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Analysis

Integrated valuation of semiarid Mediterranean agroecosystem services and disservices

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ABSTRACT

Agroecosystems are anthropised ecosystems where human activities, mainly agricultural practices, affect the innate functioning, leading to the provision of agroecosystem services (AES) and disservices (AEDS). This study presents a novel and integrated economic valuation of the AES and AEDS provided in a water-scarce Mediterranean area (south-eastern Spain), using a discrete choice experiment. The results reveal the social demand for AES and the disutility of AEDS, as well as the non-linearity in marginal utility for some of these AES and AEDS. Food provision, temperature regulation, leisure and recreation and biodiversity are socially perceived as AES. The water supply for irrigation switches between AES and AEDS depending on its provision level, while groundwater pollution is conceived as one of the AEDS. The integrated non-market value of AES and AEDS reaches 794 €/ha/year for the entire agroecosystem. This work provides guidelines for policy makers in the design of socially supported agricultural policies.

1. Introduction

Today, it is well-known that agriculture produces more than just food and fibre. The contribution of agriculture to society also encompasses the provision of non-commodity goods and services such as soil erosion control, climate regulation and biodiversity maintenance. Agroecosystem services (AES), defined as the contribution of an agroecosystem to human wellbeing (TEEB, 2010), represent an appropriate paradigm to embrace all these agricultural outputs. The ecosystem service framework, first developed by the Millennium Ecosystem Assessment (MEA (Millennium Ecosystem Assessment), 2005) and extended by TEEB (2010) and Haines-Young and Potschin (2018), constitutes an increasingly-used tool for assessment of the agricultural impacts on human wellbeing. This approach, which considers the multifunctionality of agriculture (Huang et al., 2015), covers the multiple agricultural outputs: both private and public goods and services (Fisher et al., 2009; Cooper et al., 2009). Essentially, the ecosystem service framework allows the assessment of the agroecosystems benefits for human wellbeing.

However, despite its apparent simplicity, this framework becomes complex when applied to agroecosystems, due to, among other factors, the presence of agroecosystem disservices (AEDS) and their trade-offs with AES. The concept of AEDS, defined as "the ecosystem generated functions, processes and attributes that result in perceived or actual negative impacts on human wellbeing" (Shackleton et al., 2016), reveals that some agricultural contributions to human wellbeing could be non-positive. In fact, agriculture is responsible for more than 70% of the annual consumption of water resources worldwide (WWAP, 2016), occupying nearly 50% of the Earth's land surface and emitting around 25% of the global anthropogenic greenhouse gases output (IPCC, 2019). Overlooking the existence of AEDS could have problematic consequences for research and policy orientations (Shackleton et al., 2016). Negative effects of agroecosystems on other ecosystems do exist (Power, 2010); thus, neglecting them implies not recognising a part of the overall contribution of agriculture to human wellbeing. Therefore, agricultural policy measures could be more cost-effective and efficient if they were to mitigate AEDS instead of enhancing AES (Shackleton et al., 2016).

Economic valuation of AES and AEDS allows recognition of their relative importance and, above all, the orientation of policy decisions when considering the overall contribution of agroecosystems to human wellbeing (De Groot et al., 2012). The implementation of sustainable agricultural practices implies the joint consideration of both the supply and demand of AES and AEDS. While the supply of AES and AEDS considers farmers' practices, their demand should be analysed regarding

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social preferences. The valuation of some AES, such as food provision, is straightforward due to the existence of markets. However, most AES and AEDS are non-marketed and require alternative methods to estimate their values. Stated preference methods, such as contingent valuation and discrete choice experiments (DCEs), are the methods used most for this purpose, since they allow estimation of the social demand for their provision, and thus the willingness to pay for, and the value of, AES and AEDS variations (TEEB, 2018). These methods can also be combined with multi-criteria analysis to assess the provision of information to individuals and its impact (Martin-Ortega and Berbel, 2010).

In such a context, the need for an integrated framework for the valuation of AES and AEDS should be addressed from the demand side. Specifically, the present work aims to value economically the integrated provision of AES and AEDS in a water-scarce Mediterranean agroecosystem (south-eastern Spain), using a DCE. Our results are expected to have policy relevance as they provide a direct estimation of social preferences regarding the contribution of agriculture to human wellbeing. They will also guide the design of socially accepted agri-environment policies, bearing in mind the next reform of the Common Agricultural Policy of the European Union (Pe'er et al., 2019).

Previous studies have analysed the interrelationships between AES and AEDS by using different approaches. Zhang et al. (2007) were the first authors who recognised the presence of AEDS in their assessment scenario. However, their work focused more broadly on the AEDS *to agriculture*; that is, the AEDS provided by other ecosystems to agriculture. Power (2010) assessed AEDS both *to and from agriculture*. He highlighted nutrient loss, pollution and emission of greenhouse gases as the main AEDS. Ango et al. (2014) analysed farmers' management of trees in agricultural landscapes in Ethiopia, considering their provision of AES and AEDS, while Ma et al. (2015) assessed the AES and AEDS provided by a high-production agroecosystem in China, using emergy valuation. Nevertheless, it is difficult to find in the literature studies which quantify economically AES and AEDS in an integrated way (Chang et al., 2011; Hardaker et al., 2020; Sandhu et al., 2020). Therefore, to the best of our knowledge, this is the first work which attempts to value AEDS, and it provides an integrated non-market valuation of all agricultural contributions to human wellbeing by using stated preference methods.

The paper is structured as follows. In the following section the adequacy of the integrated valuation of both AES and AEDS is discussed in more depth. Section 3 explains the methodology employed, with special attention to the selection of AES and AEDS for the assessment, while Section 4 presents the main results. The discussion of the results, as well as their theoretical implications and policy applications, is provided in Section 5. Finally, in Section 6 the conclusions are stated.

2. The need for a framework for the integrated valuation of AES and AEDS

Since the publication of the MEA (Millennium Ecosystem Assessment) (2005), there has been a growing body of work aimed at measuring the benefits provided by agroecosystems (e.g. Granado-Díaz et al., 2019; Tienhaara et al., 2020). However, little is known about the negative effects, or social cost, of agroecosystems. AEDS is not a straightforward concept. Wellbeing reductions could be caused by both reducing AES and providing AEDS (Vaz et al., 2017). Due to the synergies among socio-ecological processes, AES and AEDS could vary simultaneously, providing trade-offs among them (Blanco et al., 2019). Anthropised ecosystems, such as agroecosystems, have to deal also with the effects of human management, which could both mitigate and promote the provision of AEDS (Barot et al., 2017). The same agroecosystem functions and processes could be perceived as AES or AEDS depending on people's behaviour, preferences and the socioeconomic context (Shackleton et al., 2016; Vaz et al., 2017).

Integrated assessment of AES and AEDS is needed for at least three reasons. Firstly, because considering only AES implies taking into account just part of the overall contribution of the agroecosystem to wellbeing (Schaubroeck, 2017). Secondly, the global assessment of AES and AEDS allows integration of the trade-offs between them and considers the net impact of agroecosystems on human wellbeing (Barot et al., 2017; Blanco et al., 2019). Finally, it helps to achieve better design of policies intended to produce sustainable and resilient agroecosystems (Sandhu et al., 2019). This becomes more significant when economic valuation is included in the assessment framework. If AEDS are ignored in policy design, this could lead to overestimation of the benefits provided by agroecosystems - which will be translated into suboptimal solutions - and could lead policy makers to make wrong decisions since they have not taken into account the costs they imply.

Recent literature makes the case for the introduction of AEDS into the assessment framework (e.g. Shackleton et al., 2016; Barot et al., 2017; Blanco et al., 2019; Sandhu et al., 2019). Specifically, Blanco et al. (2019) proposed that, to strengthen the inclusion of AEDS in research and policy analysis, it is necessary to: develop an AEDS classification which unifies further research; assess AES and AEDS in both biophysical and socio-economic terms, integrating them into a common framework; broaden the analysis to trade-offs among AEDS and between AES and AEDS; evaluate spatial and temporal variation in AEDS demand and supply; and integrate the assessment of AEDS co-production in research and policy agenda.

Although some steps have been taken to establish ecosystem disservices concepts and classification, there is still no widely accepted typology of disservices, which is especially notable if the focus is only on AEDS. Escobedo et al. (2011) split up disservices into three categories - financial costs, social nuisances and environmental pollution - to account for the effectiveness of urban forests in pollution mitigation. Von Döhren and Haase (2015) provided a literature review on ecosystem disservices research and clustered the negative effects of ecosystem functioning according to thematic fields: ecological, economic, health, psychological and general impacts on human wellbeing. This typology was also applied by Campagne et al. (2018) to assess the capacity of a French natural park to provide ecosystem services and disservices. On the other hand, Shackleton et al. (2016) defined six types of disservices according to the biotic or abiotic origin and the expected impact on the different aspects of human wellbeing: bio-economic, bio-health, bio-cultural, abiotic-economic, abiotic-health and abiotic-cultural. Following the approach of Shackleton et al. (2016), Vaz et al. (2017) established that disservices could be alternatively classified among health, material, security and safety, cultural and aesthetic and leisure and recreation typologies.

