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Agri-environmental policies in the EU and United States: A comparison☆

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ABSTRACT

Agri-environmental policies (AEPs) in the United States and the European Union are examples of payments for environmental services that pay farmers to reduce the negative externalities of agricultural production, while serving as a means to transfer public funds to farmers. We show that despite similar origins, AEPs in the two regions differ both in their specific objectives and in their implementation. For example, AEPs in most member states of the EU-15 have the additional objective of using agriculture as a driver for rural development. This objective is achieved by compensating farmers for the private delivery of positive public goods, such as attractive landscapes, produced by agriculture. The rationale is market failure, and there is empirical evidence that Europeans are willing to pay for such positive externalities. No comparable provision exists in U.S. policy. By contrast, U.S. AEPs focus almost entirely on reducing agriculture's negative externalities, such as soil erosion. Second, we find that U.S. programs are more targeted than their EU counterparts, and take opportunity cost into account. The EU programs, on the other hand, address a wider range of externalities, and are focused more on the paying for a particular farming process than reducing specific negative externalities. The EU takes a broader view of AEPs than does the United States, both in terms of type of activity that can be funded, and by using less targeting by land characteristics, and so the European program could be more easily used as a mechanism for transferring income to producers. Despite this, we find evidence that many of the amenities targeted by the programs are demanded by the population.

1. Introduction

In the past few years, both Europe and the United States have placed increasing emphasis on programs that compensate farmers for the provision of environmental services (Ilbery and Bowler, 1998). The programs sponsor environmental services targeted at reducing negative externalities, such as nutrient run-off and soil erosion and an increase in positive externalities, such as scenic vistas, or, less tangibly, the spiritual and symbolic value of preserving a farming heritage (Bernstein et al., 2004; Mullarkey et al., 2001). The economic rationale for these programs is that they address a market

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failure; the provision of such unmarketed environmental services implies that the farmer is deviating from the most lucrative use of the resources at his or her disposal, and should therefore be compensated. AEPs are also used as trade-friendly means to transfer income to producers, for example to encourage farmers to remain on the land even though no output is produced. Despite the parallel growth in AEPs, the two regions have taken very different approaches to paying farmers for environmental services (Baylis et al., 2004; Bernstein et al., 2004). The goal of this paper is to highlight some of these differences, and briefly propose reasons for them. These differences have implications both for how efficiently AEPs provide environmental services, and for their relationship to other farm subsidies.

In the United States, environmental payments have been administered through the Conservation Titles of recent Farm Bills. In Europe, they have been incorporated into Pillar 2 of the Common Agricultural Policy (CAP). Important adjuncts are the cross-compliance provisions associated with the commodity payment section (Pillar I of the CAP) and certain components of the Commodity Title of the Farm Bills. Though compensation for meeting the standards required by cross-compliance would not be classified as "payment for environmental services (PES)," under the definition of this concept adopted by the Center for International Forestry Research (CIFOR, 2005), we consider that cross-compliance linkages, by requiring producers to meet minimum environmental standards before becoming eligible for any farm payments, play an important role in ensuring the efficient delivery of environmental services. To avoid confusion, we will refer below to *agri-environmental policies* (AEPs) rather than PES, with the understanding that this class of policies – incorporating the Conservation Title, most of Pillar 2 and cross-compliance – may sometimes be broader than the class that satisfies CIFOR's definition as it applies to agriculture. For example, CIFOR's definition requires that the environmental service be 'well-defined'. AEPs are frequently not well-defined; for example payments may be made for the adoption of a certain technology, even though the nature of the environmental good that adoption would promote may be unspecified. It is also uncertain whether some European AEPs meet CIFOR's conditionality requirement. While the objective for some of the EU programs is to reduce environmental pollution from agriculture, the actions that farmers are paid for do not always produce the environmental objective. For example, while producers may be paid to reduce their use of chemical inputs, the desired environmental service (for example, reduced nitrogen run-off) may not be achieved. In this sense, the payment is not conditional on the environmental service.

In Europe, the shift from direct commodity payments towards agri-environmental and rural development programs has been a cornerstone of the region's response to the Uruguay Round. The linkages between agriculture, the environment and the development or management of rural areas have become encapsulated in the concept of multi-functionality[1]. The European Union has taken the position that these additional benefits are typically not marketable and, consequently, would be under-produced relative to the levels desired by society were it not for agri-environmental payments (Ervin, 1999). In the view of some commentators, however, this justification is nothing more than a repackaging of domestic subsidies, pre-Uruguay Round style (Agra Europe editorial, 2001).

The U.S. administration has also used agri-environmental policy as a way to comply with WTO provisions. Funding for conservation programs was substantially increased and new AEPs were introduced in the 2002 Farm Bill. While motivated by concerns similar to the EU's, the U.S. approach to agri-environmental policy is quite different. We suggest that European programs focus on both the positive and negative externalities resulting from agricultural activity. European farmers are rewarded both for the public goods they provide, such as preservation of alpine pastures, and for reducing negative externalities by, for example, promoting organic production. As we explain below, European policy assumes that less intensive farming reduces input use, and therefore negative externalities. This assumption explains the EU's encouragement of methods such as organic farming which require fewer external inputs and therefore are assumed to result in less environmental damage than conventional farming techniques. In contrast U.S. policy pays more directly for the attainment of environmental goals, regardless of the methods used to achieve these outcomes. As an illustration, while EU policy pays farmers to reduce the number of animal units per land area as a means of reducing nitrogen surplus, U.S. policy pays farmers to reduce nitrogen surplus, whether they reduce their stocking rate or they invest in manure storage facilities for their intensive cattle feed-lot.

In Section 2 we compare and contrast agri-environmental policies in the two regions. Specifically, we ask how closely the policies in the two regions represent payments for environmental services, and discuss their efficiency in providing those outputs. We begin by outlining the origins and motivations for the programs, and then discuss various program characteristics, including the services identified, and the mechanisms by which they are targeted. We then move to questions of seller selection and opportunity cost, baseline and additionality, and unintended externalities produced by the policies. Section 3 considers possible reasons for these differences. We conclude with several implications of these differences for the production of environmental services.

