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Environmental Protection Agency
An Ghníomhaireacht um Chaomhnú Comhshaoil

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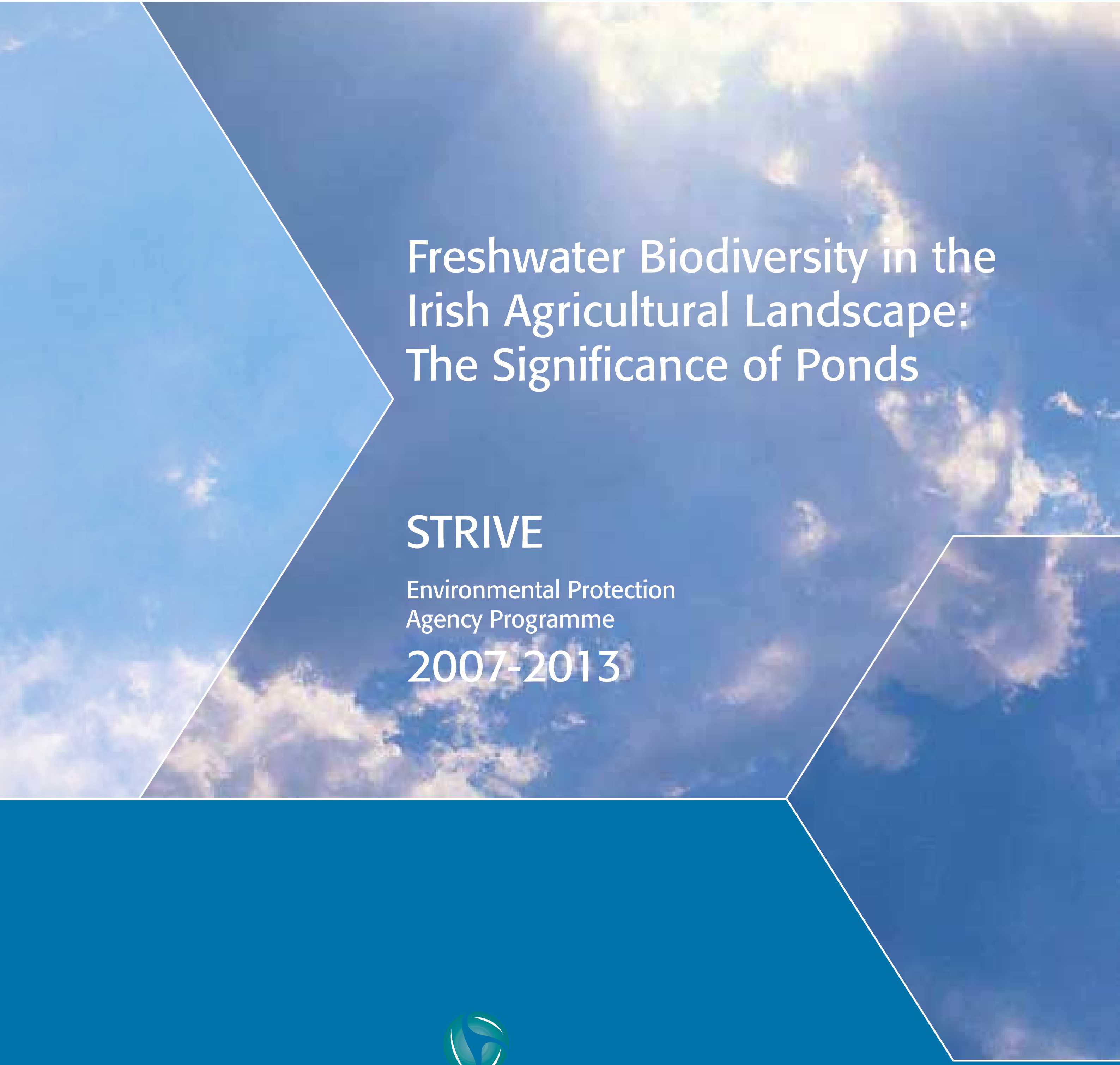

Freshwater Biodiversity in the Irish Agricultural Landscape: The Significance of Ponds

STRIVE

Environmental Protection
Agency Programme

2007-2013

SSTI
Strategy for Science, Technology and Innovation

Comhshaol, Pobal agus Rialtas Áitiúil
Environment, Community and Local Government

Environmental Protection Agency

The Environmental Protection Agency (EPA) is a statutory body responsible for protecting the environment in Ireland. We regulate and police activities that might otherwise cause pollution. We ensure there is solid information on environmental trends so that necessary actions are taken. Our priorities are protecting the Irish environment and ensuring that development is sustainable.

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The EPA is assisted by an Advisory Committee of twelve members who meet several times a year to discuss issues of concern and offer advice to the Board.

EPA STRIVE Programme 2007–2013

Freshwater Biodiversity in the Irish Agricultural Landscape: The Significance of Ponds

Biodiversity of Aquatic Coleoptera in the Irish Farmed Landscape. The Significance of Ponds

(2007-FS-B-14-M5)

STRIVE Report

Prepared for the Environmental Protection Agency

by

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DISCLAIMER

The EPA STRIVE Programme addresses the need for research in Ireland to inform policy makers and other stakeholders on a range of questions in relation to environmental protection. These reports are intended as contributions to the necessary debate on the protection of the environment.

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Executive Summary

Freshwater is one of the most diverse and yet threatened components of global biodiversity. In Ireland, structural changes in the agricultural sector have resulted in a long-term decline in water quality, in the eutrophication of surface waters, and in the drainage and reclamation of non-farmed freshwater habitats, with likely detrimental losses of biodiversity that have yet to be assessed. Despite agriculture being the predominant land use, Ireland does not yet have in place a systematic programme to monitor changes in freshwater biodiversity associated with agricultural activities. The achievement of biodiversity and water quality objectives requires investigations on the distribution of freshwater habitats and their diversity, with a particular focus on those systems whose biodiversity potential is high and whose study has been largely neglected, such as ponds. Many studies have highlighted the capacity of ponds to support more species, as well as more uncommon, rare, and threatened species compared to other freshwater ecosystems, such as lakes and rivers.