All these different proposals reveal that, despite the huge effort in defining a widely applicable classification of ecosystem disservices, greater consensus is needed to clarify and reach a widely accepted agreement on this matter. Moreover, controversy arises when the purpose is to integrate AES and AEDS in the same framework of assessment. The main accepted paradigms of ecosystem services, such as MEA (Millennium Ecosystem Assessment) (2005), TEEB (2010) and CICES (Haines-Young and Potschin, 2018), agree on the classification of services in provisioning, regulating and cultural categories for valuation purposes. However, it is a challenge to integrate AES in this categorisation, to link them with the established and proposed typologies of AEDS. This challenge might be overcome if similar classifications were applied to both AES and AEDS. Barot et al. (2017), who proposed a general framework to assess anthropised ecosystems, as well as Hardaker et al. (2020) and Sandhu et al. (2020), who provided integrated economic valuations of AES and AEDS, are a few examples of authors who claim to have applied the same existing categories for services to disservices.

The integrated assessment of AES and AEDS in both biophysical and socio-economic terms is gaining momentum in the literature (Blanco et al., 2019). Most of this research deals with some specific AEDS and how their flows impact on agroecosystems and socio-ecological systems. This is the case, for instance, of the work of Rasmussen et al. (2017), which assessed the switch between AES and AEDS that happens when wild animals and plants (biodiversity) are present excessively in agroecosystems, and of that of Pejchar et al. (2018), which showed the net effects of birds in agroecosystems. Other studies, however, have focused on empirical applications, from the analysis of trade-offs between AES and AEDS (Finney et al., 2017; Nguyen et al., 2018) to their integrated evaluation (Ma et al., 2015; Schäckermann et al., 2015; Shah et al., 2019; Blanco et al., 2020). In particular, Finney et al. (2017) showed how a mixture of cover crops provides AES and AEDS and how trade-offs among them arise, whilst Nguyen et al. (2018) optimised the provision of AES and AEDS -particularly primary production, soil organic carbon, water use, nitrogen leaching and GHG emissions- and their trade-offs by using a biogeochemical model in an irrigated corn production system. The integrated assessment of AES and AEDS has been developed also from an emergy-based approach to concrete agroecosystems in China (Ma et al., 2015) and Pakistan (Shah et al., 2019), accounting the relationships among natural and semi-natural habitats surrounding cropland and agroecosystems in terms of AES and AEDS. Blanco et al. (2020) even considered farmers' attitudes and perceptions regarding the management of rural forests in agricultural landscapes in France.

The economic valuation focus can also be broadened by accounting for AEDS, which are rarely used in integrated economic valuation. As far as we know, only a few studies have addressed the economic valuation of AES and AEDS from an integrated perspective. Chang et al. (2011) estimated the net value of AES provided by greenhouse vegetable cultivation compared to conventional cultivation, in China, by employing food production, CO_2 fixation, soil retention and soil fertility as AES, and irrigation water use, NO_3^- accumulation and N_2O emissions as AEDS indicators. Similarly, Hardaker et al. (2020) estimated the value of agricultural uplands in Wales, taking livestock and crop production, water supply, carbon sequestration and employment as AES flows, and water quality reduction and greenhouse gases emissions as AEDS flows. Sandhu et al. (2020), for their part, estimated the economic value associated with the benefits and costs of corn production systems in Minnesota (US) by adapting the TEEBagrifood framework (TEEB, 2018), which considers not only AES and AEDS flows, but also the stock of the social, human and natural capital produced. Nevertheless, all these authors used direct market and cost-based methods to estimate the economic values of AES and AEDS.

The range of methodology used to assess AEDS is as broad as that available for AES (Campagne et al., 2018; TEEB, 2018). Thus, the method employed will depend on the pursued aim. Since our study aims to value monetarily AES and AEDS in an integrated way, the methodology employed will require the use of economic valuation techniques. TEEB (2018) listed direct market value approaches, cost-based methods, revealed preference approaches and stated preference methods as the main methodological frameworks applied for economic valuation of AES and AEDS.

Our work applied DCEs in order to calculate the monetary value associated with the AES and AEDS provided by agriculture. This method was used because, based on microeconomic utility theories, it models trade-offs among AES and AEDS and allows measurement of the net wellbeing impact of agroecosystems. Wellbeing is, in turn, the root of the AES and AEDS concept. The employment of DCEs has grown in recent years, with many purposes, such as analysis of the economic value of water for irrigation (Rigby et al., 2010), the design of agri-environment schemes (Vaissière et al., 2018) and assessment of the demand for AES (Jourdain and Vivithkeyoonvong, 2017; Tienhaara et al., 2020).

Focusing on agroecosystems, previous studies were aimed at understanding the social preferences for specific AES, such as pollination (Breeze et al., 2015), biodiversity (Varela et al., 2018) and soil carbon sequestration (Glenk and Colombo, 2011; Rodríguez-Entrena et al., 2014), or for a set of AES provided by a specific type of agroecosystem. Several of these are now cited. Rodríguez-Ortega et al. (2016) developed an economic valuation of the AES provided by Mediterranean high nature value farmland, specifically the landscape aesthetic, biodiversity, forest fire prevention and supply of quality products. Jourdain and Vivithkeyoonvong (2017) estimated the social demand for food production, drought mitigation, water quality and the maintenance of the rural lifestyle in irrigated rice agroecosystems in Thailand. Novikova et al. (2017) broadened the scope of their assessment to include the entire country of Lithuania and valued the preferences for the reduction of underground water pollution, biodiversity and maintenance of agricultural landscapes as the main AES. Granado-Díaz et al. (2019) focused on olive groves in Andalusia (Spain) and valued the social demand for soil erosion prevention, carbon sequestration and the biodiversity provided in such agroecosystems. Notwithstanding, to our knowledge, DCEs have not been applied yet to the economic valuation of AEDS, or to the integrated valuation of AES and AEDS, which adds to the novelty of the present work.

The contribution of this paper to the on-going research into the economic valuation of AES and AEDS is two-fold. Firstly, it represents the first non-market valuation which integrates AES and AEDS in a common framework by using stated preference methods. It is expected, therefore, that the results will provide a better insight into social preferences for agriculture and the relationship between agroecosystems and human wellbeing. Secondly, we address the main agri-environmental challenges facing water-scarce Mediterranean agroecosystems. The case study of the Region of Murcia comprises a region where a great variety of agroecosystems exist -from rainfed to highly-intensive agroecosystems- and where the human and agricultural pressures threaten the surrounding ecosystems and even the inner agroecosystem functioning. Hence, we expect the present work will serve to better inform the design and implementation of current and future agricultural policies in areas with similar characteristics.

3. Material and methods

3.1. Case study

The case study is the agroecosystems of the Region of Murcia (southeastern Spain), within the Segura River Basin (Fig. 1). This region, bordering the Mediterranean coast, is characterised by a semi-arid climate with low rainfall (< 400 mm/year) and high mean annual temperatures (between 10 and 18 °C); hence, water scarcity is one of its main characteristics. The existence of good-quality soils has fostered the development of a very important agricultural sector here. Relevant environmental challenges in the area are the soil degradation, groundwater overexploitation and salinisation and biodiversity loss. These agri-environmental characteristics make the Region of Murcia a case study representative of most semiarid Mediterranean regions (Martínez-Paz et al., 2018).

The agroecosystems within the Region of Murcia can be classified into three different sub-systems regarding their geomorphological characteristics, water availability and inputs-outputs relations with other ecosystems. There is a rainfed agroecosystem and an irrigated one, which can be further divided into a traditional irrigated agroecosystem (Heider et al., 2018) and a highly-intensive irrigated agroecosystem (Alcon et al., 2017). The rainfed agroecosystem covers around 253,000 ha (CARM, 2017), which represents 57% of the total cropland. Water scarcity determines the crop typology: almonds and olive orchards, as woody crops, and cereals, among the herbaceous crops, predominate. Irrigated agroecosystems - traditional (25%) and highly-intensive (75%) - cover 188,000 ha (CARM, 2017). The traditional irrigated agroecosystem follows the Segura River valley, citrus orchards being the main crop. It is recognisable by its landscape, well-known as the Huerta of Murcia, with high social and cultural values (Martínez-Paz et al., 2019). The highly-intensive irrigated agroecosystem occupies the lowlands, spreading from the south to the north of the Region along the Mediterranean coastline: horticultural crops and citrus are the main crops and their production is export-oriented.