2. Comparisons

2.1. Origins and rationale

In both the EU and the United States the conservation of critical natural capital is considered to be a legitimate task of government. The fore-runner of the modern Conservation Reserve Program (CRP) began in the 1930s to protect soil (and reduce production of certain crops that were in excess supply) while in Europe agri-environmental policies were not developed until the 1980s.[2] In both regions, agri-environmental

[1] See Garzon (2006) for a discussion over the use of this term; and Evans et al. (2002) for a critique from a geographical perspective.

[2] Brouwer and van der Staaten (2002, p5) give a detailed description of the development of AEPs in the EU.

programs have had the accompanying objective of transferring public funds to farmers. For example, the current CRP acreage cap is about 10% of U.S. cropland. Taking such a large acreage out of production has positive effects on commodity prices. Duffy et al. (1994) and Barnard et al. (1997) show that these payments have now been largely capitalized into the value of farmland.

In the United States, while the 2002 Farm Bill substantially increased the size and scope of conservation programs, other legislators (with quiet support from the George W. Bush administration) proposed moving even more funds from direct price support to AEPs. However, unlike their EU counterparts, the traditional farm lobby groups, along with their representatives in Congress, have been militant in their resistance to the administration's approach. In both the House and Senate, amendments were proposed to transfer large sums of money ($2 billion per year in the House amendment) from commodity programs to conservation programs. Both these amendments were unsuccessful largely due to opposition from members representing farm constituencies. In the end, the 2002 Bill did increase funding for conservation programs and introduced two new programs, the Grassland Reserve Program (GRP) and the Conservation Security Program (CSP), but it also increased funds for commodity programs such as the new countercyclical program, and increased loan rates.

As a coordinated policy, the EU's AEPs originated in the 1992 MacSharry reforms of EU agriculture. A number of writers have described various political motivations behind the MacSharry reforms: the depressed commodity markets of the 1980s, pressure from the United States over GATT, recurrent budgetary crises in the EU caused by high levels of spending on price supports, and the projected eastward enlargement of the EU (Kay, 1998; Moyer and Josling, 2002). Further reforms followed in 2000 and 2003, and the 'Single Farm Payment' has now replaced much of the production subsidy apparatus. One implication that can be drawn from this is that AEPs are at least partially a form of compensation to European farmers for declining price supports. This compensation may have been required as a result of the consensus tradition within EU decision-making. AEPs provide the more agriculturally conservative member states, such as France, with at least a halfway house on the route to liberalization.

AEPs have also been used by ministries or departments of agriculture in both regions to make externally-imposed environmental regulations more palatable to farmers. In the United States, many of these regulations have been imposed by the Environmental Protection Agency, while in the EU they have often been introduced at the member state level. For example, in 2000 the Netherlands budgeted EUR 800 million to buy out pig production quotas, with the intention of reducing the national manure surplus (OECD, 2003).

The AEPs may also be thought of as partial compensation for cross-compliance. Cross-compliance became compulsory in the EU for farmers receiving Pillar 1 payments in 2005, after the failure of more voluntary approaches (Osterburg, 2004). Each member state now sets its own minimum Good Farming Practice (GFP) level, and to receive Pillar 1 government payments, including whole farm payments and price supports, all farmers must meet at least this level of environmental practice (EC, 2005).[3] While making the effort to achieve GFP is voluntary, it is in reality compulsory because few farmers would be able to continue in business without Pillar 1 money. Thus, for farmers who considered Pillar 1 subsidies as a 'right', GFP consisted of a form of a regulatory taking. However, if farmers (slightly) exceed the GFP baseline, they can access additional subsidies in the form of AEPs as a carrot.

Similarly, in the United States, conservation programs were partially a form of compensation for increased regulation. One of the first environmental regulations was the introduction of the cross-compliance provisions. Conservation and environmental groups argued for conservation compliance, and succeeded in including Sodbuster and Swampbuster into the 1985 farm bill. The agricultural sector accepted these compliance provisions as a compromise in order to win acceptance of the CRP, which was also introduced in 1985. There was, however, a general sense among producer groups that conservation compliance would soon be abolished. Instead, compensation for Swampbuster was introduced in 1990 in the following farm bill with the introduction of the Wetlands Reserve Program (WRP). More recently, the Wildlife Habitat Incentives Program (WHIP) was introduced, in part as a response to the Endangered Species Act (ESA). Farmers had raised strong objections to the ESA and the restrictions it would place on their activities, and so the Department of Agriculture made it easier for producers to comply with retaining and improving habitat by compensating producers if they did so. Most recently, in 2002, the EPA and USDA imposed new, stricter regulations on nutrient management from large livestock operations. In the same year, these operations were declared eligible for funds under the Environmental Quality Incentives Program (EQIP). If these regulations could have been imposed without the offer of compensation, many of the environmental benefits currently associated with the conservation programs might have been received without the additional payments. Moreover, by paying farmers to reduce their pollution, these additional payments certainly go against the "polluter pays" philosophy which nominally applies in both regions, and potentially fails the "additionality" criterion included in CIFOR's definition. That said, whether it would have been politically feasible to introduce and maintain a number of these environmental regulations without providing offsetting compensation remains an open question. Because of the long history of income support for agriculture, it has been difficult to implement the polluter pays principle in this sector.

2.2. Services targeted

While the motivations for AEPs in the two regions are very similar, there are some striking differences in outcomes, both in terms of the services targeted by the programs, and their implementation.[4] The EU has taken a wider view of what

[3] There is empirical evidence from Spain that cross-compliance helps to ensure delivery of contracted environmental goods, although not necessarily at a socially-optimum price (Varelo-Ortega et al., 2004).

