), which is close to the average EU Direct Payments under Pillar I , i.e. around 350 Euros per hectare (European Commission, 2018). AES payments covered by this review are Rural Development payments under Pillar II (i.e. subsidies for cost incurred and income foregone). The relative high average WTA that we find would suggest that to deliver environmental and biodiversity protection, farmers require payment levels closer to current Direct Payments. This finding also confirms the validity of the use of stated preference methods in the context of preferences of suppliers (such as farmers and foresters) of ecosystem services, similar to Rodríguez-Entrena et al., (2019).

Of course, receiving compensation to participate in any AES is a key but not the only driver for participation since there is some evidence for voluntary farmer participation in environmental activities. For example, Mills et al., (2018) show that farmers have been found to still undertake environmental activities in unsubsidised land, although it should be noted that this land was adjacent to subsidised land and likely to have been benefited from this proximity. Having said that, Rodríguez-Entrena et al., (2019) have shown how payments are usually the contract feature that farmers focus most on.

5. Conclusions

Compensating land managers for the provision of public goods is currently one of the European Union’s flagship policies, and one that that resonates across the globe more generally in the effort to meet Sustainable Development Goals. Despite having been central to the EU’s Common Agricultural Policy for several decades now, Agri-Environment Schemes (AES) seem to only have had limited success in preserving biodiversity and providing environmental benefits. In part, the reasons for such limited success are attributed to factors that make farmers more or less inclined to adopt the schemes and accept compensation for the delivery of these biodiversity/environmental features. This paper has systematically reviewed and quantified, for the first time, the body of evidence that explores these factors with respect to their influence on land managers' welfare through the neoclassical economics notion of willingness to accept (WTA) compensation.

A first conclusion is that the evidence base is still relatively scarce but growing in accordance to the increased focus of the European Union in biodiversity and environmental provision. While this relative scarcity is understandable, the fact the published evidence fails to report key aspects such effects of farm size and farm income is problematic, since it hampers the possibility of drawing broader conclusions. The most striking result of our review, however, resides on the amount of studies reporting negative WTA values and the complex and nuanced picture that the broader results draw with respect to this matter. These broader results lead us to conclude that farmers are generally inclined to accept compensation for the delivery of biodiversity/environmental protection. However, the current evidence base provides few clues on how best that willingness is matched by contract design formats and contract features. Providing support to farmers and exploring new ways of monitoring compliance emerge as issues generally worth considering as means of incentivising farmers to enrol in AES. Further, the broader evidence base seems to support the idea that landscape solutions are going to require new mechanisms to incentivise smaller holdings (and collective action of adjacent ones). However, this alone seems, in the overall, like a quite modest contribution from the body of evidence to inform policy design more broadly. It indeed seems that we (academia and/or policy) are not yet *hitting the right key* on how to best match farmers preferences for enrolling in AES for the delivery of biodiversity/environmental features

A significant leap forward would not simply require an increased quantity of primary studies, but a deeper reflexion on how the complexity of farmers' preferences is best captured in the design of policy instruments that have to both share common features while being adaptable to context dependent characteristics at the landscape level. This is more pressing than ever face to the unprecedented challenges of Brexit and the COVID19-induced economic recession, which is going to put every cent of public funding under the hardest of scrutinies in the years to come.

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Appendix 1.

Name	**Year**	**Country**	**Type of land**
Lindhjem and Mitani (2007)	2007	Norway	Forest
Espinosa_Goded et al. (2008a)	2008	Spain	Farm
Espinosa_Goded et al. (2008b)	2008	Spain	Farm
Broch and Vedel (2009)	2009	Denmark	Farm
Vedel et al (2009a)	2009	Denmark	Forest
Beharry-Borg et al (2009)	2009	United Kingdom	Farm
Christensen et al. (2010)	2010	Denmark	Farm
Buckley et al. (2012)	2012	Ireland	Farm
Allo et al. (2015)	2012	Spain	Farm
Schultz et al. (2012)	2012	Germany	Farm (arable)
Tyrväinen et al. (2012)	2012	Finland	Forest
Vedel et al (2012b)	2012	Denmark	Forest
Santos et al. (2013)	2013	Portugal	Farm
Villanueva et al. (2014)	2014	Spain	Farm
Vaissière et al. (2018)	2016	France	Farm
McGurk et al. (2020)	2016	Ireland	Farm
Hasler et al. (2017) - Denmark	2017	Denmark	Farm
Hasler et al. (2017)- Estonia	2017	Estonia	Farm

