The potential of collective action for the control of nonpoint source pollution in European rural areas.

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Abstract

In the European Union (EU) context, regulatory instruments and individual incentive schemes have been the main policy instruments implemented to control non-point source pollution from agriculture. There also exist some examples of voluntary cooperation among farmers to enhance water quality. However, there has been no systematic assessment of the costs and benefits of such agri-environmental cooperation. The article aims at assessing the potential of co-operative agreements for limiting diffuse nitrate pollution from agriculture. Transaction costs economics are used as a framework to assess the potential advantages of collective action as an alternative or a complement to other policy tools. To identify the conditions under which cooperation may be successful, we then use the Socio-Ecological Systems framework (Ostrom, 2009). A review of empirical studies relative to cases of collective action for agricultural water pollution prevention in the EU context serves as a basis for the identification of the factors likely to affect the success of cooperation for water quality management in agriculture. The analysis relies more particularly on two cases: the Environmental Co-operatives in the Netherlands and the *Ferti-Mieux* operations in France.

Keywords: Nonpoint source pollution; Agriculture; Cooperative approaches; Policy instruments; Transaction costs.

1. Introduction

The EU Water Framework Directive, adopted in 2000, sets the objective of achieving good water status for all bodies of surface waters and groundwater by 2015 (European Union, 2000). This Directive promotes a more integrated approach by defining the river basin as the basis for water management. Also emphasized is that participation among all concerned parties should be encouraged in river basin management (European Union, 2000).

Nonpoint source pollution from agriculture is a major threat to the quality of surface and ground waters in the European Union (EU) context. More particularly, nitrate and phosphorus pollution from agriculture accounts for the largest part of diffuse run off.

Two main types of policies have been implemented to address nonpoint source nitrate pollution in the EU: a regulatory policy (the EU Nitrate Directive) and voluntary individual incentive schemes (the agri-environmental schemes). However, in most Member States, agriculture is still responsible for over fifty percent of the total nitrogen discharge to surface waters (European Commission, 2010).

In the context of implementation of the Water Framework Directive, there is a growing concern that a cooperative approach of water quality management involving farmers and other relevant stakeholders is needed to reach the goals set by the Directive. While it is recognized that higher environmental benefits could be obtained by some cooperation among farmers at a relevant scale, initiatives in this direction remain scarce.

Previous studies assessed the relevance of cooperation for the provision of ecosystem services by farmers (Goldman et al., 2007; Stallman, 2011), including water quality management (Sarker et al., 2008; Stallman, 2011). However, there has been no systematic assessment of the costs and benefits of agri-environmental cooperation as an alternative or a complement to regulation or individual incentive schemes for the prevention of non-point source pollution (Brouwer et al., 2002).

This article deals with the question of the potential of collective action for addressing non-point source nitrate pollution from agriculture in the EU context.

A conceptual framework, combining the SES (Socio-Ecological Systems) framework and transaction costs economics, is provided for identifying the conditions under which cooperative agreements are likely to be an environmentally effective and cost-efficient policy tool for non-point source pollution control.

Transaction costs economics are used to assess the potential advantages of collective action as an alternative or a complement to other agri-environmental policy tools. For these advantages to be effective, the gains of collective action have to be superior to the associated costs. To identify the conditions under which cooperation may be successful, we use the SES framework (Ostrom, 2009; Poteete et al., 2010).

A review of empirical studies relative to cases of collective action for agricultural water pollution prevention in the EU context serves then as a basis for a first identification of the factors likely to affect the success/failure of such cooperation. The analysis relies more particularly on two cases: the Environmental Cooperatives in the Netherlands and the *Ferti-Mieux* operations in France. While the Environmental Cooperatives in the Netherlands constitute an example of collective action initiated by the farmers themselves, the *Ferti-Mieux* operations in France were initiated by public agencies. The comparison between two cases in different Member States allows further for exploring the role played by factors related to the social, economic and political contexts at the national level.

The paper is organized as follows. In a first section, the policies targeting water pollution by nitrates from agricultural sources in the European context and the range of cooperation initiatives are described. The second part of the paper introduces the conceptual framework used to assess the potential of cooperation for water pollution reduction. After an assessment of the potential advantages of collective action as an alternative or complementary tool to achieve water quality objectives, the factors identified as crucial for the success of cooperation are presented in the third section.

2. The European Union context

The policy tools available to address nonpoint source pollution include regulatory instruments, economic instruments (taxes/subsidies) and voluntary compliance approaches (Shortle and Abler, 2001). Because of the diffuse nature of nitrate pollution, it is not feasible to define policy instruments targeting the emissions by farmers. The policies implemented rather aim at modifying the agricultural practices known as influencing the extent of nitrate leaching and runoff (McCann and Easter, 1999; Shortle and Abler, 2001). Reductions in organic and mineral nitrogen fertilization, the introduction of nitrate catch crops in the cropping plan or the establishment of riparian buffers along water courses are examples of the different measures which may be implemented.