[4] See Pretty et al. (2001) for detailed analysis of the costs and benefits of internalizing agriculture's externalities in both regions.

constitutes an agricultural externality; in particular, many aspects of traditional farming, such as terraces, stone fences, and the raising of certain rare breeds of farm animals, are perceived as being desirable outcomes in and of themselves. Because these public goods are being supplied privately by farmers, EU member states consider it legitimate to offer compensation in return for their provision (MAFF, 2000). This EU position appears to have considerable public support. There is a variety of empirical evidence indicating that Europeans genuinely value these environmental services. Surveys by Eurobarometer (EC, 2004) have shown a consistently high appreciation for the work of farmers in cultivating the landscape; Drake (1992) discusses the non-market value of Swedish agricultural landscape, while Hackl and Pruckner (1997) studied the willingness to pay (WTP) of rural tourists in Austria and found that WTP exceeded local agri-environmental subsidies, even on a very conservative basis.

There is no obvious parallel in the United States to the EU's payments for positive externalities. In contrast, the bulk of U.S. conservation expenditure targets the reduction of the negative externalities produced by agriculture. Until the mid-1980s, U.S. agri-environmental policy concentrated on preventing the loss of topsoil. Since then, the focus has widened to include the, reduction of agricultural water pollution, and ensuring that farming does not result in the draining of wetlands and the loss of wildlife habitat (Claassen et al., 2008-this volume). Land retirement through the CRP, and more recently the WRP, was the major policy tool until the 2002 Farm Bill, taking about 90 cents of every conservation dollar going to farmers (USDA-ERS, 2001; Baylis et al., 2004). Conservation programs at the State level reflect the same emphasis as federal programs, i.e., the reduction of the negative externalities of agriculture, rather than the encouragement of public goods. For example, New Jersey provides technical assistance for country-based soil surveys, and help for limiting non-point source pollution (New Jersey, 2006). Minnesota offers a wide range of loans, grants and cost-sharing to help residents 'acquire, manage, conserve and protect natural resources' (Minnesota, 2006), but the amount of money available is not very large; clearly, the major conservation funding comes from federal sources. There are some interesting exceptions to the federal concentration on negative externalities, including programs designed to prevent loss of farmland to development, such as the Farm and Ranch Land Preservation Program (FRLP) discussed below (USDA-ERS, 2003).

These differing views of the nature of agricultural externalities are reflected in the ways the two regions approach the issue of environmental stewardship. The direction which EU agri-environmental policy has taken indicates that in parts of Europe, land is perceived to attain its highest environmental value when used for farming. For example, where farming is unprofitable, a significant fraction of EU agri-environmental payments is targeted at limiting land abandonment (Baldock et al., 1996). On the other hand, the direction which U.S. policy has taken indicates that in large parts of the United States, land is perceived to attain a higher environmental value when it is taken out of farming and returned to its natural state (see Dobbs and Pretty (2004) for comparison of environmental values in both regions; Hellerstein (2002) for the United States). Thus, while European policy attempts to limit land abandonment, especially in the south, farmers in the United States are often specifically paid to return farmland to its native state, principally through the CRP and WRP. An exception is the Farm and Ranch Lands Protection Program (FRLP), which was previously known as the Farmland Protection Program, renamed in the 2002 Farm Bill. The FRLP provides matching funds to State, tribal and local governments and non-government organizations for the purchase of conservation easements, with the aim of helping farmers and ranchers to keep their land in agriculture. The purchase of the conservation easement goes towards compensating landowners for voluntarily limiting future use of their land for non-agricultural uses. The budget for the FRLP is relatively small at $91 million for 2004, or 2.7% of Farm Bill Conservation Programs, but the planned increase in funding is large: nearly twenty-fold. The underlying purpose of the program is to limit urban sprawl; before the program can be used to purchase the development rights for a parcel of land, a bid from developers is required. As one might expect, allocations to States vary greatly, reflecting different pressures on agricultural land. In 2006, New Jersey was allocated 5.6% of total funding, while Louisiana received only 0.3% (USDA, 2004). There is a contrast here to EU programs which are designed to counter land abandonment in remote and mountainous areas, especially in the southern member states (Baldock et al., 1996). In the EU, the perceived threat to the land is that it will return to scrub as opposed to being turned into housing developments (Pinto-Correia and Mascarenhas 1999). In contrast the EU is attempting to retain the 'cultivated' look of the European countryside, which some studies have shown to be attractive to tourists (Drake, 1992; Hackl and Pruckner, 1997; Schmid et al., 2007). Differences in policy between the two regions might be attributed to different factor endowments, such as land; the EU has 130.4 million ha of agricultural land with a population of 378 million, while the United States has an agricultural land base of 377 million ha, and a population of 285 million. However there seems to be little evidence of a correlation between relative land scarcity, intensive agriculture and associated environmental problems either between member states of the EU or between the EU and the United States (Baylis et al., 2004).

EU policy acknowledges that agricultural production generates some negative externalities but the specific externalities addressed differ from those in the United States. EU agri-environmental policy focuses mainly on externalities that are by-products of the intensification of farming – i.e. the use of too many non-land inputs per unit of land – whereas U.S. policy focuses mainly on the by-products of extensification – i.e. the use of excessive amounts of (environmentally sensitive) land (Baldock et al., 2002). Indeed, when European agricultural stakeholders were asked to identify the most important environmental concerns associated with agriculture, they listed water pollution caused by chemical leaching, unsustainable levels of irrigation, air pollution, and a decline in agro-biodiversity as primary issues. All of these sources of pollution are linked to intensive agriculture. Stakeholders also specifically raised the concern that "high nature-value" farming systems were being lost either to intensification or abandonment (Baldock et al., 2002). Unsurprisingly, these are precisely the types of pollution that are addressed by EU agri-environmental programs. On the other hand, the negative environmental externalities targeted by U.S. conservation policy are typically associated with increased use of farmland that is marginal, either because it is highly erodible or has been reclaimed by draining wetlands. In addition, the United States has a number of programs that subsidize farmers to reduce the amount of pollution while retaining intensive production systems. For example, EQIP subsidizes farmers who install manure storage facilities and fencing systems, both of which allow higher stocking rates. In comparison, various member states in the EU address manure management by subsidizing farmers for decreasing their stocking rates (see Table 1).