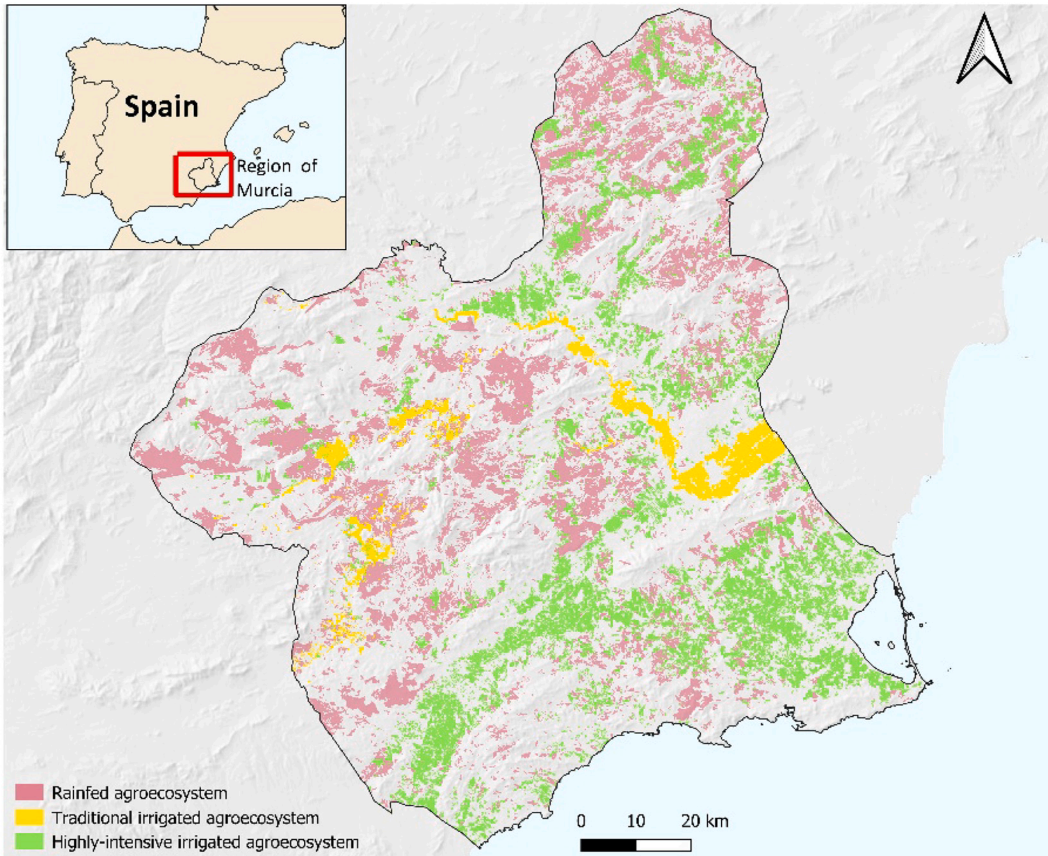

Fig. 1. Study area.

3.2. Discrete choice experiment (DCE)

The DCE is a stated preference method based on the multi-attribute utility (Lancaster, 1966) and random utility (McFadden, 1974) theories. The fundaments of this method can be found in Champ et al. (2017).

3.2.1. Experimental design

The implementation of a DCE requires, as a first step, the selection and definition of the attributes and their levels. Table 1 summarises the six attributes (and associated indicators, as well as their levels) used in the experiment. These attributes were selected based on a consultation with experts. We considered as experts those stakeholders involved in agri-environmental management in the Region of Murcia. Four different groups of stakeholders were consulted: (i) users, which included

Table 1
Attributes and levels.

	AES/AEDS	Attribute	Definition (Indicator)	Units	Levels
Provisioning services	Food provision (AES)	Yield (FOOD)	Annual almond yield produced by the agroecosystem	kg/ha/year	< 500 500–1000 1000-2000
	Water (AEDS)	Water supply for irrigation (WATER)	Irrigation water supplied to the crop system	m^3/ha/year	0 < 3000 3000-5000 > 5000
Regulating services	Climate regulation (AES)	Temperature regulation (TEMPE)	Temperature changes on the land surface due to agriculture	°C	0 −1 °C −2 °C
	Waste treatment and water purification (AEDS)	Groundwater pollution (POLL)	Nitrate concentration in aquifers	mg NO_3^-/L	< 50 50–200 > 200
	Maintenance of genetic diversity (AES)	Biodiversity (BIOD)	Bird species richness as a share with respect to potential	%	60% 80% 100%
Cultural services	Opportunities for recreation and tourism (AES)	Recreation and tourism (RECRE)	Chance of enjoying activities in the agroecosystems	–	No Yes
		Monetary payment (COST)	Part of current paid taxes directed to support agricultural policies (tax redistribution)	€/household/month	0 3 6 9 12

farmers, agricultural engineers and technicians, etc.; (ii) researchers from the public and private sectors involved in agricultural, ecosystem services and economic research; (iii) public managers, which encompassed agricultural and water management authorities; and (iv) civil society, mainly representatives from political parties, environmental associations and NGOs. The sample comprised 44 experts, equally distributed among the groups of stakeholders. The consultation with the experts was developed by face-to-face interviews, between July and September 2018, and a DCE was also used, after a pre-selection of 12 attributes based on a literature review (see Zabala et al., 2021). Based on the experts' opinions, attributes were chosen to include the overall agricultural contributions to human wellbeing, but focusing on the main agri-environmental challenges of the case study. The levels of the attribute indicators deal with the three different agroecosystems found in the Region of Murcia.

Food provision constitutes the main provisioning service provided by these agroecosystems. Hence, "yield", measured in terms of final production per unit area, was used as the indicator of this service, following Jourdain and Vivithkeyoonvong (2017). Almond production was chosen as representative since almond is the only crop present in all three agroecosystems of this case study. In addition, three levels, each associated with one agroecosystem, were assessed: rainfed (< 500 kg/ha/year), traditional irrigated (500–1000 kg/ha/year) and highly-intensive irrigated (1000–2000 kg/ha/year). This indicator was used to assess the contribution of the agroecosystems to food security (Cooper et al., 2009; Villanueva et al., 2018).

The second attribute was the water supply for irrigation, which was considered among the AEDS due to the water scarcity that dominates in the study area and generates rivalry for water resources among competing uses (Zabala et al., 2019). This implies not only the need for economic and water-allocation solutions, but also social and environmental challenges (Perni and Martínez-Paz, 2017). All three agroecosystems were included within this attribute, since the rainfed agroecosystem was comprised by a zero-water supply level.

The third attribute was the local climate-regulation service, due to the impact of agriculture on the mitigation of the effects of climate change. Temperature variation on the land surface was selected as the indicator. The selected levels ranged from 0 to 2 °C of temperature reduction, corresponding to the values of the rainfed and irrigated agroecosystems, respectively (Albaladejo-García et al., 2020).

The fourth attribute was groundwater pollution. The contribution of agroecosystems to waste treatment and water purification is expected to be negative, which means it should be considered among the AEDS (Shackleton et al., 2016). Groundwater pollution is one of the main agri-environmental challenges nowadays, largely due to the salinisation of water bodies, which can be caused by nitrate discharges from agriculture (Alcolea et al., 2019). The selected levels cover the different nitrate concentrations measured in the aquifers of the case study, which can be associated with each type of agroecosystem found in the Region of Murcia (CHS, 2017). To our knowledge, although groundwater quality has also been included as an attribute in other agroecosystem valuations (Niedermayr et al., 2018; Tienhaara et al., 2020), this is the first time that it has been considered among the AEDS.

The fifth attribute was biodiversity. This attribute was measured as the bird species richness indicator. The selection of this indicator was motivated by the fact that bird richness has been reduced in the last few years due to agricultural activity (Beckmann et al., 2019), as well as by the ease with which it is understood by ordinary citizens, as proved in several other agroecosystem valuations (Varela et al., 2018). The relationship between biodiversity and agricultural intensity is not linear (Beckmann et al., 2019). Indeed, this relationship is greatly influenced by agricultural practices (Aguilera et al., 2020). Crop diversity and heterogeneous landscapes enhance bird species richness (Stjernman et al., 2019), even in fruit-tree crops (Rime et al., 2020). In the present work, biodiversity levels were defined as the share of the potential bird richness which could be found in the agroecosystems, following the Perni and Martínez-Paz (2017) procedure. Thus, low-intensity agroecosystems together with heterogeneous landscapes, such as the traditional irrigated agroecosystem in the case study, provide greater bird species richness. Therefore, it is expected that the highly-intensive irrigated agroecosystem, dominated by monoculture, would exhibit a value of 60% for bird richness with respect to the potential richness, while for the rainfed agroecosystem, of low intensity but with homogeneous landscapes, the value would be 80%.