In the European context, two main types of policies have been implemented to address nonpoint source nitrate pollution: a regulatory policy (the EU Nitrate Directive) and voluntary incentive schemes (the agri-environmental schemes).

2.1 The policies targeting non-point source pollution

The EU Nitrate Directive has been adopted by the European Communities Council in 1991. The application of this policy includes the designation of vulnerable zones, where nitrate concentrations in surface and ground waters are above 50 mg/l, or above 40 mg/l with an upward trend. Every farmer in a vulnerable zone has to comply with the measures included in specific action programs (e.g. reduced fertilization application levels; establishment of buffer strips near water courses). Additionally, a national code of good agricultural practices is to be voluntarily applied outside the vulnerable zones. Since 2005, the payment of the Common Agricultural Policy (CAP) subsidies has become subject to the farmers' compliance with all environmental regulations, including the Nitrate Directive.

Since 1992, agri-environmental schemes constitute a compulsory component of rural development plans in EU Member States. Under these schemes, farmers voluntary commit for at least five years to adopting practices with positive effects on the environment. In exchange, they receive a financial compensation for the associated costs and income losses. Agri-environmental schemes are co-financed by the EU and Member-States. They may be defined at the national, regional or local level and adapted to different specific agricultural and environmental conditions. They cover a large range of environmental issues, including biodiversity and landscape conservation, wildlife management and water quality management. The extensification of agricultural systems or conversion to organic farming can also be funded through agri-environmental schemes.

The two policies differ in who bears the costs of the prevention of water pollution. The Nitrate Directive regulation follows the “polluter pays” principle while with the agri-environmental schemes, the "beneficiaries pay” principle prevails. Farmers located in vulnerable zones have to follow the restrictions prescribed by the Nitrate Directive without any compensation payments[1]. Agri-environmental schemes compensate financially the farmers for the provision of environmental services. However, agri-environmental commitments must go beyond the Nitrate Directive standards, i.e. beyond the obligations specified in action programs in vulnerable zones and the provisions of the code of good agricultural practices outside vulnerable zones.

The implementation of the Nitrate Directive is still incomplete, mainly relating to insufficient designation of vulnerable zones and not conformity of action programs designed at the national or regional level. A low awareness level of farmers with regard to the requirements of the Directive and difficulties in establishing an efficient control of compliance are two other constraints on the implementation process (European Commission, 2010). As for agri-environmental schemes, they were found to have positive effects on farmers' practices, with a decrease in fertilizers use at the EU level[2]. However, the evaluation of the impact of the schemes on water quality showed that the extent of the area covered was affecting significantly their global effectiveness, beyond their effects within the individual farm boundaries. The importance of continuity effects was also underlined in the case of the establishment of buffer strips along river courses (Oreade-Breche, 2005).

2.2 Evidence of agri-environmental collective action

Evidence of collective action involving farmers for the reduction of water non-point source pollution in the EU context is scarce and scattered. Examples include co-operative agreements between water suppliers and farmers, commonly found in Germany or voluntary initiatives of farmers to control water pollution, observed in Finland, France and Scotland (Brouwer et al., 2002; Davies et al., 2004). These initiatives correspond mainly to ad hoc networks of farmers and possibly non-farmers, whose cooperation is framed by a policy intervention. The Environmental Cooperatives (ECs) in the Netherlands are, to our knowledge, the only example of long-term structured collective action for agri-environmental management. In the paper, we focus more particularly on this case as well as on the case of the voluntary *Ferti-Mieux* operations in France. While the ECs in the Netherlands constitute an example of collective action initiated by the farmers themselves, the *Ferti-Mieux* operations in France were initiated by public agencies.

The ECs have developed as a new governance structure since the beginning of the nineties. They are regional groups of farmers, including in some cases other rural stakeholders (e.g. environmental organizations and local authorities) (Wiskerke et al., 2003). By 2004, 124 ECs were active in the Netherlands, covering fifty-five percent of farmland and involving 10000 members, including one-fourth of non-farmers (Franks, 2010). The ECs environmental

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[1] Member States may still offer aid to farmers to adjust their fertilizing practices, especially in making financial investments (Brouwer et al., 2002). For example, in France, a program financing partly the improvement of storage capacities for manure of cattle breeding farms has been implemented since 1993, as a way to ease compliance with the Nitrate Directive requirements.

[2] The impact of agri-environmental schemes varies from one Member-State to another. In France, the implementation of agri-environmental measures between 2000 and 2006 was evaluated as having very limited effects on farmers' fertilization practices and water quality (AND International, 2008).

activities are not restricted to water quality management and one EC can be active in several environmental “domains”, including also biodiversity conservation or wildlife management (Franks and Mc Gloin, 2007).

The development of ECs led to innovations in the implementation of agri-environmental policy in the Netherlands. In some cases, ECs have been allowed to develop themselves the measures and instruments to achieve the regulatory objectives defined by state agencies (Renting and van der Ploeg, 2001). With regard to the implementation of agri-environmental schemes, priority is now given to collective rather than individual applications (Franks and Mc Gloin, 2007).