The conservation of ponds represents a major opportunity to maintain and even enhance biodiversity, due to their large number and the relatively small size of their catchments areas. Ponds, however, are amongst the least-known habitat in the mosaic of semi-natural or man-made ecosystems in the Irish agricultural landscape. To address this issue, this study investigated the ecological quality and potential conservation value of a range of ponds in two regions in Ireland. Water beetles (Coleoptera) and wetland plants were used as indicators of pond biodiversity, and a number of local environmental and management variables were measured. To identify the main drivers of pond biodiversity, an assessment of the relationship between plants and beetles and between each taxonomic group with a range of environmental variables was made. Since the heterogeneity of the local and regional conditions characterizing ponds may hamper the identification and proper quantification of biotic–abiotic relationships, an analytical framework that allows the identification of the main drivers of pond biodiversity was developed and a number of recommendations were made. To address issues associated with the collection of extensive pond biodiversity data, including temporal and financial constraints, a quantitative assessment of the potential use of vegetation data as a surrogate for pond biodiversity was also carried out.

This study shows that farmland ponds contribute substantially to the maintenance of freshwater biodiversity in Ireland. A large number of water beetle and plant species were recorded from 54 farmland ponds. Of the 76 beetle species collected from the study ponds and representing over 30% of the Irish water beetle fauna, 4 species are included in the Red List of Irish Water Beetles. It was confirmed that grazing intensity and nutrient enrichment have a detrimental effect on the diversity of plant and beetle assemblages, and that the presence of a buffer zone of marginal vegetation as well as that of a fence system improved the overall ecological quality of ponds. Temporary ponds were significantly less diverse than permanent ponds, but they may contribute to maintaining water beetle diversity at the landscape level by improving connectivity between freshwater systems. The diversity of the study ponds was only moderately correlated with pond surface area, indicating that the maintenance or the creation of small ponds characterized by a shallow zone may play a central role in the conservation of biodiversity. Despite not being quantified, a general decrease in the number of farmland ponds was observed as a consequence of the intensification of agricultural practices and housing developments.

The use of a combination of novel and traditional statistical techniques to analyse pond data makes this study the first rigorous quantification of the relationship between plant and beetle community patterns in ponds and allow the identification of the best predictors of pond biodiversity. Plant community data showed a positive capacity to predict patterns in water beetle species composition. The value of plant community data in predicting water beetle species composition was comparable to that of environmental data. Plant and beetle communities responded in a similar way to the same environmental gradients, indicating that the use of

one group only to make a preliminary evaluation of the overall ecological quality of a pond would be sufficient. Water beetles proved to be a suitable taxonomic group to evaluate the ecological quality of ponds since they were found in any study pond and always formed multispecies assemblages.

The conservation freshwater biodiversity in Ireland requires extensive information on the distribution of ponds and on their biodiversity value and ecological quality as well as the development of long-term monitoring programmes aimed at evaluating changes in biodiversity associated with anthropogenic activities, including the intensification of agricultural practices, urbanization, climate change and the introduction of invasive alien species. In this study, a number of research needs were highlighted. In particular, an urgent need for the development of environmental policies specifically aimed at protecting freshwater biodiversity and ponds was identified. Since small freshwater ecosystems, even those that are temporary, are important in supporting uncommon species and in maintaining habitat connectivity, specific legislations should be developed to protect these systems:

- A recording scheme aimed at evaluating the distribution of freshwater biodiversity should be initiated and this should include plant and animal groups. In particular, patterns of freshwater biodiversity should be monitored regularly, to assess the effect of environmental changes on biodiversity and to identify areas/systems where biodiversity is decreasing.
- A classification system of Irish ponds based on their ecological quality and biodiversity value should be developed.
- A ‘pond creation’ incentive should be developed with the aim of maintaining or even increasing freshwater biodiversity via the creation of good-quality habitats for pond life. This would in turn increase the connectivity among freshwater habitats and thus reduce the probability of species extinctions associated with habitat losses and degradation due to anthropogenic activities and climate change.
- Financial benefits should be created to promote the conservation of existing ponds, which are currently under the threat of degradation from the use of nitrates and phosphates, pesticides, herbicides and from direct grazing.
- To improve the ecological quality of existing ponds, incentives should be provided as compensation for a decrease in the use of fertilizers, herbicides and pesticides. Incentives should be offered for the creation of fences aimed at preventing direct grazing by cattle.
- Care should be taken in ensuring that enough space is left between the pond margin and the fence (at least 1 m), in order to allow the marginal vegetation to establish. This would also achieve the objective of reducing the inputs of nutrients directly in the pond water, in addition to preventing the loss of microhabitats suitable for animal life.

1 The Biodiversity Value of Farmland Ponds

1.1 Introduction

1.1.1 Freshwater Biodiversity

Freshwater is one of the most diverse (Dudgeon *et al*., 2006) and yet threatened components of global biodiversity, with losses that greatly exceed those occurring in the most affected terrestrial ecosystems (Ricciardi and Rasmussen, 1999; Sala and Jackson, 2006). Amongst freshwater ecosystems, ponds (here defined as water bodies comprised between 1 m^2 and 2 ha in surface area, either permanent or temporary, natural or man-made [see Biggs *et al*., 1998, 2005]) have been found to play a central role in the maintenance of regional freshwater biodiversity, supporting more species, as well as more uncommon, rare, and threatened species compared to that of other freshwater systems, including rivers and lakes (e.g. Williams *et al*., 2004; Biggs *et al*., 2005). In particular, ponds are among the most diverse type of ecosystem in the agricultural matrix, where they may act as biodiversity hotspots when properly managed (Céréghino *et al*., 2008; Davies *et al*., 2008; Brainwood and Burgin, 2009). The importance of ponds in maintaining biodiversity also extends to temporary ponds, as indicated by previous investigations (e.g. Fairchild *et al*., 2003), showing that temporary ponds may support uncommon and nationally scarce wetland plant species, as well as uncommon and rare invertebrates species (Nicolet *et al*., 2004; Williams *et al*., 1998, 2004). The central role of ponds in maintaining freshwater biodiversity was recognized with the launch of the European Pond Conservation Network (EPCN) in Geneva in 2004. Despite their potentially high conservation value at the local and landscape level, ponds are however continuously lost and degraded by the intensification of agricultural activities (Dudgeon *et al*., 2006).

Ireland has made commitments under the Convention on Biological Diversity (ICBD), agreed in 1992, which aims at the conservation of biodiversity at individual species, community, landscape and ecosystem levels. Part of this task is to quantify the effects of human activities such as agriculture on biodiversity. The increasing productivity of agriculture, involving the use of fertilizers, pesticides, and herbicides, and the progressive mechanization of crop and grassland management have resulted in eutrophication, a long-term decline in water quality, and in the decrease of non-farmed habitats, such as ponds and field-boundaries, with detrimental effects on biodiversity (e.g. Boothby, 2003; Biggs *et al*., 2005; Menetrey *et al*., 2005). At present, Ireland does not have a systematic system in place to monitor trends in freshwater biodiversity associated with agriculture, despite this sector being the predominant land-use type. Because ponds are not explicitly mentioned in the Water Framework Directive, each European member state must develop policies and legislation specifically targeted at protecting these ecosystems if biodiversity is to be conserved (see EPCN, 2007).