2.3. *Degree of targeting of environmental services*

Payments for environmental services can also be categorized by how they target their objectives. At one extreme, programs can pay producers on the basis of the measured quantity of the environmental service produced; at the other extreme, program payments may be tied exclusively to input or technology use. The U.S. CRP uses a very detailed targeting instrument, the Environmental Benefits Index, which imputes the environmental services expected from ceasing to farm a certain field (see FSA (1999) for methodology; Babcock et al. (1997) for efficiency calculations for targeting of CRP and a comparison of alternative tools). To receive agri-environmental payments in the EU, by contrast, it is typically sufficient to commit to using agricultural inputs and/or technologies that have been designated as environmentally friendly. The same payment is received regardless of whether those techniques are used on particularly sensitive land (e.g., a riparian area) or on a parcel where the technology change will have little environmental impact. In some instances, input choices themselves are deemed to have intrinsic environmental merit. For example, European farmers receive funding for enhancing the welfare of their livestock and/or farming in accordance with traditional methods. In other instances, input choices are subsidized because they are expected to result in a portfolio of valued environmental outputs. A recent EC document goes so far as to say: "adherence to organic production standards is more likely to lead to general improvements in environmental quality than the production of specific environmental goods" (Lampkin, 1999, p. 40). Table 2 presents a categorization of inputs and outputs, showing that no EU agri-environmental programs explicitly target measurable environmental outputs.

The CRP's higher degree of targeting has a larger information requirement. To calculate the EBI, for example, the U.S. government needs information on environmental characteristics of the applicant fields, and detailed information on the benefits produced by any one or a combination of actions (such as retiring the land alone, versus retiring the land and planting native grasses). It also requires a specific weighting of objectives. Targeting the input alone does not have such demanding requirements for regional environmental data and scientific study.

The degree of targeting will determine how well a program meets its objectives (see Hodge (2000) for an analysis), but there may be cases where one wants to obtain a wide variety of benefits that are difficult to quantify and/or to measure. For example, suppose manure can best be managed by a highly-intensive livestock operation with state-of-the-art manure storage facilities, but this process produces unquantifiable negative "landscape" externalities. Further, assume that encouraging the producer to reduce his stocking rates will reduce manure run-off, but less effectively than the manure storage facility; however, lower stocking rates will produce positive landscape externalities. In this case, targeting the measurable externality, manure run-off, may result in fewer environmental services than targeting the input. A closely related issue is that agri-environmental programs in the two regions tend to differ in their degree of focus. Relative to the United States, EU programs are oriented towards multiple, sometimes nebulous goals. For example, these objectives include protecting the biosphere, keeping farmland from being abandoned, and preserving various broadly defined

Table 1 – Agri-environmental programs in support of extensive vs. intensive agriculture

Program target	E.U.	U.S.
Reduce intensification	Chemical reduction payments; payments for organic farming	None
Reduce extensification	None [a]	CRP CSP WRP Sodbuster, Swampbuster
Support intensification (and limit pollution therefrom)	Additional LFA payments for intensive production in Italy and Spain	EQIP payments for confined animal feeding operations
Support extensification	LFA payments; non-abandonment payments	None
Support move from intensive to extensive	Payments for low stocking rates	None
Support move from extensive to intensive	None	None

[a] Some biodiversity payments might be included, but it is unclear how exactly they are allocated. (Source: Baylis et al., 2004).

Table 2 – Programs by degree of targeting

	Input only	Area target	Expected environmental outputs with explicit ranking
EU	Organic Reduction of chemical inputs Afforestation	Natura	
U.S.	CRP cost-share component: 7% WRP cost-share: 9% FPP		CRP rent: 93% EQIP WRP rent: 81% WHIP

Percentages based on 2001 expenditure (USDA-ERS, 2003). (Source: Baylis et al., 2004).

features of the rural landscape This broadness has been criticized on conceptual grounds by Rickard (2004) who strongly suggests that the EU's goals of reform, competitiveness and sustainability are incompatible; In contrast to EU broadness, U.S. programs tend to be focused on more easily measured targets. That said, several authors suggest that more could be done: Feather et al. (1999) claim that more effective CRP targeting would almost double environmental benefits.[5]

2.4. Seller selection and opportunity cost

Another difference between the two regions is the degree to which the government differentiates payments by opportunity cost. Some U.S. programs are designed to reflect producer's opportunity cost. Competitive auction-based programs such as the CRP require farmers to submit bids based on environmental benefits specific to their land. Contracts are then allocated based on highest benefits for least cost. Opportunity costs are automatically taken into account because farmers are unlikely to submit a bid which is lower than the amount of profit forgone by loss of productive use of their land (see also Ferraro, 2008-this volume, for the relationship between bids and opportunity costs). The CRP often has more bids than it can afford to fund.. For example, during the September 2004 sign-up for the CRP, over 26,000 offers for 1.7 million acres were received, while approximately 19,700 offers were accepted for about 1.2 million acres. Thus, farmers are forced by competition to moderate their bids, and cannot earn excessive rents. In California, on the other hand, very little land is enrolled in the CRP, because the opportunity cost of retiring land is very high. Under EQIP, contracts may be for as short a period as 1 year, reflecting the program's working-land basis. An 'offer index' is calculated for each proposal. This is the ratio of the environmental benefits to the amount of funding requested by the farmer. Conservation plans with the most favorable offer index are approved up to the point where the funds are exhausted.[6] There is now no bidding.