The sixth non-monetary attribute was related to the cultural contribution of agroecosystems to human wellbeing, another of the AES. This was measured by means of a dummy indicator which reveals the chance of enjoying agroecosystems. Although most agroecosystem valuations have included the aesthetic landscape as a cultural service (e.g. Rodríguez-Ortega et al., 2016; Niedermayr et al., 2018), this experiment considers recreation, leisure and the contribution to ecotourism. These comprise the benefits derived from enjoying agricultural landscapes through participation in, for instance, sporting activities, farm tours, and bird watching, according to the experts consulted.

The monetary attribute referred to the reallocation of household taxes (Rogers et al., 2020) to support agricultural policies in the Region of Murcia; this currently amounts to around 6 €/household/month (CARM, 2018). Five levels were included, extending above and below this value and ranging from 0 to 12 €/household/month. Thus, the respondents could choose levels below or above the current value.

The attributes and levels were combined through a Bayesian efficient design, using as priors the coefficients estimated with a conditional logit model developed after the consultation with experts (see Zabala et al., 2021). The choice-sets were generated with Ngene software (ChoiceMetrics, 2012). The Bayesian D-error for the final design was 0.000194. The final design resulted in 20 choice-sets grouped in 4 blocks. Each choice-set was composed of 3 unlabelled alternatives (Jin et al., 2017), which represented different agroecosystems. The respondents were asked to choose the agroecosystem that they would like to be implemented in the Region of Murcia, according to their preferences and budget restriction. Besides, the numerical attribute levels were encoded as categorical (high, medium and low levels) to ease their understanding during the survey (Barkmann et al., 2008) and to avoid the *endpoint problem* (Kontogianni et al., 2010). The fifth choice-set in every block was readapted to include the assessment of preference monotonicity. For this, the third alternative in each fifth choice-set encompassed a dominated choice alternative (Mattmann et al., 2019). If an individual chooses this alternative, it reveals their non-monotonous preferences and, thus, this observation should be removed from the sample. The attribute levels within the choice-sets were presented with visual aids to make their understanding easier. An example of a choice-set is provided in Appendix A.

A forced choice design was employed. This design was selected due to the aim of the research: we intended to value the AES and AEDS, not the changes in their provision because of specific agricultural policies. So, there was no need to include an opt-out option covering the non-support of a specific policy in each choice-set. Besides, since there are different types of agroecosystems within the case study, it would be inviable to define a fixed agroecosystem as a *status quo* or *business as usual* alternative. In this context, the forced choice design prevents the respondents from selecting the opt-out alternative as a strategy to elude the cognitive effort of revealing their preferences (Rigby et al., 2010; Alemu and Olsen, 2018). Nevertheless, a zero-cost level was also included in the design, to cover non-preferences in the public support of agriculture.

3.2.2. Data collection

Data were collected between January and February 2019, by face-to-face interviews. These were conducted by trained enumerators. An information brochure was given to the respondents to provide specific information about the definition of the attributes, indicators and levels. The target population was the households of the Region of Murcia

(539,000 households), the final sample comprising 433 households. Households were randomly selected, following a stratification by county. The survey was administrated in public spaces, such as markets, parks and squares, in order to ensure the randomness of the sample. The sample size, for a 95% confidence level, provided a sample error term below 5%.

Steps were taken to mitigate hypothetical bias, following the recommendations of Loomis (2014). Specifically, two *ex-ante* strategies were applied: (1) the respondents were noticed that the survey results would be used to inform agricultural policies and, so, would have a consequent impact on the public budget distribution; (2) a brief cheap-talk about the aims of the research and the definitions of AES and AEDS was provided. Champ et al. (2009) demonstrated that cheap-talk is effective when the respondents are not familiar with the goods or services to be valued, which is the case of AES and AEDS. In fact, 88% of the respondents admitted not knowing about the concepts of AES and AEDS before the interview.

3.3. Econometric and valuation framework

According to the random utility theory (McFadden, 1974), the utility (U_{ijt}) provided for an individual i from choosing an agroecosystem alternative j in a choice set t can be decomposed into an observed (V_{ijt}) and an unobserved part (ε_{ijt}), considered additively:

$$U_{ijt} = V_{ijt} + \varepsilon_{ijt} = \sum\nolimits_{k=1}^{K} \beta_{ik} X_{kjt} + \varepsilon_{ijt} \tag{1}$$

where V_{ijt} is the deterministic part of the utility, determined by the k attribute levels (X_{kjt}), and ε_{ijt} is a stochastic error term, identically and independently distributed following a Gumbel-distribution. Assuming V_{ijt} to be a weighted sum of the attribute levels, β_{ik} is the individual marginal utility obtained from each of the k indicators for AES and AEDS, reflecting how the utility level changes if the provision of AES and AEDS increases.

However, despite its wide use, a linear utility function (Eq. (1)) is not always the best way to model social preferences, since marginal utility could be non-constant. Preferences for goods and services tend not to be linear, but to have a concave form. In order to deal with this, terms describing the interactions among attributes have been included in the assessment (Karaca-Mandic et al., 2012). Hence, one can distinguish squared terms of the continuous attributes, which generate a quadratic utility function, from the interactions terms among different attributes. This adds to the analysis of the relationship among the attributes considered in the interaction. Applying both types of interactions terms to the model specification gives:

$$U_{ijt} = V_{ijt} + \varepsilon_{ijt} = \sum\nolimits_{k_1=1}^{K} \beta_{ik_11} X_{k_1jt} + \sum\nolimits_{k_1=1}^{K} \beta_{ik_12} X_{k_1jt}^2 + \sum_{k_1,k_2} \beta_{ik_1k_2} X_{k_1jt} X_{k_2jt} + \varepsilon_{ijt} \tag{2}$$

$\forall k_1, k_2 = 1, \ldots, K, k_1 \neq k_2$

The model applied most commonly to estimate the utility function is the mixed logit (MXL). It allows the coefficients to be individual-specific, through the assumption that they follow a density function $\beta \sim f(\beta|\rho)$, ρ being the set of parameters which describe their distribution. This permits one to model unobserved heterogeneity across individuals, and to overcome the independence of irrelevant alternatives (Hensher et al., 2005). The MXL model is estimated using the maximum simulated likelihood estimator (Train, 2009). Specifically, the utility function was modelled in R software (R Core Team, 2019), using the Apollo package (Hess and Palma, 2019) and 500 Halton draws for the simulation of the log-likelihood function.

The economic value of AES and AEDS is estimated using the marginal rate of substitution (MRS). When a cost attribute is included in the DCE, the MRS between the non-cost attributes and the cost attribute shows the willingness to pay (WTP) for the non-cost attributes. Following Eq. (3), it is calculated as follows:

$$MRS_c^{k_1} = WTP_{k_1} = \frac{\partial U_{ijt} / \partial X_{k_1jt}}{\partial U_{ijt} / \partial COST} = -\left(\frac{\beta_{k_11} + 2\beta_{k_12} X_{k_1} + \beta_{k_1k_2} X_{k_2}}{\beta_c}\right) \tag{3}$$

where β_c refers to the marginal utility of the cost attribute. Since this specification could imply non-constant marginal utility for some of the attributes included, $X_{k1,\ 2}$ represents the provision level for the mentioned attributes k_1 and k_2, respectively. WTP_{k1} represents, in monetary terms, how much the respondents are willing to pay for a unit increase in each AES or AEDS k_1 provided by the agroecosystem.

In order to estimate how a certain provision level of AES or AEDS impacts on human wellbeing, we need to calculate the consumer surplus (CS) associated with this provision level. It can be derived as follows (Freeman et al., 2014):

$$CS_{k_1}(X_{0k_1}) = \int_0^{X_{0k_1}} WTP_{k_1}\, dX_{k_1} = \int_0^{X_{0k_1}} -\left(\frac{\beta_{k_11} + 2\beta_{k_12} X_{k_1} + \beta_{k_1k_2} X_{k_2}}{\beta_c}\right) dX_{k_1} \tag{4}$$

where CS_{k1} (X_{0k1}) represents the consumer surplus associated with the AES or AEDS k_1 evaluated at provision level X_{0k1}.

Aggregating CS_{k_1} for the $k_1 = 1, \ldots, K$ AES and AEDS provided by an agroecosystem, the total economic value (TEV) provided by the agroecosystem can be calculated:

$$TEV = \sum\nolimits_{k_1=1}^{K} CS_{k_1}(X_{0k_1}) \tag{5}$$

3.4. Sample characteristics

The sample comprised 433 households. Descriptive statistics for the main sociodemographic characteristics of the sample are shown in Table 2. The sample was totally representative of the regional census data in terms of gender, monthly income and educational level, which ensures the results represent social preferences. Furthermore, 17% of the respondents admitted that at least one household member worked in farming. This is also representative in terms of the active population, guaranteeing an appropriate distribution between farmer and non-farmer-related households.