1.1.2 Ponds as Potential Biodiversity Hotspots

The conservation of ponds represents a cost-effective opportunity to maintain biodiversity, due to their large number in the landscape and the relatively small size of their catchment area compared to that of lakes and rivers (Biggs *et al*., 2005). Ponds may also be created (e.g. Boothby, 1999 and references therein), with a potential to maintain and even enhance biodiversity at the local and regional level, by increasing habitat connectivity. In Ireland, ponds are amongst the least-known ecosystems in the mosaic of semi-natural or man-made habitats (Feehan, 2003). In particular, information on the ecological quality of ponds nationally or on how their distribution has changed over times is scarce. Ponds in Ireland have been subject to degradation and destruction following the intensification of agricultural production, following a trend that has been reported for other countries (e.g. Williams *et al*., 1998; Wood *et al*., 2003). Existing data on structural changes in farming practices suggest that the number of Irish ponds has declined dramatically, particularly after the introduction of a number of agricultural schemes that have followed the entry of Ireland into the European Union in 1973 (Burdon, 1986). A large proportion of ponds was likely lost due to arterial and field drainage schemes, under which approximately 30% of Ireland was drained by 1986 (Burdon, 1986), including areas of scientific importance (Feehan, 2003). The now suspended Farm Improvement Programme, which was introduced in 1986, resulted in further drainage works, with potential additional detrimental effects on biodiversity, although no information is available on the extent of the areas that have been reclaimed (Feehan, 2003).

1.1.3 Threats to Ponds and Pond Biodiversity

The main threats to ponds in Ireland and the biodiversity they support are associated with changes in land use, the intensification of agricultural practices and the reclamation of non-farmed habitats, and recent increases in urban and tourism developments in rural areas. The degradation of the ecological quality of existing ponds in Ireland is likely associated with heavy use by livestock, sewage disposal and dumping. Cattle, in particular, represent a major threat to invertebrate diversity; these are almost ubiquitous throughout the Irish agricultural landscape and their activity is often concentrated along the margins of ponds and streams. Cattle may affect the fauna and flora of water-bodies by trampling of the vegetation, breaking down of banks by their hooves, grazing of the emergent flora, and via manure falling into or near the water, causing further nutrient enrichment. Non-point sources of pollution, including nitrates, phosphates, ammonia, pesticides and herbicides represent a major threat to existing ponds and are already known to be the principal cause of water pollution and eutrophication of freshwater systems in Ireland, although a slight improvement in water quality has been reported for the past couple of years (Lucey, 2009). Studies on a number of lakes have shown that increasing eutrophic conditions typically cause increases in primary productivity and a rapid growth of filamentous algae, which, in turn, tend to result in conditions of hypoxia or anoxia due to the proliferation of bacterial decomposers, increases in turbidity, changes in aquatic pH, and increases in substrate, with subsequent effects on those species with specific oxygen demands and tolerance to nutrient concentrations. Eutrophic conditions in shallow waters may result in a substantial or complete loss of submerged plant communities, which are then replaced by dense phytoplankton communities. Moreover, some species of cyanobacteria produce toxins that are harmful to other organisms, with additional species losses.

1.1.4 Conserving Pond Biodiversity

Information on the ecological quality of ponds is important for improving our understanding of the role of these ecosystems in maintaining biodiversity at the local and regional level. The spread of invasive alien species also represents a potential threat to pond biodiversity, although little information is currently available on this issue in Ireland. Habitat fragmentation, which is associated with habitat destruction and degradation, also represents a major concern by decreasing habitat connectivity and by subsequently increasing the distance between populations. Metapopulation models (Hanski, 1999) predict that isolated populations are more prone to extinction than populations that are connected to each other, due to a loss of genetic diversity that can affect a population's ability to respond to environmental changes. Ultimately, the effects of a decrease in pond connectivity at the landscape level will also negatively affect pond biodiversity, with an overall impact that could be more important than the effects of pollution, disturbance and alien species (see Hanski, 1999).

Given the potential importance of ponds in maintaining biodiversity, there is a clear need for extensive research on the ecological quality of ponds, their distribution in the landscape, and on the local and regional factors that are most important in determining pond biodiversity patterns. A number of issues are however associated with the collection and processing of large-scale data on pond biodiversity. First, the heterogeneous nature of ponds at the local and regional level may complicate the search for consistent patterns and the identification of the principal drivers of biodiversity patterns in ponds. The large number of ponds compared to that of other freshwater ecosystem types (De Meester *et al*., 2005) makes the task of collecting large datasets on the ecological quality and biodiversity value of ponds even more complicated (Gioria *et al*., 2010a).

1.1.5 Aims of the Study

This study had three major aims: 1) to improve our understanding of the ecology of ponds and of the potential conservation value of ponds in the Irish agricultural landscape; 2) to develop an analytical framework that allows the identification of the main drivers of pond biodiversity; and 3) to develop an analytical protocol that allows the identification of surrogate taxa for biodiversity that can be used to make rapid or preliminary pond biodiversity assessments.

The potential conservation value of ponds was here evaluated using water beetles (Coleoptera) and wetland plants as indicator groups. These taxonomic groups were selected since they possess a number of characteristics that make them suitable surrogates for biodiversity in ponds. Plants represent an ideal surrogate group for biodiversity in terrestrial and aquatic systems, since they are relatively easy to identify compared to other taxonomic groups, and are sensitive to environmental conditions due to their sessile nature (Rodwell, 1995; Sætersdal *et al.*, 2003; Schaffers *et al.*, 2008). The use of water beetles in the evaluation of the conservation value of ponds and in rapid pond biodiversity assessments has recently been recommended (Menetrey *et al.*, 2005; Bilton *et al.*, 2006), due to a number of characteristics that make them a suitable biodivdicator. First, this is a speciose group, with at least 244 species recorded in Ireland (Foster *et al.*, 2009). Second, water beetles may contribute substantially to macroinvertebrate diversity (Biggs *et al.*, 2005) and are particularly numerous in temporary ponds in comparison to other invertebrates (Fairchild *et al.*, 2003). Third, their functional diversity is high and includes a range of predators, algivores, detritivores, consumers of filamentous algae, and herbivores (Eyre *et al.*, 1986). Water beetles are sensitive to changes in environmental conditions, and their distribution is largely dependent upon the characteristics of microhabitats (Fairchild *et al.*, 2003), including excess nutrient levels (Foster *et al.*, 1992; Menetrey *et al.*, 2005). Finally, they tend to be ubiquitous (Larson *et al.*, 2000) and are relatively easy to collect.