In the EU, by contrast, farmers are offered an amount per hectare based on the individual member state's calculation of the income forgone and additional costs resulting from the commitments (EC, 2005). Enrollment is voluntary and there is often no competition (Crabtree et al., 1999). This approach leaves no room for inclusion of individual opportunity costs, although an additional discretionary payment of up to 20% can be made to induce farmers with higher opportunity costs, and presumably higher environmental benefits, to subscribe. An implication of the difference between the highly targeted U.S. programs and the more 'broad-brush' EU programs is that the former extract more environmental benefits per dollar spent than equivalent EU programs. This is because competitive bidding, compared to fixed-rate payment, can significantly increase the cost-effectiveness of conservation contracting (Latacz-Lohmann and VanderGansevoort, 1997). Thus, even with the side-objective of transferring income to producers inherent in both regions' programs, we suggest that U.S. programs extract more environmental benefits per dollar spent than equivalent EU programs. Along with the more direct targeting mentioned above, the United States is able to capture some of the producer surplus associated with selling environmental services. (See Ferraro (2008-this volume) this volume for an examination of contract design under conditions of asymmetric information.)

2.5. Baseline and additionality

As noted above, in both the EU and the United States, if one assumes that environmental regulation could have been introduced without the commensurate AEPs, the baseline agricultural production would include the GFP and Sodbuster and Swampbuster (at least when non-AEP agricultural subsidies are sufficiently lucrative). Where the two regions differ substantially is their attempts at ensuring additionality. As noted above, the large U.S. programs allocate funds by competitive bidding, which should help ensure that producers are not paid (much) for taking actions they would have taken anyway. Further, due to their greater targeting, the U.S. programs direct more of the funds where they will have the largest environmental impact. Compare this to the EU paying a producer to convert to organic production in a region that is not environmentally-sensitive, or where the producer might well have converted anyway.

Further, in the EU, there is no cost-sharing with farmers. In contrast many U.S. programs are cost-shared with producers, which reduces the opportunities for overpayment. Clearly, the knowledge that a successful bid will require at least some funds from one's own pocket will help to ensure realistic bids. Further, the lower the amount of cost-sharing required, the higher the chance that land will be accepted into a conservation program. Yet, it is worth noting that the CSP pays farmers annually based on conservation practices they had on their land before enrolment, violating the principle of additionality.

In contrast to the site-specificity of the U.S. programs, EU programs appear broad-brush, and opportunities may exist for producers to claim money despite a lack of additionality. Falconer and Whitby (1999) note that some EU member states, for example the United Kingdom, require additionality before payments are made, acknowledging the possibility that additionality might not be created. Continuing with the example of the United Kingdom, Falconer and Whitby (1999) contrast that country's administrative expenditure with that of Greece. As Table 3 shows, the countries differ by a factor of more than five. The explanation is that the United Kingdom has invested heavily in AEP infrastructure in recent years, with a resulting high degree of monitoring and compliance control. The IRENA project, discussed below, is a response to differentials in monitoring ability between member states.

There is an interesting difference between the ways in which baselines are set in the two regions. The USDA is a federal agency, disbursing federal funds. Individual states within the United States have little or no input on the setting of standards. By contrast, EU AEPs, although a compulsory accompanying measure to the Rural Development Regulation, are co-financed and structured independently by each member state. Each member state is also responsible for setting its

[5] See also Hansen et al. (1999) and Babcock et al. (1996).

[6] The USDA (ERS, 2003) has voiced concern about the discontinuation of competitive bidding for EQIP in the 2002 Farm Bill, suggesting that federal cost-share payments may now be greater without the auction process.

Table 3 – Transaction cost comparison

Country	Average ratio of administrative costs to compensation costs for case study schemes in each country (%)
Austria	8.8
Belgium	63.4
France	87.1
Germany	12.3
Greece	8.6
Italy	6.6
Sweden	11.3
UK	47.9
US	
CRP	4.6
EQIP	30
WHIP	33
WRP	45

Sources: EU data from Falconer and Whitby (1999); US data from USDA-ERS, (2003).

own baseline standards, or GFP, and it is clearly in the best interest of each state not to raise its own baseline higher than those of competing trading partners without good reason. Osterburg (2004) has compared GFP definitions within the EU, finding considerable variations, with control by member states carried out using few and indirect indicators. Different member states regard AEPs with varying degrees of interest, and so control frequency varies greatly. Generally there are few incentives to agree on standards and, equally important, no incentives to raise them. Member states are responsible for their own administration costs, and naturally wish to avoid high control costs. National attitudes and the availability of state funds are therefore factors in determining the penetration of AEPs into national policy.

In contrast to the relative weakness and consensus requirement of EU decision-making, the USDA is to some extent able to mandate standards throughout the country, although this power is attenuated by industry lobby-groups. This may change: there have been suggestions that Washington may devolve some decision-making to the States (USDA, 2004). About one third of the USDA annual outlay, $22 billion, could be eligible for transfer, including commodity and conservation programs. If a 'block grant' funding procedure were to be adopted, there would be obvious similarities with EU funding methods.

2.6. *Leakage and spill-over*

In the United States, land retirement programs under the CRP have a positive effect on commodity prices. There are conflicting opinions on whether these price increases have induced farmers to cultivate land that would not have been cultivated if the CRP had not been in place. Wu (2000) estimates that for each 100 acres of cropland retired under the CRP in the central US, 20 acres of non-cropland were converted to cropland. In a lively response, Roberts and Bucholtz (2006), using the same data, found a spurious correlation rather than evidence of slippage. In any case, high site-specificity of U.S. targeting ensures that fragile land is enrolled in preference to more robust land. Therefore, should a farmer intensify production on land not under contract to make up for 'lost' output, the environmental damage will be limited to more resilient soils. Under a provision of the 2002 Farm Bill, the possibility that the CSP may cause further land to be cropped has been countered by a requirement that cropland is eligible for enrolment only if farmed in four of the six years prior to 2002.

By contrast, the EU policy of discouraging land abandonment and actively encouraging extensification may lead to unintended negative externalities, such as increased road traffic from tourists drawn to attractive rural areas. The requirement that land be kept in good *agricultural* condition in order to receive some subsidies, notably the Single Farm Payment (EC, 2003) precludes the afforestation of that land. Such action may not be environmentally and ecologically optimal (Schmid et al., 2007).