There is no formal appraisal of the environmental impact of ECs, however, they are perceived as being ecologically effective, for their positive influence on the participation of farmers to agri-environmental schemes and the evolution in agricultural practices (Franks and Mc Gloin, 2007). Data collected between 1995 and 2000 about the trends in nitrogen surpluses of member farms of the two first ECs (*VEL* and *VANLA*[3]) and a regional reference group of farms show that member farms realized lower nitrogen surpluses and reduced these losses at a higher rate than the regional average over the five years (Renting and van der Ploeg, 2001).

The *Ferti-Mieux* operations have been launched in 1991 by the French Ministry of Agriculture. Managed by the National Association for Agricultural Development (ANDA)[4], the goal was to promote and to label local collective actions of farmers for a better management of nitrogen use at a water-catchment level (Papy and Torre, 2002). In this frame, farmers voluntary commit to changing their practices to limit water pollution, with no financial compensation, along collectively defined prescriptions.

Between 1991 and 2001, sixty-five operations were labelled, involving about 35000 farmers and representing 4.6 percent of the agricultural area. Following the dissolution of ANDA in 2002, the *Ferti-Mieux* operations were officially stopped at the national level. However, the Agricultural Chambers[5] decided to undertake similar operations, under the name of *Agri-Mieux*. These new operations have the additional objective of reducing diffuse pesticide pollution (Verron, 2007).

The effects of the *Ferti-Mieux* operations on water pollution were mixed, with no evidence of decrease in nitrate rates in groundwater bodies; however, one explanation could be the inertia of hydrologic processes in groundwater systems. In areas where surface waters were targeted, more than a half of the operations led to a decrease or a stabilization of nitrate rates, demonstrating the environmental relevance of the operations (Papy and Torre, 2002).

Table 1 presents the main characteristics of the Environmental Cooperatives and the *Ferti-Mieux* operations.

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[3] Respectively, *Vereniging Eastermar’s Lânsdouwe* (VEL) and *Vereniging Agrarish Natuur en Landschapsbeheer Achtkarspelen* (VANLA).

[4] This association, which was disbanded in 2002, had a mixed membership of representatives from agricultural interest groups and the State. Its role was to provide advice to the Ministry of Agriculture and to fund agricultural development programs.

[5] Agricultural Chambers in France are public organizations led by representatives of agricultural and other rural stakeholders. Agricultural Chambers are active at the regional and the *département* level. At the national level, the APCA (*Assemblée Permanente des Chambres d'Agriculture*) is in charge of the coordination of the network of Agricultural Chambers.

Table 1: Two cases of agri-environmental collective action in the EU context: the Environmental Co-operatives in the Netherlands and the *Fertimieux* operations in France.

	Environmental Co-operatives (The Netherlands) *Source: Franks and McGloin (2007)*	***Fertimieux* operations (France)** *Source: Verron (2007)*
Initiative	Farmers	Public agencies (Ministry of Agriculture)
Composition	Farmers or Farmers and non-farmers	Farmers and non-farmers
Structure	Formal organization	Ad hoc network
Funding	Member fees Public agencies (direct support, contracts)	Public agencies (Agricultural Chambers, Water Agencies, ANDA, local governments)
Environmental "domain"	Multiple (water quality, biodiversity, wildlife)	Water quality
Activities	▪ Information sharing and advice provision ▪ Coordination of changes in agricultural practices ▪ Fund raising ▪ Interest representation/lobbying ▪ Research activities	▪ Information sharing and advice provision ▪ Coordination of changes in agricultural practices

3. Conceptual framework

This section presents the conceptual framework used to assess the potential of collective action as a tool for nonpoint source pollution control. A comparative perspective in terms of transaction costs is adopted with regard to the alternative policy instruments. Further, the factors affecting the success of collective action are identified on the basis of the Socio-Ecological Systems (SES) framework.

3.1 A transaction costs analysis of policy options

Transaction costs are the resources used to define, establish, maintain, and transfer property rights (Allen, 2000). Transaction costs arise because information is incomplete and asymmetrically held by parties to exchange (North, 1990). Depending on the characteristics of the good or service considered, the level of transaction costs linked to market coordination will be more or less important.

Like many other environmental goods, water quality presents some public good characteristics. Pure public goods are goods which are non-exclusive and non-subtractive (Ostrom, 2005). The restoration or maintenance of water quality by farmers constitute a public good, as (i) everyone can benefit from the improvement in water quality without diminishing others' benefits (non-subtractability) and (ii) it is difficult (impossible) to prevent anyone from enjoying the benefits of water pollution reduction (non-excludability).

In this situation, the transaction costs associated to decentralized market exchange are so high that the public good will be underprovided. For example, in the case of diffuse nitrate pollution, the costs of identifying the sources of the pollution as well as the other affected individuals will highly constraint market coordination (Falconer et al., 2001). Some form of organization is needed to overcome the suboptimal provision of the public goods (Ostrom and Walker, 2000).