4. Results

4.1. Estimated choice models

The social utility function was estimated employing different specifications, with a final sample of 425 observations, after removing eight cases associated with individuals who stated non-monotonous preferences (Mattmann et al., 2019). Food provision (FOOD) and water supply

Table 2
Sample and population descriptive statistics.

Variable	Sample	Region of Murcia	
Sociodemographic information			*t*-test (*p*-value)
Age (years)	43.36	47.90[a]	−4.91 (0.00)
Gender (% women)	50.81	50.32[a]	0.20 (0.84)
Household income (€/month)	2406	2429[b]	−0.43 (0.67)
Educational level (%)			Pearson χ^2 (p-value)
Lower education	6.70	10.00[c]	1.25 (0.74)
Primary education	8.08	9.80[c]	
Secondary education	44.80	46.60[c]	
Higher education	40.42	33.50[c]	
Relation to farming			
Does any member of your household work in farming? (%)	16.86	13.40[c]	1.92 (0.06)

[a] INE (2018a).
[b] INE (2018b).
[c] INE (2019).

for irrigation (WATER) were rescaled to tonnes and dm^3, respectively.

Table 3 shows the main estimated models. Model 1 presents an *MXL-Linear* specification. The coefficient signs verify the consideration of AES and AEDS established previously. Food provision (FOOD), contribution to biodiversity (BIOD) and the chance to do recreational activities within the agroecosystem (RECRE) have a positive sign, revealing their provision has a positive impact on human wellbeing and, thus, that their consideration as AES was correctly specified. In contrast, water supply for irrigation (WATER) and groundwater pollution (POLL), which were predefined as AEDS, show negative signs. This specification implies that the marginal utility is constant and, thus, independent from the provision level of AES and AEDS.

However, microeconomics suggests the existence of concave utility functions with diminishing marginal utility. To overcome this challenge, a non-linear relationship between attributes and social utility was tested using squared attributes (Model 2) and also including interaction terms (Model 3). A step-wise procedure was followed to select the squared and interaction terms that better fitted both models; concretely, all feasible squared and interaction terms were tested (saturated models) and non-significant terms were deleted until reduced models which better explained the choices were obtained. All models were estimated assuming a normal distribution for non-monetary and squared attributes, whilst the cost coefficient and interaction terms between different attributes were set as fixed.

Model 2 shows an *MXL-Quadratic* specification. Significant coefficients of squared attributes were obtained for FOOD, WATER and POLL, revealing the non-linearity in the utility function. The coefficients of the squared attributes show negative signs, indicating the concavity of the utility function and the diminishing marginal utility provided. The *MXL-Quadratic* specification provided a better fit than the *MXL-Linear* model ($LR = 106.25$; $\chi^2_{0.05;6} = 12.59$). The inclusion of interaction terms between attributes (Model 3) also improved upon Model 2 ($LR = 13.53$; $\chi^2_{0.05;2} = 5.99$). Therefore, Model 3 is the preferred model to be used in the follow-up assessment. This model is an MXL model with two significant interactions: *FOOD*WATER* and *WATER*POLLUTION* (Table 3).

For Model 3, the mean coefficients are significant at least at the 10% level, while the standard deviation estimations are significant for all attributes except FOOD and WATER. These results reveal that the perceived impact of food provision and water supply for irrigation on human wellbeing was homogenous across the respondents.

The mean coefficients for AES and AEDS have the expected sign. Food provision has a positive sign, which reveals that people feel human wellbeing when agriculture provides society with food. However, it shows diminishing marginal utility, as revealed by the negative sign of the squared coefficient. This implies that a high level of food production provides decreasing levels of marginal utility; that is, increments in food production are expected to have higher positive effects on utility when production is low.

Similar statements could be applied to the case of water supply for irrigation. In relation to this attribute, people are aware of the importance of using water for agriculture, and they even consider that they get utility when some water is supplied to the agricultural sector. However, it should not be done on an unlimited basis. The negative sign of $WATER^2$ shows diminishing marginal utility, revealing that alternative uses for the water destined to irrigation could be preferred by the respondents under some circumstances.

The interaction between FOOD and WATER is also significant, revealing the negative relationship between them. The utility provided by food provision depends on the level of water supply for irrigation, and vice versa. Hence, high levels of water supply for irrigation reduce the marginal utility of food provision. As the water supplied for irrigation increases, there is a decline in the utility provided by food provision.

The temperature regulation (TEMPE) coefficient has a positive sign, which evidences that people also demand the cooling effect provided by agriculture, which could reach a 2 °C reduction in the case of the irrigated agroecosystems.

As expected, groundwater pollution (POLL) has a negative impact and shows diminishing marginal utility, this decrement being quicker as pollution increases. The estimated utility function also shows the significance of the interaction between WATER and POLL. High levels of groundwater pollution have negative effects on the marginal utility of the water supply for irrigation, and vice versa.

The respondents considered that agriculture provides an enjoyable environment that promotes recreational activities and tourism, as shown by the significant mean and standard deviation coefficients for the RECRE variable. Similarly, the agricultural contribution to biodiversity was also positively valued. The greater the bird richness in an agroecosystem, the more utility people get.

The cost coefficient has the expected negative sign, which shows the disutility people get when tax payments increase and thus provides consistency to the results.

4.2. Valuation of AES and AEDS

Table 4 shows the marginal WTP for AES and AEDS calculated with Model 3. The results indicate that, on average, people are willing to pay around 0.23 €/household/year in order to increase food provision. However, this WTP depends on the amount of food provided and the water supplied to achieve this level of food production. Therefore, low levels of food production and irrigation water supply will have positive and greater values of marginal WTP.

Regarding the water supply for irrigation, the results reveal that people are willing to pay for it, but they prefer that not all the available water is used for agricultural purposes. Actually, a negative WTP shows that people are willing to pay to reduce the water supply to irrigation to the level which maximises their utility: the satiation point, around 2600 m^3. For instance, if the water supplied to agriculture is around 2000 m^3/ha/year, people are willing to pay 0.02 €/m^3 to increase its availability for agriculture. However, if agriculture actually uses around 4000 m^3/ha/year, people are willing to pay 0.06 €/m^3 to reduce the water supplied for irrigation and promote alternatives uses. Furthermore, the marginal utility of WATER also depends on the level of FOOD –the higher the food provision, the less people are willing to pay to increase the water supply for irrigation– and the level of groundwater pollution –the higher the groundwater pollution, the less people are willing to pay to enhance the water supply for irrigation–.

The results contribute to the consideration of groundwater pollution as one of the AEDS. The WTP for this attribute is negative across all the levels considered. This has two direct implications: (1) the desired level of pollution is zero; and (2) people are willing to pay in order to reduce groundwater pollution. Moreover, this attribute shows diminishing marginal utility, revealing that the higher the pollution, the more people are willing to pay to reduce it. For instance, if groundwater pollution reaches the mean level (158 mg NO_3^-/L), people are willing to pay around 1.42 €/mg NO_3^- to reduce it.

People are willing to pay, on average, 13.21 €/year/household in order to support policies which contribute to a 2 °C temperature reduction. The WTP for supporting agricultural policies which contribute to biodiversity is, on average, 2.67 €/year/household per percentage point (p.p.) increment. Recreational and leisure activities within agricultural landscapes also contribute to wellbeing, and are valued at 117.64 €/year/household, on average.

The results of the WTP analysis allowed estimation of the value of agroecosystems with different provision levels of AES and AEDS. For this, the provision levels for the three most representative agroecosystems of the case study were obtained from Alcon et al. (2013) and Almagro et al. (2016) in the case of food production and water supply for irrigation of almond orchards, Albaladejo-García et al. (2020) in the case of temperature and following Perni and Martínez-Paz (2017) in the case of biodiversity. Groundwater pollution levels were obtained from the water authority in charge of the regional water management (CHS,

Table 3
Estimation results from MXL models.