However, EU AEPs may have a number of ancillary benefits. Haring et al. (2004) suggest that organic farming will continue to expand, and that this expansion will result in environmental improvements. However organic premiums may be reduced as more supply is available. In the United States, there is an attempt to capture (and count) these ancillary benefits, whereas this is less true in the EU. The USDA (ERS, 2003) calculates that public benefits from pheasant hunting and wildlife viewing have risen by almost $370 million a year, despite a static acreage and cost.

2.7. *Transaction costs*

Comparing the transaction costs of the two regions is presently not possible, because AEPs are relatively new, and the large set-up costs have not yet been amortized. Further, each member state applies its own standards with its own bureaucratic tools. Table 3, below, compares U.S. marginal transaction costs for conservation programs in FY 2001, and presents those of selected EU member states.[7] The CRP appears to be relatively inexpensive to operate, while EQIP, WRP and WHIP cost proportionately more. This may be because an essential part of the bidding process for CRP is the calculation of the Environmental Benefits Index (EBI), which uses a set methodology and existing environmental data. The research and development costs for the EBI must have been high, but those fixed costs do not appear in CRP's transaction costs. On the other hand, land retirement programs such as the CRP have the benefit of being relatively easy to implement, and they are also easy to monitor and enforce, resulting in low marginal administrative costs.

The transaction costs listed above include technical assistance, extension and administration to the U.S. government, and not the transaction costs of farmers who bear the cost of bid preparation. The figure for CRP rental includes US $122.6 million cost-sharing for seeding with native species and reverting land back to its pre-farmed state.

[7] Since the fixed costs are often not explicitly associated with the program in question, and, such as in the case of the National Resource Inventory used for the EBI, may be associated with multiple programs they are not readily available. Because of this, we cannot explicitly compare the transactions costs of the EU programs to the US programs; we can only loosely compare marginal costs.

Corresponding transaction costs for EU agri-environmental programs are not easily available because of variations among member states. It is reasonable to expect that transaction costs are lower because of the less precise targeting carried out by member states, although Falconer (2000) suggests that there are certain hidden transaction costs in conversion to AEPs, such as permits for organic certification. Falconer and Whitby (1999) provide an analysis of transaction and administration costs across eight EU member states, and it is their results which appear in Table 3. There is clearly very considerable variation, ranging between 9 and 75 ECU/ha spent on annual administrative costs, explained by Falconer and Whitby as being due to experience with AEPs and the type of AEPs being implemented. Austria for example has had many years of experience with AEPs, and concentrates mainly on the preservation of upland pastures. The figures for Belgium and France are especially high, and can be explained by the small size of the Belgian agri-environmental sector over which to the fixed costs of policy evaluation and development are spread, while France, as Falconer and Whitby (1999) note, has a 'deeply entrenched' administrative infrastructure.

2.8. *Monitoring and compliance*

In both the United States and EU, contracts are signed with farmers over land-use activities (or restrictions thereon), not on the volume of services provided.[8] The infrastructure for monitoring land-use is therefore more developed than that for services provided, although the latter is arguably more important. Here we discuss land-use monitoring and then move on to service provision monitoring. For the EU, the Integrated Administration and Control System (IACS) prescribes a minimum 'control', or inspection, rate of 5% of all beneficiaries of agri-environmental programs and LFAs (the control rate for beneficiaries of Pillar 1 schemes is 1%). Failure to follow cross-compliance regulations through negligence can cost farmers up to 5% of direct payments, and up to 15% for repeated failure. Intentional failure costs more: from 15–100% of direct payment. If the intentional failure to satisfy cross-compliance is related to a particular aid scheme, then the farmer would usually be excluded from that scheme in the following calendar year (Varelo-Ortega and Calatrava, 2004). Many agri-environmental measures have been in place for less than 5 years, and the development of monitoring infrastructure has been, as the EU itself admits, somewhat slow. In addition, as we have noted above, the uptake of AEPs and the zeal for their adoption by farmers has strongly varied by member states. Potter (2002) reports that non-compliance is not widespread in EU AEPs but clearly this does not mean that no breaches have ever occurred (Fraser and Fraser, 2005). In the United States, 5% of EQIP participants who have certified themselves as being a Limited Resources Farmer, Rancher, or Beginning Farmer or Rancher are randomly selected for an annual administrative verification and review. The penalties for non-compliance include refunding of all payments and barring from certain producer loan programs. Although equipped with such penalties, the USDA seems reluctant to enforce them because this might "reduce applications and participation in areas of real conservation need and could reduce net social benefits more than what occurs from the withdrawals" (USDA-ERS, 2001 p27). The USDA claims that 98% of farmers receiving conservation compliance payments are in fact in compliance, but this high figure has been disputed by the General Accounting Office (USDA-ERS, 2004).

Work has begun only relatively recently on data collection on the services produced by conservation practices. In the United States, the Conservation Effects Assessment Program (CEAP), a product of the 2002 Farm Bill, aims to quantify the environmental benefits of conservation practices used by private landowners participating in USDA conservation schemes (USDA, 2004). In the EU, the EC admits that more work is needed on constructing a strategic and long-term monitoring and evaluation system (EC, 2005), but the, IRENA system (Integration of Environmental Concerns into Agriculture Policy), in collaboration with the European Environment Agency, is a major advance (EC, 2006). The IRENA system provides a set of agri-environmental indicators and related databases for the assessment of the integration of environmental concerns into EU agricultural policy.

2.9. *Permanence*

In the United States, USDA conservation expenditure is historically negatively correlated with commodity prices (Heimlich, 1998). This implies that farmers are less likely to enroll land into conservation schemes when crop prices are high. However, the relatively long tenure of conservation contracts (10 to 15 years for CRP, and 30 for WRP and CREP) may reduce this effect. Further, land retirement programs have had positive effects on farm income (Bernstein et al., 2004). The economic benefits to the farmer of land enrolment would therefore appear to be attractive, and worth working to retain.