However, the alternative institutional arrangements (including the diverse types of government interventions) also present transaction costs and the question is then which arrangement allows for the provision of the public good at the lowest costs.

Coase (1960) suggested adopting a comparative perspective on the relative benefits and costs (including transaction costs) of the different social arrangements. Such an approach has only recently been developed in the field of environmental policy. A growing body of research seeks to include transaction costs in the analysis and evaluation of environmental policies (Coggan, 2010, McCann et al., 2005). Several studies have measured empirically the extent of transaction costs linked to the implementation of environmental policies, showing their high significance (Falconer, 2000, 2001; McCann and Easter, 1999; Mettepenningen, 2009). However, there is still a limited literature on the factors influencing the type and the level of transaction costs associated to different environmental policy instruments (Coggan, 2010).

With regard to the implementation of environmental policies, transaction costs correspond to the search and information costs, bargaining and decision or contracting costs and monitoring, enforcement and compliance costs (McCann et al., 2005). Falconer (2000) underlines that an important aspect to consider in assessing the transaction costs associated to policy implementation is which party bears these costs, i.e. the public agencies or the stakeholders targeted. Transaction costs are also likely to vary along the implementation of a policy due to learning and the presence of fixed costs, which are incurred primarily at the beginning of the program (McCann et al., 2005).

Adopting a comparative perspective between different policy options, collective action as a tool for non-point source pollution control may present some advantages in terms of transaction costs (see section 4.1.). However, collective action itself bears some costs and for it to be a valuable option, these costs are to be less that the benefits accrued to participants. This will depend upon a range of factors we identify on the basis of the SES (Socio-Ecological Systems) framework.

3.2 Identifying the factors affecting collective action: the SES framework

The SES framework was developed as a tool for the analysis of complex Socio-Ecological Systems (SES) (Ostrom, 2007; 2009; Poteete et al., 2010). This ontological framework lists and structures the variables which have been found in previous research to influence the patterns of interactions and outcomes in diverse SES. The characteristics of the natural resource considered (resource system and resource unit), the characteristics of the users and the characteristics of the governance system are the four main sets of variables considered as potentially important to analyse the outcomes achieved in a given SES (Figure 1). Also the broader social, economic and political contexts as well as the related ecosystems are included as interacting with the other subsystems. This framework constitutes an extension of the Institutional Analysis and Development approach (Ostrom, 1998), with a specific attention given to the characteristics of the biophysical systems and their impact on natural resource management (McGinnis, 2011).

Among the variables identified as potentially relevant, a sub-set of ten factors likely to affect the benefits and costs of collective action for the sustainable management of natural resources (Ostrom, 2009; Poteete et al., 2010) is seen to be critical for the success of self-organization by users of common-pool resources (Figure 1).

The characteristics of the resource systems identified as affecting the likelihood of self-organization by users include the size and productivity of the system and the predictability of system dynamics. A moderate size of the resource system is seen as conducive to self-organization as a larger size means higher management costs and a smaller size may imply a less valuable flow of products from the system. A moderate level of resource scarcity (productivity of the system) is also likely to induce collective action by users, unlike situations where the resource is either already exhausted or abundant. A low predictability of the system dynamics will increase the management costs of the resource, thereby reducing the likelihood of self-organization. Management costs also depend on the resource unit mobility, stationary units (e.g. water in a lake) being less costly to manage than mobile units (e.g. water in a stream) (Ostrom, 2009; Poteete et al., 2010).

The number of users, their knowledge of the socio-ecological system, the importance of the resource, the presence of leaders and the existence of norms and/or social capital within the group are the characteristics of users which have been frequently found to affect the likelihood of self-organization in empirical studies.

A larger number of users means higher transaction costs, however a small group size may be a constraint on the pooling of resources needed to sustain collective action. The sharing of a common knowledge of the socio-ecological system is seen as decreasing the perceived costs of organizing by users. The importance of the resource to users, in terms of income or non-economic value, will affect the expected benefits from collective action, relative to its costs. The presence of well-respected local leaders and the existence of norms of reciprocity and/or social capital within the group are likely to decrease the transaction costs associated to collective action (Ostrom, 2009; Poteete et al., 2010).

Finally, identified as crucial for the success of self-organization is the autonomy users have to define and enforce the rules governing resource management (Ostrom, 2009; Poteete et al., 2010).