	Model 1			Model 2			Model 3		
	MXL - Linear			MXL - Quadratic			MXL – All interactions		
Mean	*β*	SE		*β*	SE		*β*	SE	
FOOD	0.95	0.07	***	4.14	0.68	***	4.58	0.67	***
WATER	−0.13	0.02	***	0.62	0.12	***	0.73	0.12	***
TEMPE	0.00	0.04		0.08	0.05	*	0.07	0.04	*
POLL	−0.01	$5.43 \cdot 10^{-4}$	***	$-3.84 \cdot 10^{-3}$	$1.74 \cdot 10^{-3}$	**	$-3.76 \cdot 10^{-3}$	$2.06 \cdot 10^{-3}$	*
BIOD	0.01	$2.23 \cdot 10^{-3}$	**	0.01	$2.65 \cdot 10^{-3}$	***	0.02	$2.68 \cdot 10^{-3}$	***
RECRE	0.45	0.08	***	0.59	0.09	***	0.67	0.10	***
COST	−0.04	0.01	***	−0.06	0.01	***	−0.07	0.01	***
FOOD^2				−1.29	0.26	***	−1.33	0.25	***
WATER^2				−0.12	0.02	***	−0.11	0.02	***
POLL^2				$-1.34 \cdot 10^{-5}$	$4.90 \cdot 10^{-6}$	***	$-9.26 \cdot 10^{-6}$	$4.86 \cdot 10^{-6}$	*
FOOD*WATER							−0.10	0.03	***
WATER*POLL							$-3.82 \cdot 10^{-4}$	$1.75 \cdot 10^{-4}$	**
SD									
FOOD	0.64	0.09	***	0.02	0.39		−0.45	0.35	
WATER	−0.13	0.06	**	−0.03	0.11		0.02	0.03	
TEMPE	−0.35	0.10	***	−0.27	0.13	**	−0.29	0.12	***
POLL	$4.73 \cdot 10^{-3}$	$5.99 \cdot 10^{-4}$	***	$3.73 \cdot 10^{-3}$	$1.45 \cdot 10^{-3}$	***	$4.23 \cdot 10^{-3}$	$8.95 \cdot 10^{-4}$	***
BIOD	0.02	$4.58 \cdot 10^{-3}$	***	−0.02	0.01	***	−0.02	0.01	***
RECRE	0.76	0.13	***	−0.79	0.13	***	0.80	0.14	***
FOOD^2				−0.30	0.04	***	−0.24	0.10	***
WATER^2				−0.03	0.01	***	0.03	0.01	***
POLL^2				$1.29 \cdot 10^{-3}$	$3.02 \cdot 10^{-6}$	***	$9.19 \cdot 10^{-6}$	$2.53 \cdot 10^{-6}$	***
LL		−1785.37			−1732.25			−1725.48	
R^2-Adjusted		0.23			0.25			0.25	
AIC		3596.75			3502.50			3492.97	
BIC		3670.35			3610.06			3611.86	

Statistically significant at a level of *0.1, **0.05, and ***0.01.

Table 4
Marginal WTP for AES and AEDS (€/household/year).

	Mean	Confidence interval (95%)[4]	
FOOD (€/kg/ha)[1]	0.23	0.18	0.33
WATER (€/m^3/ha)[2]	−0.04	−0.05	−0.03
TEMPE (€/°C)	13.21	−3.74	30.74
POLL (€/mg NO_3^-/L)[3]	−1.42	−1.91	−1.09
BIOD (€/p.p.)	2.67	1.69	4.20
RECRE	117.64	78.33	177.39

[1] The marginal WTP of FOOD was evaluated at the mean levels of the attributes FOOD (1100 kg/ha) and WATER (3454 m^3/ha).
[2] The marginal WTP of WATER was evaluated at the mean levels of the attributes FOOD (1100 kg/ha), WATER (3454 m^3/ha) and POLL (158 mg NO_3^-/L).
[3] The marginal WTP of POLL was evaluated at the mean levels of the attributes WATER (3454 m^3/ha) and POLL (158 mg NO_3^-/L).
[4] Obtained using bootstrapping (1000 samples).

2017).

Table 5 summarises the decomposition of the global value obtained for each agroecosystem. The traditional irrigated agroecosystem was the most valued agroecosystem, with a TEV of about 988 €/household/year, followed by the rainfed agroecosystem (with a TEV of around 667 €/household/year) and the highly-intensive irrigated agroecosystem, which provides 500 €/household/year of utility to the people of the Region of Murcia.

Aggregating these values across the target population (539,000 households), the TEV could be calculated for each agroecosystem, as well as for the entire case study (Table 6). Thus, the whole agroecosystem provides more than 350 M€/year of human wellbeing, equivalent to 794 €/ha/year, which represents around 22% of the agricultural gross value added of the case study.

Table 5
AES and AEDS levels, CS and TEV. Valuation of agroecosystems (€/household/year).

	Rainfed agroecosystem		Traditional irrigated agroecosystem		Highly-intensive irrigated agroecosystem	
	Level	Value	Level	Value	Level	Value
FOOD	500 kg	345.31	1000 kg	538.98	2000 kg	538.96
WATER	0 m^3	0.00	2000 m^3	126.85	4000 m^3	−5.36
TEMPE	0 °C	0.00	−1 °C	13.21	−2 °C	26.42
POLL	25 mg NO_3^-/L	−9.38	125 mg NO_3^-/L	−75.75	250 mg NO_3^-/L	−219.47
BIOD	80%	213.86	100%	267.32	60%	160.39
RECRE	Yes	117.64	Yes	117.64	No	0.00
TEV		667.43		988.25		500.95

5. Discussion

5.1. Looking into the results

The aim of this work was to integrally value both AES and AEDS in order to integrate them into a common framework for agroecosystem valuation. Regarding the economic value of the AES and AEDS, the marginal WTP for food production, which could reach a maximum of 0.81[1] €/kg according to our results, is less than the market price received by almond farmers, which averages 1.32 €/kg (CARM, 2019). This reveals that people are not willing to support private benefits from agriculture (Jourdain and Vivithkeyoonvong, 2017), but they do value the contribution of agroecosystems to food security. Martínez-Paz et al. (2019) also found that, of the AES, fruit and vegetable production was the one valued most in the Huerta of the Region of Murcia. These authors showed that people living near the city of Murcia valued the contribution of this traditional agroecosystem to food provision at 6.83

[1] Marginal WTP of FOOD evaluated at the zero level of the attributes FOOD (0 kg/ha) and WATER (0 m^3/ha)

Table 6
TEV extension. Valuation of the agroecosystems in the Region of Murcia.

	Rainfed agroecosystem	Traditional irrigated agroecosystem	Highly-intensive irrigated agroecosystem	Total Region of Murcia
Area (ha)	253,269	48,077	139,757	441,103
TEV (€/year)	206,555,381	58,056,779	85,549,348	350,161,508

€/household/year. This value contrasts with the results obtained in this study for the traditional irrigated agroecosystem (538.98 €/household/year), since our results consider the agricultural contribution to food security.

The supply of water for irrigation is socially supported, and the WTP can reach 0.12[2] €/m^3. This value represents, on average, one-third of the current price paid by farmers (CCRC, 2019). These results imply people are willing to support the use of water for irrigation. However, the diminishing marginal utility means that this WTP will depend on the current level of water supply for irrigation, as well as the level of food provision and groundwater pollution. It also means that this economic value could become negative, which indicates that the use of additional water for irrigation would be translated into a social cost. The satiation point for WATER, around 2600 m^3, establishes the boundary between positive wellbeing and social cost. Rigby et al. (2010) also estimated the value of irrigation water for farmers in the Region of Murcia using a DCE and obtained a mean WTP of 0.45 €/m^3. Therefore, it seems the private value of irrigation water is higher than its public value.

The value of agricultural services regarding climate regulation has been estimated in most cases according to the social demand for reduction of CO_2 emissions, or the improvement in CO_2 sequestration due to agricultural activity (Granado-Díaz et al., 2019). However, people do not perceive these flows as an agricultural impact on their wellbeing, but they do perceive changes in the local temperature as an agricultural effect on climate regulation. In fact, we estimate that people are willing to pay around 13 €/household/year per degree of temperature reduction.

The literature concerning the non-market value of groundwater pollution is scarce, but there are many reports related to the economic valuation of water quality in agricultural contexts. Niedermayr et al. (2018) estimated that the value of groundwater which is able to be used without treatment ranges between 66 and 87 €/household/year, in an area in the northeast of Austria. Similarly, Jourdain and Vivithkeyoonvong (2017) estimated a value of 64 €/household/year in the case of its use for swimming purposes. These values are in line with the ones obtained in this work.

Contrastingly, biodiversity is one of the AES most employed for agroecosystem valuation using DCEs (e.g. Vaissière et al., 2018; Granado-Díaz et al., 2019). We found a WTP of 2.63 €/household/year per p.p. increment in bird diversity, very close to the values of Rodríguez-Ortega et al. (2016), who estimated that the value associated with the presence of bearded vultures in mountain agroecosystems in northeast Spain ranges between 1.82 and 2.08 €/household/year per percentage point of increment. However, our value contrasts with the one obtained by Perni and Martínez-Paz (2017) for a human-created wetland located close to the case study site: 0.18 €/household/year per percentage point of increment. These differences could be due to the extent of the two ecosystems: our work focused on all agroecosystems within the Region of Murcia, while that of Perni and Martínez-Paz (2017) was centred on one specific wetland ecosystem.