Since a rigorous characterization of the impact of environmental and management variables on pond biodiversity is critical for developing conservation and management strategies aimed at maximizing the biodiversity value of these ecosystems, a range of novel and traditional analytical procedures was used here to identify the factors that best explain and predict patterns in pond biodiversity. This information is urgently needed to halt the decline of freshwater biodiversity in Ireland.

1.2 Methods

1.2.1 Study Sites

The findings presented in this report refers to data collected from 54 ponds located in two intensively farmed regions in Ireland (R1, Wexford, Co. Wexford, 25 ponds: centroid: 52°23'N, 6°23W, June–July 2008; R2, Mullingar, Co. Westmeath, 29 ponds: 53°33'N, 7°25'W, June–July 2009). Altitude ranges from 24 to 66 masl in R1 and from 105 to 125 masl in R2. Mean monthly rainfall during the sampling period was 70.2–102 mm in R1 and 86.6–97.2 mm in R2 (Met Eireann, 2009), while average daily temperature ranged between 10.5 and 13.2°C in R1 and 14.4 and 15.1°C in R2. Mean monthly soil temperature for the same period was 16.6–18.3°C in R1 and 17.7–17.9°C in R2.

The number of ponds identified over the study area was substantially higher. However, only ~30% of the identified ponds could be sampled, due to difficulties in obtaining sampling permission from pond owners. The majority of farmers and land owners interviewed in Co. Wexford expressed concerns about the collection of water samples for chemical analysis. A more positive attitude towards biodiversity research was expressed by farmers and land owners in the Mullingar region.

Ponds were classified according to permanency into: 1) temporary (T, annual wet/dry phases), and 2) permanent (P), and according to the grazing intensity into 1) ungrazed (U = ponds surrounded by either arable land or by the presence of a buffer zone, 2) fenced (Fe = ponds surrounded by an electric fence that prevented direct access of livestock to the ponds, and 3) grazed (G) = ponds heavily used by cattle. Pond dominant substratum was classified into: 1) clay/mud (C) and 2) gravel (Gv).

1.2.2 Plant Surveys and Water Beetle Collection

Plants were recorded along the margin of the study ponds and in shallow waters, over the same surface area where water beetle samples were collected, i.e. an area comprised within 2 meters from the pond margin at the time of sampling, to ensure that the sampling surface for plants and beetles was comparable. The composition of the vegetation was evaluated at two levels: 1) plant species composition; and 2) plant community type composition (*sensu* Rodwell, 1995). If a comparable value of plant species composition and plant community type in predicting patterns of beetle species composition was found, there would be no need to identify the vegetation at the species level. This would obviate the need to identify plant species that are present in low abundance or difficult to identify, at least in preliminary surveys.

The abundance of each plant species and plant community type was expressed in terms of percentage cover within the sampled area. A small number of *ad hoc* community types not described by Rodwell (1995) were defined here and three additional habitat types were described (algae, detritus = dead vegetation, bare substratum = absence of vegetation). The inclusion of these additional community types/habitats allowed an improvement in understanding the spatial distribution of beetles in ponds. Plant nomenclature followed Stace (1997).

Adult water beetles were collected using a standard three-minute protocol, with sampling time (net-in-the-water time) divided equally among the microhabitats that had been identified at each pond (see Biggs *et al*., 1998). Typical microhabitats were represented by swamp communities, submerged vegetation, floating-leaved and free-floating plants, algal mats, bare muddy substratum, and flooded marginal grass. A 250 x 250 mm framed pond-net (mesh size: 1 mm) was vigorously swept in each microhabitat, while in shallow, densely vegetated waters (0–10 cm depth), a 15-mm diameter sieve (mesh size: 1 mm) was used as an alternative to the pond-net. The use of a smaller sampling device may have affected beetle abundance per unit of sampling time. However, since the diameter of the sieve was larger than the maximum pond depth in these shallow aquatic habitats and since the samples were collected over the same sampling time, differences in the abundance data recorded with this device were unlikely (Gioria *et al*., 2010a). After collection, the specimens were preserved in 70% ethanol and identified to species level. Beetle nomenclature followed Holmen (1987), Hansen (1999) and Nilsson (2001).

1.2.3 Data Analysis

A rigorous characterization of plant and beetle assemblages in the study ponds was made using similarity percentages analysis (SIMPER; Clarke, 1993). This technique was developed to identify the species that contribute most to dissimilarities between groups of samples defined *a priori*. SIMPER computes the average contribution of each species recorded in all the samples (ponds) to the overall dissimilarity between two groups (e.g. temporary and permanent ponds). SIMPER is based on Bray-Curtis similarity matrices (Bray and Curtis, 1957; the equation is described in Section 2.2.2). The ratio between the average contribution to the dissimilarity between groups and its standard deviation (SD) can be considered as a predictor of how consistently a species contributes to the overall dissimilarity. Therefore, species with a large ratio are those that best discriminate between two types of pond. In this study, SIMPER analyses were performed to quantify the contribution of each species, separately for plants and beetles, to the average between-group dissimilarity for pairs of ponds grouped according to 1) grazing intensity, 2) permanency, and 3) type of substratum.

1.3 Results

1.3.1 Wetland Plant Assemblages

A total of 67 wetland plant species (Table 1.3.1), grouped into 25 plant community types (Table 1.3.2), was recorded from the 54 study ponds, with 42 species classified as emergent, 7 as floating-leaved, and 18 as submerged. Mean values for each pond category are presented in Table 1.3.3. On average, 5.4 plant species (± 2.5 SD) were recorded in temporary ponds, compared to 11.1 (± 6.3 SD) found in permanent ones. *Potamogeton natans* L. was the pondweed most frequently recorded in permanent ponds (37 ponds), while the bulrush *Typha latifolia* L. formed dominant communities in the emergent vegetation of 19 permanent ponds.

Table 1.3.1. List of wetland plant species collected from 54 farmland ponds located in two regions in Ireland. Class: E = emergent; S = submerged; F = floating-leaved.