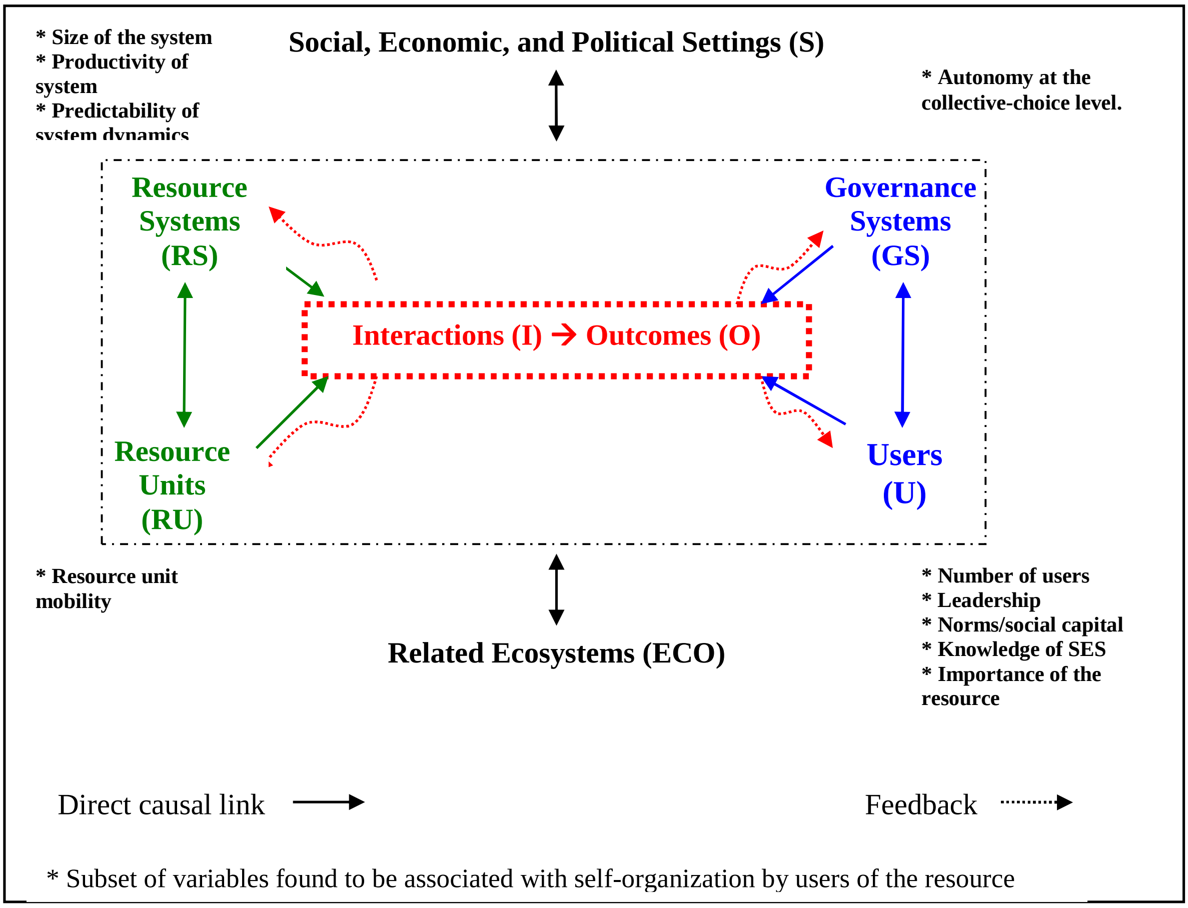

Figure 1: The SES framework (Source: Poteete et al., 2010)

While the SES framework is designed primarily for the study of common pool resources problems, it provides useful insights for the analysis of the provision of environmental services presenting public goods characteristics such as the restoration/maintenance of water quality.

In both situations, participants to collective action face a potential free-riding problem, due to the difficulty in excluding beneficiaries for failing to contribute to the maintenance of the common-pool resource/ the public good (Ostrom, 2005; Ostrom, 2010).

The main difference between water pollution reduction by farmers and a typical common-pool resource dilemma lies in the benefits drawn from the resource by the actors (users). Unlike users of a common-pool resource in most instances, farmers' activities are rarely dependent on water quality. Thus, they have little direct economic incentive to engage in collective action for the control of non-point source pollution. Still, they can draw indirect economic advantages as well as non-monetary benefits from such cooperation (see section 4.2.1).

4. The potential of collective action

As an alternative or complement to other agri-environmental policy instruments, cooperation between farmers or between farmers and other stakeholders may present some advantages for the control of nonpoint source pollution.

4.1 The advantages of collective action for nonpoint source pollution control

Collective action is needed for the objectives of water quality improvement to be reached, as this requires action beyond farm boundaries, at the scale of the drinking water catchment or the watershed. Coordination at the appropriate scale of management can be achieved by regulations, through zoning. For example, the EU Nitrate Directive is implemented in areas designed as "vulnerable zones" for nitrate pollution. Also, agri-environmental incentive schemes are frequently offered in specific areas chosen to match the environmentally relevant scales. However, the incentives provided, on an individual farm basis, are not linked to the implementation of measures at the larger scale (Goldman et al., 2007). Franks and Mc Gloin (2007) note that a key contribution of the Environmental Cooperatives in the Netherlands is to enable Dutch environmental agencies to contract with groups of farmers, allowing for the environmental actions to be taken across land management boundaries.