Finally, the enjoyment of leisure and recreational activities within agricultural landscapes is valued at about 110 €/household/year according to our results. However, comparison with the values obtained in other works located near to the study area, 2.85 and 2.81 €/household/year for García-Llorente et al. (2012) and Martínez-Paz et al. (2019), respectively, suggests our value is an overestimation. These differences could be related to the fact that our work focused on different agroecosystems, which include activities such as wine tourism (Cebrián and Rocamora, 2017), ecotourism and environmental education (Robledano et al., 2018), together with sport activities. Nevertheless, this result reveals that agricultural management should also include culture-friendly approaches, to integrate all dimensions of human wellbeing.

The results presented here encompass the non-market valuation of the AES and AEDS provided by the agroecosystems studied. However, their benefits and costs for society could broaden beyond the scope considered in this study. The market valuation of trading agricultural outcomes could also be integrated with the non-market values estimated here. Therefore, of the AES, food provision is the one which provides both market and non-market values to society. The market value of food provision could be summarised by the gross margin, as an indicator. For instance, assuming the almond gross margins for the rainfed, traditional and highly-intensive irrigated agroecosystems to be around, respectively, 350, 1000 and 1500 €/ha/year (Alcon et al., 2013; Lehtonen et al., 2020), the integrated market and non-market value of each agroecosystem rises to, approximately, 1150, 2200 and 2100 €/ha/year, respectively.

Consideration of the market values of AES provides an additional perspective on the integrated contributions of agroecosystems. In fact, these values reinforce the results showing that differences in productivity cannot overcome differences in the values of AES and AEDS. The market and non-market values are lowest for the rainfed agroecosystem; however, the two irrigated agroecosystems have similar values. This reveals that greater provision of AES -and lower provision of AEDS- by traditional irrigated agroecosystems compensates differences in productivity with respect to highly-intensive irrigated agroecosystems. The integrated market and non-market values provided by both irrigated agroecosystems show that similar values could be reached with greater AES and lower AEDS. Hence, this illustrates again that higher food production is not always socially desired, but it must be considered in the overall contributions to human wellbeing.

5.2. Policy implications: Because AEDS matter

The production of enough healthy food for a growing population, while mitigating negative impacts on ecosystems and human wellbeing, is the main agricultural challenge for the next decade (Sandhu et al., 2019). This implies the integration of multiple contributions, both positive and negative, of agriculture to human wellbeing. Hence, the results of this study provide evidence that AEDS should be valued integrally together with AES. As Shackleton et al. (2016) pointed out, not considering AEDS when valuing agroecosystems may produce an overvaluation. For instance, for our results (Table 7), this overvaluation could reach 44% of the TEV of the highly-intensive irrigated agroecosystem.

At this point, the results of the present work may serve as a decision-support tool for agricultural policy makers, to improve the design and implementation of agri-environmental policies, either *ex-ante* or *ex-post*, in Mediterranean regions with water-scarcity issues. Since the results highlight the main agricultural contributions to human wellbeing and their intensity, they could be used to define policies and measures which promote these positive contributions and reduce the negative ones. In addition, this framework also allows measurement of the *ex-post* impact of agricultural policies on human wellbeing. A simple simulation exercise allows estimation of the economic values of greening actions

[2] Marginal WTP of WATER evaluated at the zero level of the attributes FOOD (0 kg/ha), WATER (0 m^3/ha) and POLL (0 mg NO_3^-/L).

Table 7
Relative importance of AES and AEDS in the TEV of different agroecosystems.

	Rainfed agroecosystem	Traditional irrigated agroecosystem	Highly-intensive irrigated agroecosystem
FOOD	0.52	0.55	1.08
WATER	–	0.13	−0.01
TEMPE	–	0.01	0.05
POLL	−0.01	−0.08	−0.44
RECRE	0.18	0.12	–
BIOD	0.32	0.27	0.32
TEV	1.00	1.00	1.00

described in the last CAP reform. These are expected to increase biodiversity (15%) and to reduce groundwater pollution (25%) (due to the reduction in fertiliser needs). Thus, based on the current situation, the impact of these measures is expected to range from 23.51 €/ha/year in the case of the traditional irrigated agroecosystem to 103.01 €/ha/year for the highly-intensive irrigated agroecosystem.

Thus, the positive impact of the greening practices will depend on the agroecosystem considered, which reveals that the efficiency of different agricultural policies would be higher if they were directed to the right agroecosystem. If only the criteria for the gain in wellbeing are considered, agri-environmental policies may focus on more degraded agroecosystems, where the expected impact is higher. However, this may imply the allocation of economic resources to those agroecosystems which pollute more, instead of rewarding the ones which actually perform better. This illustrates the challenge that arises in the design of agri-environmental policies, regarding not only socio-economic but also ethical issues, which requires the consideration of multidisciplinary and intertemporal approaches (Varela et al., 2018).

5.3. *Theoretical implications: diminishing marginal utility, social demand and the interdependence of AES/AEDS*

The results make it clear that attributes may not be as independent from each other as we may think. In theory, in the design of a DCE, it is considered that all attributes are independent (Hensher et al., 2005). However, this assumption may not be realistic when applied to the case of AES and AEDS. In fact, the perceived marginal utility of food provision or groundwater pollution also depends on the water supply for irrigation, and vice versa. Fig. 2 summarises, ceteris paribus, the contributions to the total utility function of the AES and AEDS whose indicators are continuous. It also reveals this dependence among attributes.

The impact of each of the AES and AEDS on human wellbeing is not linear, and also depends on the level of provision of other AES or AEDS. Food provision was expected to have a linear, positive impact on wellbeing; that is, the more food agriculture produces, the more wellbeing people get. Nevertheless, the results reveal that people prefer agroecosystems that provide food up to the satiation point (1600 kg/ha). Translating this into the ecosystem service approach, it shows that food provision will be considered as one of the AES while the level of food provision is below this maximum - that is, while increasing the level of food provision generates positive marginal utility. This is clearly defined by the concave form of the total utility function for *FOOD* (Fig. 2).

High levels of food production are usually linked to high levels of water supply for irrigation, and this is socially perceived as well. At this point, it is necessary to differentiate between *interaction* and *confusing* effects (Karaca-Mandic et al., 2012). The decline in the contribution of food provision to the total utility after its satiation point is related to this *confusing* effect, since high levels of food production are *confused* with high levels of water supply for irrigation and this induces the decrement

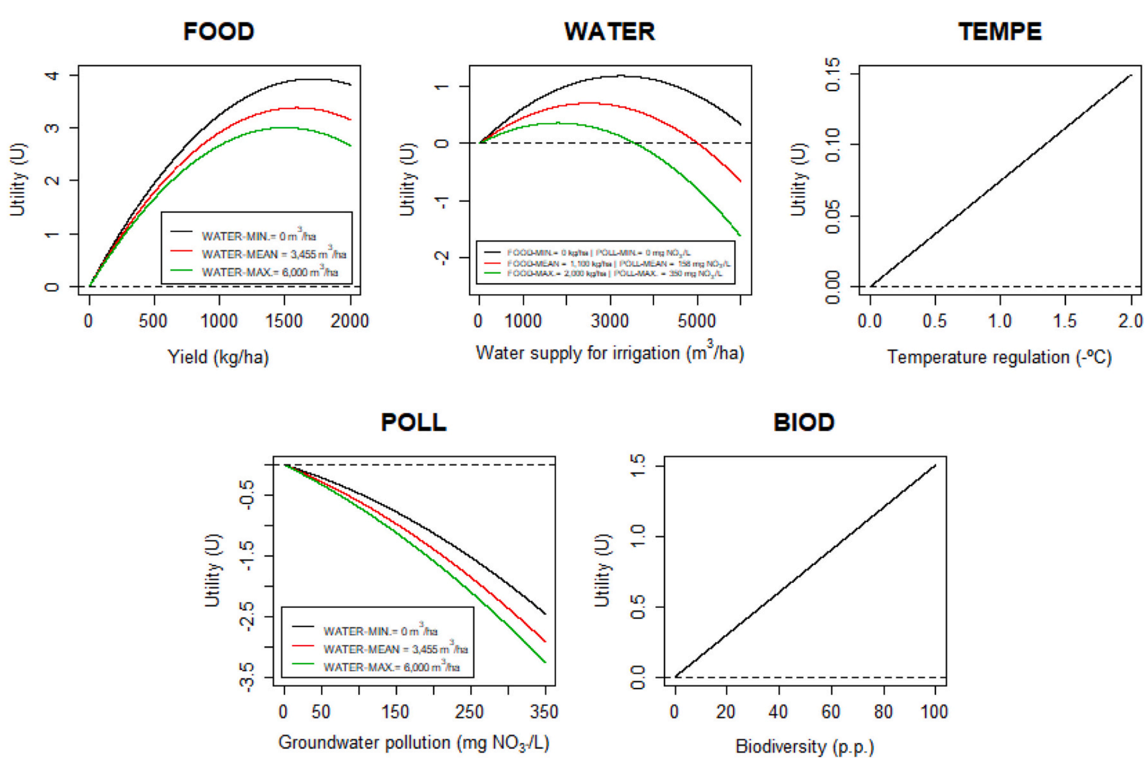

Fig. 2. Utility functions of AES and AEDS.
Note: The utility provided by each of the AES and AEDS is presented. In particular, the utility provided by food provision (FOOD), water supply for irrigation (WATER) and groundwater pollution (POLL) depends on the value taken by some other attributes. This interdependence is shown in their respective graphs by the black (minimum expected level of the interdependent attributes), red (mean expected level of the interdependent attributes) and green (maximum expected level of the interdependent attributes) lines. The utility provided by temperature regulation (TEMPE) and biodiversity (BIO) does not depend on other attributes. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

in utility (see Appendix B). These results highlight that maximisation of food provision alone should not be the main focus of agricultural policy.