Taxa	Class		Class
Agrostis stolonifera L.	E	*Lolium perenne* L.	E
Alisma plantago-aquatica L.	E	*Mentha aquatica* L.	E
Alopecurus geniculatus L.	E	*Menyanthes trifoliata* L.	E
Apium nodiflorum (L.) Lag.	E	*Myosotis scorpioides* L.	E
Azolla filiculoides Lam.	F	*Myriophyllum aquaticum* (Vell.) Verdc.	S
Butomus umbellatus L.	E	*Myriophyllum spicatum* L.	S
Callitriche platycarpa Kuetz.	S	*Myriophyllum verticillatum* L.	S
Callitriche stagnalis Scop.	S	*Nuphar lutea* (L.) Smith	F
Carex pendula Huds.	E	*Nymphoides peltata* Kuntze	F
Carex riparia Curt.	E	*Persicaria hydropiper* (L.) Spach	E
Caltha palustris L.	E	*Persicaria maculosa* Gray	E
Chara denudata A.Br.	S	*Phalaris arudinacea* L.	E
Crassula helmsii (Kirk) Cockayne	F	*Phragmites australis* (Cav.) Trin. Ex Steud.	E
Elatine hexandra (Lapierre) DC.	E	*Polygonum amphibium* L.	S
Eleocharis palustris Roem. and Shult.	E	*Potamogeton natans* L.	F
Elodea canadensis Michx.	S	*Potamogeton obtusifolius* Mert. and Koch	S
Equisetum fluviatile L.	E	*Potamogeton polygonifolius* Pourr.	S
Equisetum palustre L.	E	*Potamogeton gramineus* L.	S
Galium palustre L.	E	*Potentilla anserina* L.	E
Glyceria fluitans (L.) R.Br.	E	*Potentilla palustris* (L.) Scop.	E
Hottonia palustris L.	E	*Ranunculus bulbosus* L.	E
Hippuris vulgaris L.	E	*Ranunculus flammula* L.	E
Hydrocotyle vulgaris L.	E	*Ranunculus hederaceus* L.	E
Hydrocotyle ranunculoides L.	E	*Rorippa nasturtium-aquaticum* (L.) Hayek	E
Hypericum elodes L.	E	*Rumex hydrolapathum* Huds.	E
Iris pseudacorus L.	E	*Scirpus lacustris* L.	E
Juncus articulatus L.	E	*Sparganium erectum* L.	E
Juncus bufonius L.	E	*Trifolium repens* L.	E
Juncus conglomeratus L.	E	*Typha latifolia* L.	E
Juncus effusus L.	E	*Utricularia vulgaris* L.	E
Juncus inflexus L.	E	*Valeriana dioica* L.	E
Lemna minor L.	F	*Veronica anagallis-aquatica* L.	E
Lemna minuta Kunth.	F	*Veronica beccabunga* L.	E
Lemna trisulca L.	F		

Table 1.3.2. Plant community types present (including algae, detritus, and bare substratum). Sub-communities were grouped into one community type.

Plant community types	
1	*Azolla filiculoides* community
2	*Callitriche stagnalis* community
3	*Carex riparia* community
4	*Equisetum fluviatile* community
5	*Glyceria fluitans* community
6	– *Alopecurus geniculatus* sub-community
	– *Glyceria fluitans* sub-community
7	*Iris pseudacorus* community
8	*Lemna minor* community
9	*Lemna minor, Lemna trisulca* sub-community
10	*Nuphar lutaea* community
11	*Phalaris arundinacea* community
	– *Epilobium hirsutum-Urtica dioica* sub-community
	– *Phalaris arundinacea* sub-community
12	*Phragmites australis* community
	– *Galium palustre* sub-community
	– *Menyanthes trifoliata* sub-community
	– *Phragmites australis* sub-community
13	*Polygonum amphibium* community
14	*Polygonum persicaria* community
15	*Potamogeton gramineus* community
16	*Potamogeton natans* community
17	*Potamogeton obtusifolius* community
18	*Sparganium erectum* community
19	*Typha latifolia* community
	– *Alisma plantago-aquatica* sub-community
	– *Mentha aquatica* sub-community
	– *Typha latifolia* sub-community
Ad-hoc community (not described by Rodwell, 1995)	
20	*Apium nodiflorum* community
21	*Chara denudata* community
22	*Elodea canadensis* community (including *Myriophyllum* species)
23	Grassland community: *Lolium perenne, Potentilla anserina, Agrostis stolonifera, Holcus lanatus*
24	*Juncus* community
25	*Rorippa nasturtium-aquaticum* community
Additional type	
26	Algae – *Cladophora* spp.
27	Detritus
28	Bare substratum

In temporary ponds, in particular, the average similarity in plant species composition was high (Fig. 1.3.1, Table 1.3.3), with four species (*Lolium perenne* L., *Agrostis stolonifera* L., *Lemna minor* L., and *Juncus effusus* L.) together explaining over 95% of these similarities in both regions. These species were also the major contributors to the vegetation of grazed ponds. Fenced ponds supported grasses and *Juncus* species typical of grazed and temporary ponds, as well as species characteristic of the swamp zone, such as *T. latifolia*. The species that contributed most to dissimilarities between ponds with a clay/muddy substratum and those with gravel in both regions were species such as *Elodea canadensis* Michx., *Lemna trisulca* L., and *Rorippa nasturtium-aquaticum* L. Hayek, all characteristics of permanent gravel ponds. The results of SIMPER analyses showing the contribution of individual species to the similarity/dissimilarity within and between pairs of pond grouped according to grazing intensity, permanency, and dominant substratum, are presented in Appendix I.

A number of invasive alien plants were recorded from a large proportion of study ponds. The most common invasive alien species was the Canadian pondweed *Elodea canadensis*. The parrot feather *Myriophyllum aquaticum* (Vell.) Verdc., the water fern *Azolla filiculoides* Lam., *Hydrocotyle ranunculoides* L. and *Crassula helmsii* (Kirk) Cockayne were also recorded from some of the study ponds.