The agricultural impact in terms of nitrate diffuse pollution presents a high spatial variation depending on hydro-climatic conditions, soil types and agricultural practices. The adaptation of measures to local conditions is thus of importance for an effective reduction of water pollution (Lacroix et al., 2010). However, designing precise measures may be very costly for the public agencies in terms of information collection and processing. Relying on collective action allows for taking advantage of the knowledge hold by farmers about their own farming system and local environment (Wiskerke, 2003).

The level of information asymmetries between the public agencies and farms is an important issue in monitoring diffuse nitrate pollution. Relying on a collective for controlling and enforcing farmers' practices will lower the public costs (Renting and van der Ploeg, 2001; Slangen, 1994). Compared to public agencies, farmers' collectives may be advantaged by an easier access to information and the possibility of using enforcement mechanisms such as trust or reputation. Still, the public agencies will remain ultimately accountable for the group's performance and thus will incur some monitoring and enforcement costs (OECD, 1998).

Farmers may be more willing to comply with measures they contributed to design as opposed to measures externally imposed to them. There is growing empirical evidence on the effects of external interventions (positive monetary rewards or regulations with sanctions) on the intrinsic motivation of individuals (Frey and Jegen, 2001). More particularly, external interventions may crowd out motivation if the individuals affected perceive them to be controlling. As a result, the individuals react by reducing their intrinsic motivation in the activity controlled (Frey and Jegen, 2001). Self-organization for the definition of actions to address diffuse pollution may thus improve the extent to which the policy objectives are reached, by enabling the farmers to endorse the policy goals (OECD, 1998).

Several studies have shown that significant transaction costs were associated to the implementation of agri-environmental schemes in the EU context, both for implementation agencies and for the participating farmers (Falconer, 2000; Falconer et al., 2001; Mettepenningen et al., 2009). Transaction costs borne by the farmers include the search and information costs involved in their decision to participate to an agri-environmental program as well as the contracting costs linked to the administrative tasks required by participation. The public agencies also incur contracting costs with the management of farmers' applications and monitoring and enforcement costs of farmers' compliance with the prescriptions. The transaction costs borne by farmers were shown to be a potential constraint on their participation in the schemes and thus on the achievement of the environmental objectives. In this regard, the smaller farms may be relatively more affected due to the presence of high fixed transaction costs (e.g. required farm audits for participation) (Falconer, 2000).

Transactional economies of scale may be achieved by making collective management agreements (Falconer, 2000). Information and knowledge sharing about the agri-environmental schemes and the implications of participation within a group may reduce the costs of decision-making for individual farmers (OECD, 1998). Collective applications for participation to the agri-environmental schemes may also reduce the contracting costs, both for the farmers and for the public agencies in charge. Franks and McGloin (2007) report effective transaction-costs savings in the implementation of agri-environmental schemes in the Netherlands, with the development of collective contracts between the administration and the Environmental Cooperatives.

Collective action may thus present some advantages to reach water quality objectives in a cost-efficient way. The achievement of cooperation between farmers will however depend on a number of conditions we identify on the basis of the SES framework.

4.2 The factors affecting the potential of collective action

We consider first the factors likely to affect the extent of the benefits farmers can draw from a collective action oriented towards the management of water quality. In a second sub-section, we review the variables which may influence the costs of cooperation.

4.2.1. The factors affecting the private benefits of farmers

As farmers do not bear primarily the costs of water pollution or do not enjoy the benefits of water quality improvements, they have generally little incentive to engage in collective action for reducing the level of pollution (preventing the public bad) or contributing to the maintenance of water quality (providing the public good).

In some cases, farmers may reap some private economic benefits from changing their agricultural practices towards less polluting practices. A better management of fertilization (e.g. by modifying the crop rotation plan) may lead to some savings in fertilizers expenses, without any decrease in yields. However, the extent of these cost savings will vary according to the farming system. In the context of intensive crop farming, reducing mineral fertilization may be quickly costly while cattle breeding farms have the potential to substitute organic to mineral fertilization to a certain extent (Lacroix et al., 2010).

Another economic motivation for farmers to participate to collective action can be the possibility to maintain or increase their profits through the certification of their products (e.g. eco-labels). The logic is that the costs incurred by farmers for changing their practices in favour of the environment are compensated by better marketing opportunities. This will depend on the existence of a demand for “green” products (Ribaudo et al., 2010; Grolleau and McCann, 2012). Conversely, if market incentives are such that the participation to collective action leads to drastic income reductions, then farmers will be less willing to engage into cooperation. For example, crop farmers in the plain areas in France often draw their income from contracts with agro-industrial cooperatives for the production of high quality cereals. Restrictions on fertilization have a direct impact on their capacity to fulfil the conditions of the contracts in terms of product quality (e.g. the high protein content of the produce) or quantity. Hence one can expect less motivation from the farmers to cooperate for water pollution reduction if this prevents them from fulfilling the terms of the contracts on which they depend economically (Amblard and Mann, 2011). Not surprisingly, the *Ferti-Mieux* operations involving agro-industrial cooperatives or other trading partners were identified as attracting a stronger support from farmers (Verron, 2007).