Although some land that had been eligible for the CRP in 1985 lost its eligibility in 1992, conservation programs frequently create a vocal claimant group, making them hard to cut, and increasing the probability of continued funding. Since the amount of federal money flowing into conservation programs has shown a continuous increase, for example an 80% increase under the 2002 Farm Bill (Bernstein et al., 2004), permanence seems likely. A related concern is that farmers might be tempted by high commodity prices to break their CRP contracts. The penalties for this seem small and unlikely to be invoked. Although the farmer is liable for liquated damages, these are waived if collecting them is 'not in the best interests' of the program (Federal Register, 2003).

As we have discussed above, EU policy is framed more by socio-economic goals such as maintaining farm income in less favoured areas than the reduction of strictly measurable negative externalities. Baylis et al. (2005) find that European AEPs are at least partly a response to 'green' taxpayer demand, and studies within the EU show that an appreciation of the benefits of a less intensive agriculture is growing (Yrjola and Kola, 2004). Increasing public demand for agriculture's positive externalities and growing sensitivity to its negative externalities makes an influential combination, and EU AEPs are therefore likely to remain an institutional fixture, and possibly

[8] Although, as we have discussed above, the basis for compensation for service provision is quite different, with the costs and benefits of U.S. programs being more strictly measured than their EU counterparts.

increase. European farmers are also unlikely to refuse the extra income earned through compliance, especially as most price supports have now been replaced by the Single Farm Payment.

Both the EU and U.S. seem reluctant to be harsh on farmers who breach AEP contracts, and neither region prescribes permanence. There are a number of possible explanations for this, one of which might be recognition by policy-makers that agriculture's full productive capacity should remain retrievable.

3. Analysis

Having identified a number of fundamental differences between the EU and U.S. policies, we now turn to the question of why these differences have arisen. This question is not merely of academic interest: it has important implications for the design for future payments for environmental services. Further, since the PES programs discussed in this paper have the joint objective of transferring income to producers, understanding the dynamics of the tradeoffs between these objectives can inform us about the effect on environmental services of constraints placed on the ability of countries to transfer income to producers by other means. Last, understanding the policy bargain underlying these programs may indicate whether environmental benefits can be achieved in a more efficient manner. We will attempt to explain these differences in terms of demand for environmental services, the supply of those services and political considerations.

From our description above of the different ways that European and American citizens view agriculture, it is reasonable to suppose that the utilities they assign to various environmental services will be different. We have suggested above that Europeans prefer to see a lightly-farmed nature, while in the United States nature is at its most attractive when human intervention is minimized. Hellerstein (2002) discusses the role of public preferences for rural amenities in the United States and concludes that, in the context of farmland preservation programs, the continuation of active agriculture is not a predominant concern. By contrast, as discussed above, Hackl and Pruckner (1997) find that tourists in rural Austria have a WTP that exceeds local agri-environmental subsidies.

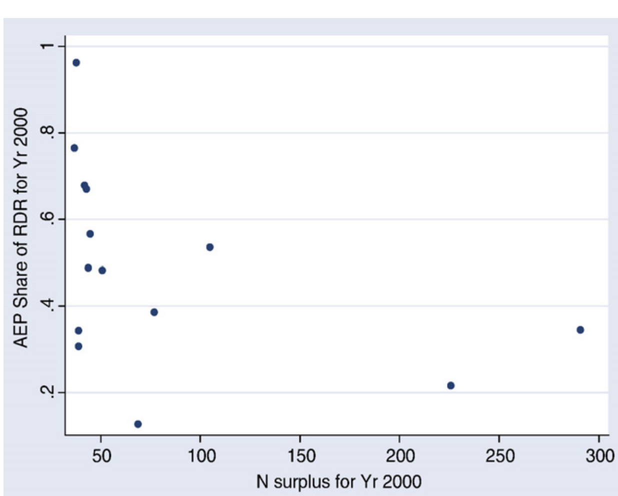

Fig. 1 – Nitrogen surplus against AEP share of RDR for year 2000.

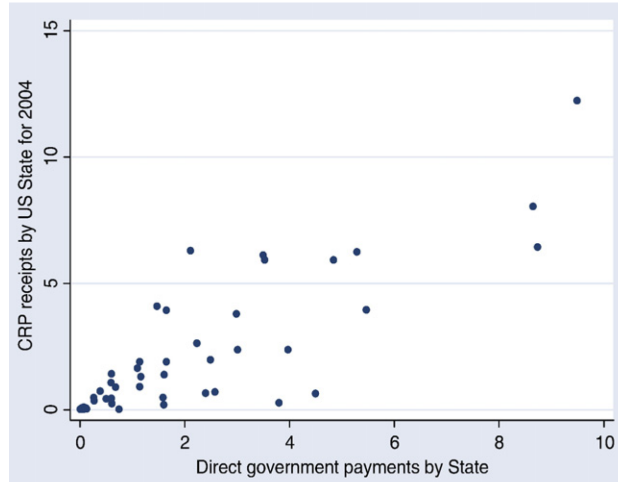

Fig. 2 – Direct government payments for agriculture against CRP receipts for 2004. Source: USDA-ERS, 2006.

Due to production technology and environmental characteristics, agricultural production can produce different externalities in different regions. We would expect that the distribution of agri-environmental programs within the EU would follow a pattern: those member states with the largest externalities should spend the most on agri-environmental measures. However, we see countries with extensive agricultural systems, such as Sweden, pursuing agri-environmental programs that increase this extensification, while countries such as the Netherlands, with highly-intensive agriculture, and the EU's highest nitrogen surplus (EEA, 1998), do very little to promote extensification, as Fig. 1 shows. The implication is that money is being wasted because of weak targeting, which might be remedied by greater site-specificity, as exemplified by some U.S. working-lands programs, such as EQIP.