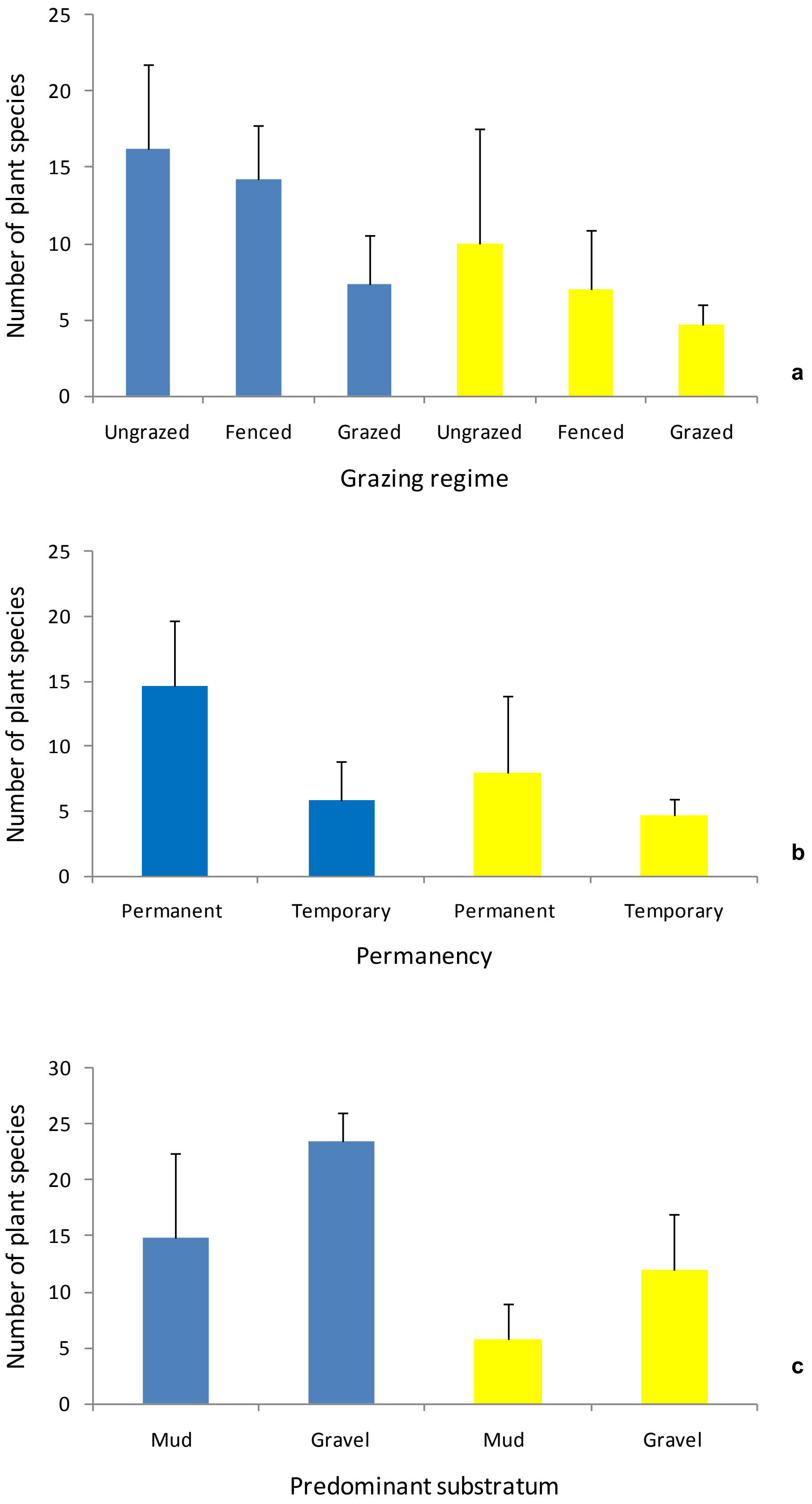

Figure 1.3.1. Mean number of plant species (± standard deviation) recorded from two regions (R1 = blue, R2 = yellow) within ponds classified according to a) grazing intensity, b) permanency and c) predominant substratum.

Table 1.3.3. Results of SIMPER analyses showing the average dissimilarity percentage between pairs of pond groups as well as the average similarity percentage within each pond group (P = permanent; T = temporary; U = ungrazed; G = grazed; F = fenced; M = mud or clay substratum; Gv = gravel substratum; R = region).

	Plant			Beetle		
	R1	**R2**	**R1+R2**	**R1**	**R2**	**R1+R2**
Dissimilarity %						
Permanency: P-T	93.2	73.2	79.7	64.0	78.9	73.0
Grazing: U-G	84.8	82.7	80.1	64.2	75.1	73.3
Grazing: U-F	69.6	85.2	77.2	51.8	70.5	70.9
Grazing: F-G	79.6	75.8	77.4	57.5	72.5	70.4
Substratum: M-Gv	76.8	75.8	81.0	59.0	76.0	63.6
Similarity %						
Permanency						
Permanent	30.1	23.0	23.0	47.1	31.5	30.4
Temporary	60.3	82.3	56.5	37.6	33.4	29.8
Grazing intensity						
Ungrazed	26.1	24.2	24.0	49.8	34.5	33.7
Grazed	26.2	41.2	31.5	34.2	27.9	27.2
Fenced	39.1	17.4	22.4	52.2	34.9	31.9
Substratum						
Mud/clay	23.3	23.8	24.0	42.5	31.4	30.6
Gravel	32.1	41.1	35.6	74.6	49.0	42.8

1.3.2 Water Beetle Species Assemblages

In total, 76 species were collected from the study ponds (Table 1.3.4), with the Dytiscidae (39 species, corresponding to 54.2% of the total species number), the Hydrophilidae (9 species), the Helophoridae (6 species), and the Haliplidae (7 species) being the most speciose families, forming multi-species communities in each pond. Five species are listed in the Red List of Irish Water beetles as IUCN Near Threatened, Vulnerable, or Data Deficient (Foster *et al.*, 2009), while this was the first record for two species, *Donacia crassipes* Fab. and *Donacia bicolora* (Zschach) in their respective regions (NBDC, 2009). Temporary ponds did not support any species unique to this habitat, although two recently created ponds (one in each region) supported diverse beetle assemblages (14 and 18 species). SIMPER analyses showed a moderate to high average dissimilarity between beetle assemblages of different pond groups (Table 1.3.3; Fig.1.3.2). The differences between grazed and ungrazed ponds were mainly explained by a small number of species with high abundances in grazed ponds, including *Helophorus* species, *Hydroporus planus* (Fab.), *Hydroporus palustris* L., and *Hygrotus inaequalis* (Fab.). The contribution of these species to the composition of fenced ponds was higher compared to that of ungrazed ponds, hence the high dissimilarity also between fenced and ungrazed ponds. The average dissimilarity between ponds with a clay/muddy substratum and those with gravel was mostly determined by a dominance of *Helophorus* species in the former, while *Hydroporus*, *Hygrotus* and *Anacaena* species dominated in gravel ponds (see Appendix II).