The threat of government regulations or penalties may encourage voluntary collective action by reducing expected net farm profits (Ayer, 1997). This can be illustrated by the example of the emergence of the first environmental cooperatives in the Netherlands. The two cooperatives VEL and VANLA were created in the Frisian Woodlands area as a reaction by farmers to a series of national regulations aiming at limiting the environmental impact of agriculture (including the reduction of nitrogen losses by dairy farms). Farmers considered that these policies were threatening the viability of their local farming system and were likely to be ineffective with regard to the environmental objectives targeted (Wiskerke et al., 2003). The two cooperatives negotiated with the Dutch government for exceptions concerning the application of the state regulations and committed in exchange to undertake some alternative actions to reduce nitrogen losses (Franks, 2010; Wiskerke et al., 2003).

Another motivation for cooperation can be to gain political weight. Farmer-led collectives may be seen by farmers as a political tool to influence policy decisions. The Environmental Cooperatives have now become major actors in agri-environmental policy definition in the Netherlands (Franks and McGloin, 2007). Their emergence and success among farmers may be explained by the initial dominance of ecological expertise in the definition of environmental policies in the Dutch context, which led to a weak representation of farmers' interests (Daniel and Perraud, 2009). The role of political representation as a strong driver for environmental collective action may be less important in other national contexts, e.g. in France, where an institutionalized co-management of agri-environmental policies between the state and farmers' organizations has for a long time permitted farmers to effectively represent their perspective.

Finally, besides economic benefits, non-monetary incentives may play a role in the willingness of farmers to participate to collective action. Farmers with preferences for environmental preservation will be more willing to participate to cooperate for the reduction of water pollution (Lubell et al., 2002).

4.2.2. The factors affecting the costs of collective action

The number of participants to collective action and their heterogeneity are recognized as two crucial and interrelated factors in the literature on self-organization for the management of common-pool resources. However, the relationships between these two aspects and the success of collective action were found to be highly variable across empirical settings (Poteete et al., 2010).

On the one hand, the transaction costs associated with coordination and monitoring and enforcing the rules of collective action may increase with the size of the group of participants. On the other hand, a smaller size may be not enough to pool the resources needed to sustain collective action (Poteete et al., 2010).

In the *Ferti-Mieux* case, the success of the operations was clearly related to the number of participants. The operations initially involving a large number of farmers either failed or split up in smaller sub-groups (Verron, 2007). The environmental co-operatives in the Netherlands show a great variation in the number of their members, ranging from 15 to 1700 members (Franks and McGloin, 2007). The literature reviewed does not provide for insights about the implications of the co-operatives size for their functioning.

With regard to the group composition, heterogeneity in the production systems and the individual abatement costs of farmers may increase the bargaining costs of defining the actions to implement for the prevention of water pollution. Heterogeneity in the preferences for environmental preservation of the group members is also likely to constraint the agreement process (Lubell et al., 2002). The participation of farmers and non-farmers may also increase the decision-making costs. The different goals and perspectives of farmers and other stakeholders (e.g. environmental associations, water suppliers) can even lead to conflicts (Franks and McGloin, 2007).

In relation to the size and heterogeneity of the group of participants, the size of the water catchment or watershed will affect the likelihood of successful collective action. A larger water basin means a larger number of farmers and potentially a higher heterogeneity in their farming systems.

The type of water system targeted is also likely to affect the costs of defining the actions to implement and the costs of assessing their impact on water quality. In the case of ground waters, due to the complexity of hydro-geological processes and the time lag between changes in agricultural practices and the evolution of nitrate concentrations, the relationship between implemented actions and water quality may be difficult to establish. In contrast, the shorter response time of surface waters makes the evaluation and adjustment of actions easier (Nimmo Smith et al., 2007). The availability of relevant water quality data, depending on the existence of monitoring stations and the possibility of discriminating the sources of nitrate pollution, also affects the management costs of farming practice changes (Verron, 2007).

Transaction costs associated to collective action will depend strongly on the operational rules defined with regard to the decision-making process and the enforcement of decisions within the group.

In France, the operational rules of the *Ferti-Mieux* program were externally defined at the national level and common to all operations. At the local level, a steering committee and a technical committee involving farmers' organizations, local public agencies and in some cases water suppliers and agro-industrial cooperatives were in charge of the supervision of the projects. A coordinator (usually working for a local Agricultural Chamber) was responsible for defining, together with the participating farmers, the plan of measures to be implemented for the reduction of water pollution. The official label *Ferti-Mieux* was attributed and renewed if the agreed changes in farmers' practices were effective. The evaluation of farmers' practices was realized by the local technical committee and then validated at the national level. The evaluation was based on direct visits and checks on a representative sample of individual farms (Verron, 2007).