Similar statements could be applied to the case of water supply for irrigation. Supplying water to agriculture will provide wellbeing until its satiation point is reached. However, if water is supplied to agriculture at a level higher than this maximum, it will be considered as one of the AEDS. This is linked to the water scarcity in the case study area and reveals that, alternatively, water may be supplied to other ecosystems rather than agroecosystems. A similar situation was exposed by Zabala et al. (2019) for the competitive use of reclaimed water in agricultural irrigation or for environmental purposes. The utility provided by supplying water for irrigation is related to the level of food provided by such agroecosystems and the level of groundwater pollution. Thus, the higher the food provision and groundwater pollution, the lower the utility that people get from supplying water for irrigation (Fig. 2), which reveals the social trade-offs among these AES and AEDS.

The agricultural contribution to groundwater pollution shows diminishing marginal utility. However, in this case, the utility is always negative, independently of the pollution level. This shows that groundwater pollution due to agricultural activity is always considered as one of the AEDS; thus, the socially-demanded level of groundwater pollution is zero. As the interaction term between *WATER* and *POLL* shows, the disutility obtained from pollution will be higher when it is perceived jointly with the water supply for irrigation. Nutrient leaching from irrigation water to groundwater is responsible for the poor ecological status of several water bodies in the case study area (Pellicer-Martínez and Martínez-Paz, 2016). Hence, there is a societal awareness, reflected in the social demand, of the physical relationship that may arise between irrigation water and groundwater pollution.

The temperature regulation and biodiversity are considered as AES, since the results show a linear, positive relationship between provision and utility.

As these results reveal, agricultural outputs can switch from AES to AEDS depending on their provision level. This idea was first presented by Rasmussen et al. (2017), and was applied to agroecosystems in Laos. A further step forward will be achieved here with the new categorisation of AES and AEDS that we propose. Thus, three main categories of AES/AEDS are suggested: (1) pure AES, for which the more that is provided, the more utility people get; (2) pure AEDS, for which the more that is provided, the more disutility people obtain; (3) quasi-AES, whose positive or negative impact on human wellbeing depends on their provision level. With this categorisation, food provision, temperature reduction and contributions to biodiversity and leisure and recreational activities can be considered as pure AES. By contrast, groundwater pollution can be placed in the pure AEDS category while, between the two extremes, water supply for irrigation may be placed in the quasi-AES category. This also evidences that AES and AEDS are not static concepts, but are context-dependent (Shackleton et al., 2016).

The theoretical implications of these results have been extrapolated to the case of marginal WTP functions (Fig. 3) and values. Thus, pure AES are related to positive and constant, or even rising, WTP values, while pure AEDS imply negative and constant, or decreasing, values. Quasi-AES have decreasing WTP functions, which could have positive and negative sections.

6. Conclusions

The contributions of agroecosystems to human wellbeing have been addressed here. In the main, previous studies of agroecosystems and their outputs have been focused only on the positive contributions of agriculture to human wellbeing. Hence, little is known about the negative impacts, in neither biophysical nor social terms. This study

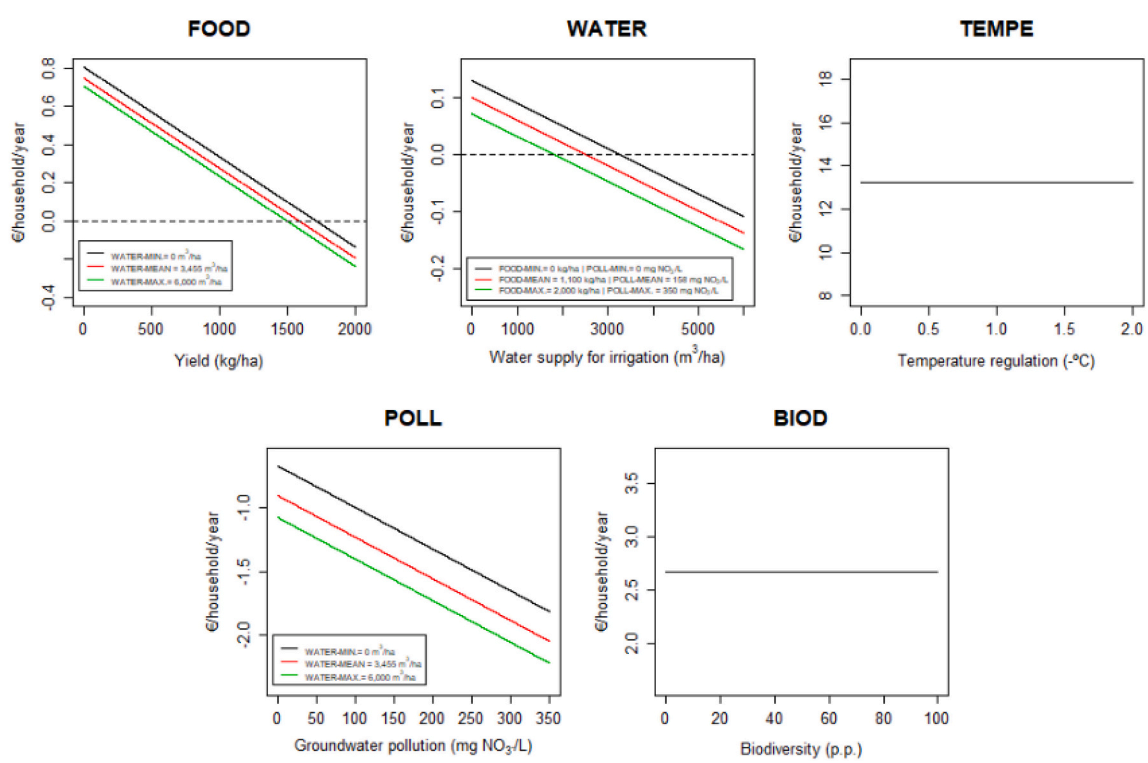

Fig. 3. Social demand. Marginal WTP functions of AES and AEDS.

Note: The marginal WTP for each of the AES and AEDS is presented. In particular, the WTP for food provision (FOOD), water supply for irrigation (WATER) and groundwater pollution (POLL) depends on the value taken by some other attributes. This interdependence is shown in their respective graphs by the black (minimum expected level of the interdependent attributes), red (mean expected level of the interdependent attributes) and green (maximum expected level of the interdependent attributes) lines. The marginal WTP for temperature regulation (TEMPE) and biodiversity (BIO) does not depend on other attributes. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

addresses the need for an integrated framework which gathers together both positive and negative agricultural outputs: namely, AES and AEDS, respectively. For this purpose, an integrated economic valuation of the AES and AEDS provided by the agroecosystems of the Region of Murcia (south-eastern Spain) has been developed. A DCE has been employed to reach the pursued aim, using food provision, climate regulation, recreational and leisure activities and biodiversity as AES, and water supply for irrigation and groundwater pollution as AEDS.

The results show that people value both AES and AEDS, which provides a net economic valuation of the overall impact of agriculture on human wellbeing. As such, the people surveyed showed non-linear preferences for food provision, water supply for irrigation and groundwater pollution, which disclose diminishing marginal utility for these AES and AEDS. This finding also suggests that the marginal value (WTP) of these AES and AEDS depends on their provision level. Thus, social demand functions for their provision could be estimated, to calculate the value not only of each of the AES and AEDS, but also of the entire agroecosystems in the case study. Therefore, this work presents a novel framework for measuring the overall value of agriculture to society, assessing all contributions to human wellbeing.

These results will be very useful for policy makers in the development of sustainable and cost-effective agricultural measures. New agricultural policies need to deal with the environmental impacts of agricultural activity without overlooking food production and the consumption of natural resources. This could be translated into new, socially-supported agricultural policies, with agricultural practices that promote water saving, pollution mitigation, biodiversity and climate regulation. Thus, further studies may analyse the provision of AES and AEDS from a supply point of view (farmers), to explore both trade-offs and economic value and integrate them with the current assessment framework.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolecon.2021.107008.

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