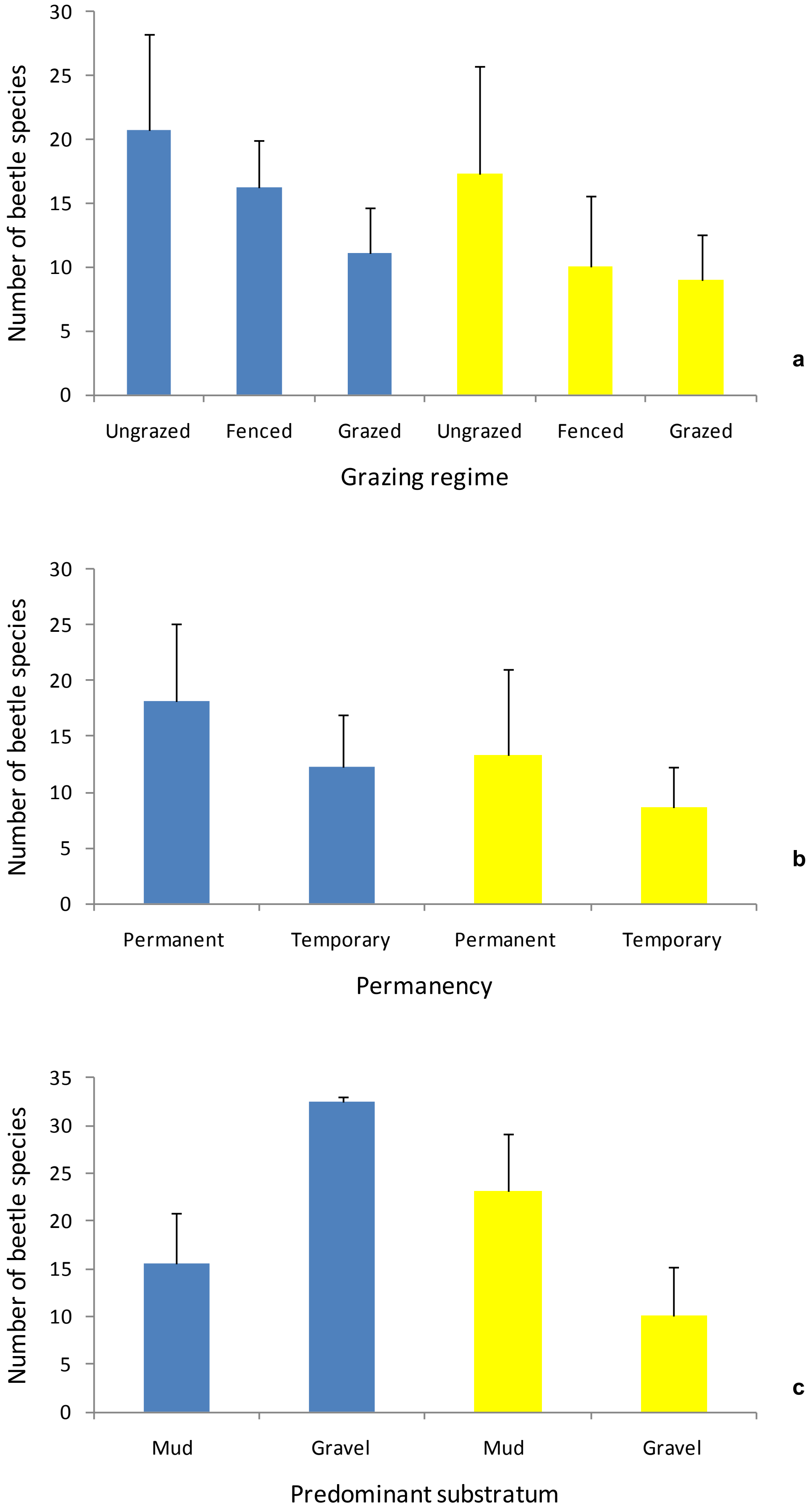

Figure 1.3.2. Mean number of beetle species (± standard deviation) recorded from two regions (R1 = blue, R2 = yellow) within ponds classified according to a) grazing intensity, b) permanency and c) dominant substratum.

Table 1.3.4. List of beetle species collected from 54 farmland ponds located in two regions in Ireland. The species in bold are listed in the Red List of Irish Water Beetles (from Foster *et al*., 2009).

Taxa
GYRINIDAE
Gyrinus caspius Ménétriés
Gyrinus substriatus Stephens
HALIPLIDAE
Haliplus confinis Stephens
Haliplus flavicollis Sturm
Haliplus fulvus (Fab.)
Haliplus immaculatus Gerhardt
Haliplus lineatocollis (Marsham)
Haliplus ruficollis (De Geer)
Haliplus sibiricus Motschulsky
NOTERIDAE
Noterus clavicornis (De Geer)
HYGROBIIDAE
Hygrobia hermanni (Fab.)
DYTISCIDAE
Acilius sulcatus (L.)
Agabus bipustulatus (L.)
Agabus nebulosus (Forster)
Agabus sturmii (Gyllenhal)
Colymbetes fuscus (Linnaeus)
Copelatus haemorrhoidalis (Fab.)
Dytiscus marginalis (Linnaeus)
Dytiscus semisulcatus Müller
Graptodytes pictus (Fab.)
Hydaticus seminiger (De Geer)
Hydroporus angustatus Sturm
Hydroporus erytrocephalus (L.)
Hydroporus incognitus Sharp
Hydroporus nigrita (Fab.)
***Hydroporus obsoletus* Aubé**
Hydroporus palustris (L.)
Hydroporus planus (Fab.)
Hydroporus pubescens (Gyllenhal)
Hydroporus striola (Gyllenhal)
Hydroporus tessellatus Drapiez
Hydroporus umbrosus (Gyllenhal)
***Hydrovatus clypealis* Sharp**
Hygrotus confluens (Fab.)
Hygrotus impressopunctatus (Schaller)
Hygrotus inequalis (Fab.)
Hygrotus quinquelinatus (Zetterstedt)
***Hygrotus versicolor* (Schaller)**
Hyphydrus ovatus (L.)
DYTISCIDAE
Ilybius ater (De Geer)
Ilybius fuliginosus (Fab.)
Ilybius quadriguttatus (Lacordaire)
Laccophilus minutus (L.)
Porhydrus lineatus (Fab.)
Rhantus exsoletus (Forster)
Rhantus frontalis (Marsham)
Rhantus grapii (Gyllenhal)
Rhantus suturalis (MacLeay)
Stictonectes lepidus (Olivier)
Suphrodites dorsalis (Fab.)
HYDRAENIDAE
Limnebius truncatellus (Thunberg)
HELOPHORIDAE
Helophorus aequalis Thomson
Helophorus brevipalpis Bedel
Helophorus grandis Illiger
Helophorus griseus Herbst
Helophorus minutus Fab.
Helophorus obscurus Mulsant
HYDROPHILIDAE
Anacaena globosus (Paykull)
Anacaena limbata (Fab.)
Anacaena lutescens (Stephens)
Cercyon convexiusculus Stephens
Enochrus coartatus (Gredler)
***Enochrus melanocephalus* (Olivier)**
Enochrus testaceus (Fab.)
Hydrobius fuscipes (L.)
Laccobius minutus (L.)
DRYOPIDAE
Dryops luridus (Erichson)
CHRYSOMELIDAE
Donacia bicolora Zschach
Donacia crassipes Fab.
***Donacia marginata* Hoppe**
Donacia simplex Fab.
Donacia versicolorea (Brahm)
Phaedon armoraciae (L.)
Prasocuris phellandrii (L.)
STAPHYLINIDAE
Lesteva longoelytrata (Goeze)