Hasler et al (2017) - Finland	2017	Finland	Farm
Hasler et al. (2017) - Poland	2017	Poland	Farm
Hasler et al. (2017) - Sweden	2017	Sweden	Farm
Geussens et al. (2017)	2017	Uganda	Farm
Chèze et al. (2017)	2017	France	Farm
Niskanen et al. (2021)	2017	Finland	Farm
Czajkowski et al. (2019)	2018	Poland	Farm
Villamayor-Tomas et al (2019) - Switzerland	2019	Switzerland	Farm
Villamayor-Tomas et al (2019) - Spain	2019	Spain	Farm
Villamayor-Tomas et al. (2019) - Germany	2019	Germany	Farm
Latacz-Lohmann and Breustedt (2019)	2019	Germany	Farm

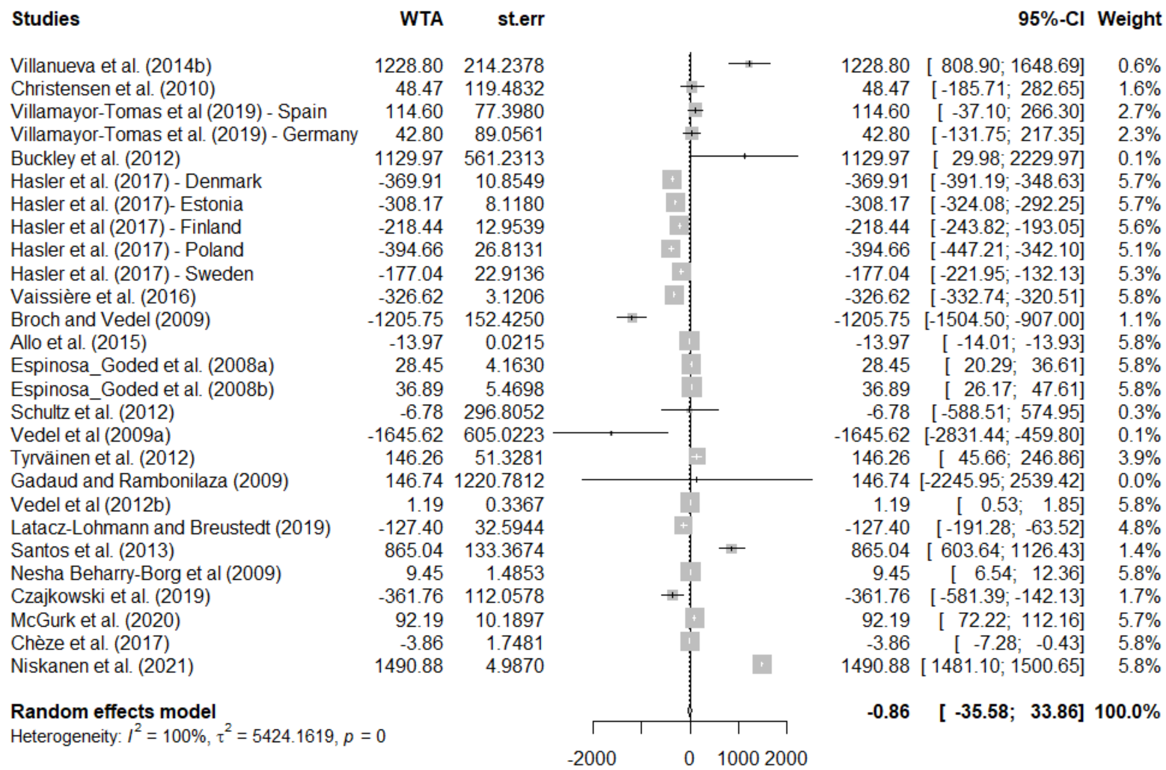

Figure 2: Mean effect from random-effects meta-analysis of AES focusing on biodiversity or environmental features from 26 studies (both positive and negative WTA).