In some cases, farmers created autonomous structures (e.g. associations) through which they could define and enforce their own rules, while remaining in the general *Ferti-Mieux* frame. This greater autonomy of farmers was identified as a positive factor on the durability of the operations (Kockmann et al., 2003).

In the Netherlands, all Environmental Cooperatives have developed their own rules (Franks and McGloin, 2007). Generally, they are managed by a board elected annually. Sub-committees are responsible for managing individual projects and developing new activities. Regular meetings take place, as well as an annual general meeting at which changes to the group's plan and constitution can be made (Franks and McGloin, 2007). Individual members decide for themselves whether to participate in any EC activity. While they can suggest programs for the ECs to be involved in, only those initiatives supported by a large share of members will be supported by the ECs (Franks, 2010).

Most cooperatives have developed monitoring systems, involving members or external professionals (Polman and Slangen, 2002). The board of the cooperatives may exclude individual members who do not respect the agreed rules (Wiskerke et al., 2003).

The presence of a local leader/social entrepreneur able to stimulate and animate collective action also appears to be an important factor (Davies et al., 2004). The most successful *Ferti-Mieux* operations were characterized by the involvement of a coordinator familiar with the local context and considered as knowledgeable and trustworthy by famers (Verron, 2007). The role of respected leaders was also identified as crucial in the success of ECs (Franks, 2011).

The existence of trust and shared norms of reciprocity will lower the costs of reaching agreements and the costs of monitoring and enforcing these agreements (Poteete et al., 2010). Lundqvist (2001) documents the case of a failed attempt to induce collective action in a water catchment in Sweden where the collective memories of trust and reputation within the farming community seemed to rule out any possibility of cooperation. Davies et al. (2004) stress that the match between the optimal management scale and informal social networks is context specific. They found in the Scottish context that, most often, strong social relationship did not fall in contiguous spatial patterns, but may be scattered throughout a local area. The authors suggest that the strong tradition of self-reliance of Scottish farmers may explain this outcome.

Finally, government policies can contribute to lower the costs associated with collective action. The Dutch Ministry of Agriculture supported the development of Environmental Cooperatives through grants to cover start-up costs and by adjusting the national agri-environmental program to include the option of joint applications from EC members (Franks and Mc Gloin, 2007). In the French context, public funding compensated the extra-costs of coordination and follow up of the *Ferti-Mieux* operations (Verron, 2007).

5. Conclusion

The objective of this paper was to identify, on the basis of existing case studies, the conditions under which collective action could be an alternative or a complementary tool to regulation or individual incentive schemes to address the problem of water non-point source pollution in European areas.

The adoption of a comparative perspective on the different policy tools led us to specify the advantages of farmers' cooperation in terms of transaction costs.

Compared to regulation, self-organization by farmers will be associated with lower design and enforcement costs for the public agencies. A greater participation of farmers to the definition of the measures they have to implement is likely to increase their compliance and thus to foster the realization of the water quality objectives.

Collective action for joint applications to agri-environmental schemes may allow for transaction costs savings both for farmers and for public agencies, compared to individual schemes. Cooperation will also improve the environmental outcomes, as water quality improvements require action at a larger scale than individual farms.

The potential of collective action to be an environmentally effective and cost-efficient policy tool will further depend on a number of factors we identified on the basis of the SES framework.

Among the characteristics of the actors involved, the type of farming system and the preferences of farmers were found to potentially influence the private benefits drawn from collective action. Farmers incurring fewer costs in changing their practices and/or having strong preferences for environmental preservation will be more likely to participate to collective action. At the group level, a large number of participants together with a greater diversity in preferences and farming systems may increase the costs of collective action. The presence of a leader or the existence of trust and social capital within the group of participants are likely to decrease these costs.

The size of the water system targeted, in conjunction with the number of potential participants and their degree of heterogeneity, will affect the likelihood of successful collective action, with a collective management of larger basins or catchments involving greater transaction costs. Other relevant characteristics of the resource system, likely to affect management costs, are the type of water system and the existence of water quality monitoring infrastructures.

The characteristics of the governance system are identified as a crucial factor for the success of collective action in the SES framework. No detailed information was available on the different sets of rules governing the collectives taken as examples in this paper and on their implications for the outcomes of cooperation. However, the two cases presented here (Environmental Co-operatives and *Ferti-Mieux* operations) highlight the positive effect of an autonomous design of rules by the participants.

Finally, the nature of market incentives, the political context and the existence of government support were identified as determining strongly the emergence and sustainability of agri-environmental cooperation. The importance of these context conditions can be related to the public good nature of non-point source pollution control by farmers. In the presence of few direct economic incentives, the success of collective action will substantially depend on external economic and political incentives.

The Socio-Ecological Systems framework proved to be a useful tool to guide the identification of the factors likely to affect the benefits and costs of collective action for water quality management in agriculture. The results presented here were drawn from empirical studies either dealing only indirectly with agri-environmental collective action or using different conceptual approaches to address this issue. Further investigation is thus needed, including direct empirical data collection for the test of the assumptions made on the basis of the insights provided by the SES framework.

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