Structural factors may also affect policy outcomes. For example, the EU member states jointly finance agri-environmental programs while in the United States the national programs, with minor exceptions, are federally funded. In terms of the issue space in the EU, there is an implicit joint restriction on the set of negotiable policies because individual states retain the right to tailor programs to meet their own needs. In the United States, the national government can direct expenditures based solely on national constraints.

The United States has argued that the EU is using AEPs to prop up a cosseted agricultural sector. However, contrary to this argument, countries with the largest AEPs (Austria, Sweden and Germany) do not receive the largest amount of overall agricultural subsidies. Furthermore, our econometric analysis, available in Baylis et al. (2005) provides evidence that the EU's AEPs have been provided, at least in part, in response to demand from 'green' taxpayers. It is reasonable to suggest that the demand for AEPs is strongest from an educated, environmentally aware and predominantly urban populace, with the agricultural sector somewhat grudgingly obliging.

In contrast, as shown in Figs. 2 and 3, CRP expenditure in the United States is highly correlated with the distribution of other agricultural subsidies, while EQIP, or working-land expenditure is not so highly correlated (largely due to the requirement that 60% of EQIP funding (50% prior to 2002) go to

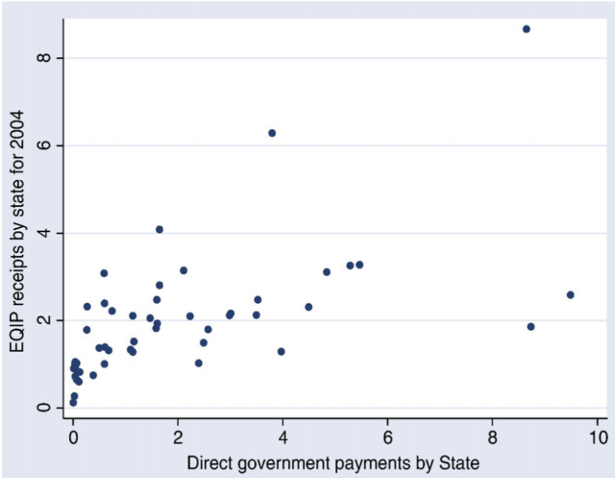

Fig. 3 – Direct government payments for agriculture against EQIP for 2004. Source: USDA-ERS, 2006.

livestock producers, who do not receive price supports or many other agricultural subsidies). Thus, those states that benefit the greatest from other agricultural supports also gain the most conservation dollars.

In comparison, the distribution of farmers and agricultural production differs significantly from the distribution of support payments. Some states with a large number of horticultural producers, such as California and Florida, have a large agricultural GDP but they receive little support from the government in the form of conservation or other agricultural payments. Thus, there is evidence that those producers (and commodities) that are best able to lobby for agricultural subsidies, appear also to be best at obtaining conservation dollars. Further, many states generally associated with a high degree of environmental awareness (e.g. California, Oregon, Pennsylvania, Massachusetts, Connecticut, Vermont, New Hampshire, Maine) seem not to be receiving the bulk of conservation funding. Continuing, note which Senators were members of the Agriculture Committee during the writing of the last farm bill. Membership indicates that at least these state representatives felt that agriculture was important enough to their states that they should lobby to be on the Committee. Again, the Great Plains states are well represented.

Clearly, U.S. agri-environmental programs are designed for the benefit of either the farmer or the environment, but not for either the consumer or the non-farming taxpayer. In contrast, as we have explained above, our econometric analysis shows that many EU programs are responding to 'green' demand. For example, EU agri-environmental measures promote the provision of public goods such as access to privately-owned land for purposes such as hill-walking (e.g. the UK's 'Right to Roam'). Due to the strong belief in private property, particularly by U.S. farmers and ranchers, it is difficult to imagine such benefits being offered in the United States.

4. Conclusion

In this paper, we compared and contrasted U.S. and EU policies directed at paying for environmental services produced by agriculture. We argued that agri-environmental policy in the EU primarily addresses the positive environmental externalities generated by agricultural production, while the U.S. policy mainly addresses negative externalities. We noted that to the extent that it does address negative externalities, agri-environmental policy in the EU focuses on externalities that are by-products of the intensification of farming – i.e. the use of too many non-land inputs per unit of land – whereas U.S. policy targets the by-products of extensification – i.e. the use of excessive amounts of environmentally sensitive land. In the United States, agri-environmental funding is largely based on the anticipated environmental outputs associated with certain activities, while to obtain payments for environmental services in the EU, it is typically sufficient to commit to using agricultural inputs and/or technologies that have been designated as environmentally friendly. The U.S. programs also take individual opportunity cost into account (primarily by requiring bidding for its largest program) whereas the EU programs, which are based on a national or regional set fee, do not. Thus, seller selection in the United States is more geared to maximizing environmental benefits per dollar than in the EU. That said, this higher degree of targeting may come with a higher cost. With the exception of the CRP, which calculates benefits based on a fixed formula incorporating pre-existing micro-level data, attempts to fine-tune program targeting may increase transaction costs, particularly if disaggregated, geographically-specific environmental data is not readily available.

Last, we explored the reasons for these differences. As we showed, there is evidence that citizens within EU countries highly value cultivated landscapes, and there is a revealed willingness to pay for them. This willingness to pay may also exist in some regions in the United States (for example in Vermont or in the Napa Valley of California), but the current slate of conservation programs is not well geared to take advantage of this demand.

In both countries, there is a close relationship between agri-environmental programs and other farm subsidies. It is possible that agri-environmental programs to date have enhanced the relationship between agricultural production and the provision of environmental services. However, the fact remains that such programs represent only a single-digit percentage of expenditure on agriculture, and barely that (OECD, 2003). That said, the funds being allocated to agri-environmental programs are increasing in both regions, while other forms of agricultural subsidies are being constrained by budgets and trade agreements. The challenge will be to design programs that provide the environmental services that the public demands at a reasonable cost to taxpayers, while meeting the political constraints imposed by farmer stakeholder groups.

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