1.4 Discussion

The study ponds supported over 30% of the Irish water beetle fauna, and included uncommon and IUCN near threatened species, as well as species for which more information is required at the national level (Foster *et al*., 2009). This value is remarkable when considering that the total number of ponds sampled in this study represented only a sample of the ponds found in a relatively small area within two counties only. Moreover, approximately half of the study ponds were heavily grazed or temporary in nature, thus being characterized by high levels of nitrates and phosphates.

Species richness for both plant and beetle assemblages was higher in permanent ponds compared to temporary ponds and in ungrazed ponds compared to fenced ponds. The flora and fauna of grazed ponds was similar to that of temporary ponds, as indicated by the results of SIMPER analyses, which allowed identifying the species that best characterize each pond category and showed that grazing intensity, permanency, and dominant substratum play a major role in determining the composition of both plant and beetle assemblages.

Direct access of cattle to ponds had a major impact on plant and beetle assemblages, via the grazing and trampling of the vegetation, the direct deposition of fecal material in the water and around the pond margins, and by creating large areas of bare substratum. The latter also increase water turbidity and the movement of nutrients in the water. Indirectly, the presence of cattle increases the input of nutrients in and around a pond, which ultimately flows into the water as runoff. The vegetation of grazed ponds was very similar to that of temporary ponds, and was principally characterized by the presence of common grass species such as *Lolium perenne* and by *Juncu*s species, which are not palatable to cattle and were often recorded in over-logged marginal areas in a field. Fenced ponds supported some marginal vegetation, although its abundance was dependent upon the distance between the pond margin and the fence system (only one farm had a functioning electric fence around four ponds). Fenced ponds were also characterized by high levels of nutrients, and supported a large abundance of pondweeds (mostly *Potamogeton* species).

Ungrazed permanent ponds, with a gravel substratum and/or dominated by swamp plant communities supported the richest beetle assemblages. Swamp areas dominated by the bulrush *Typha latifolia* L. supported a large number of beetle species, particularly predaceous diving beetles. The presence of the bur-reed *Sparganium erectum* L. was consistently associated with leaf beetles of the genus *Donacia*, which confirmed to be strongly associated with this plant species (Cox, 2007; Foster *et al*., 2009). *Donacia* species are probably unrecorded in Ireland, due to practical difficulties associated with their observation and collection, including the relatively short period of adult activity (Nelson *et al*., 2007; Foster *et al*., 2009), thus their general uncommon status. Four leaf beetles were recorded in this study: 1) *Donacia marginata* Hoppe, which has a IUCN near threatened status, is uncommon and confined to stagnant water (Foster *et al*., 2009), 2) *Donacia bicolora* Zschach, 3) *Donacia versicolora* (Brahm), and 4) *Donacia crassipes* Fab. One species, *D. bicolora,* was recorded for the first time in Co. Westmeath from a shallow pond heavily grazed by bullocks, suggesting that even impaired ponds may play an important conservation role at the landscape level. The importance of species such as the bur-reed and the bulrush for the conservation of freshwater invertebrate diversity should be accounted for when developing management and conservation strategies.

Temporary ponds supported only a few species and none unique to this habitat type. These ponds were highly degraded, as they were all grazed or located in heavily grazed fields. However, these ponds may be important in improving the overall habitat connectivity at the landscape level and may also play an important role in maintaining beetle diversity at the landscape level. Their potential importance for the conservation of freshwater biodiversity was indicated by the presence of the dytiscid *Acilius sulcatus* (L.) in two ponds that were created only two months prior to data collection. This species has been found to sense the presence of fish by using chemical cues (Åbjörnsson *et al*., 1997), thus suggesting that temporary and newly created small ponds may be important to the maintenance of beetle species that avoid predators restricted to permanent waters.

2 Identifying the Drivers of Pond Biodiversity

2.1 Introduction

The conservation of ponds ultimately relies on a solid understanding of the principal determinants of pond biodiversity (Gioria *et al.*, 2010b). Extensive research has been conducted on the effects of selected environmental variables on community patterns for certain taxonomic groups. Pond biodiversity has been found to be a function of regional variables as well as of processes of dispersal and speciation, land-use history, and degree of habitat patchiness (Heino, 2000; Wood *et al.*, 2003; Céréghino *et al.*, 2008). Local factors such as dominant substratum, eutrophication, and hydro-period also play a central role in determining patterns of pond biodiversity (Friday *et al.*, 1987; Gee et al., 1997; Heino, 2000; Fairchild *et al.*, 2003; Céréghino *et al.*, 2008). Contrasting results have, however, been found when assessing the relationship between pond biodiversity and factors such as pond size (area and/or depth), pond age, or water chemical gradients (Gee *et al.*, 1997; Heino, 2000; Oertli *et al.*, 2002; Gioria *et al.*, 2010a).

The difficulties in making generalizations on the main drivers of pond biodiversity have often been attributed to the heterogeneity in the biotic and abiotic conditions that characterize each pond; random colonization events; and a broad tolerance to abiotic gradients for many taxa (Heino, 2000; Jeffries, 2008). Although it is undeniable that such factors affect the strength of biotic–abiotic relationships substantially, the difficulties in finding consistent patterns could at least partly stem from the statistical approaches that have been used to traditionally analyse pond biodiversity data.

Identifying the most appropriate analytical procedures for analysing the effects of a range of environmental and management variables on pond biodiversity is critical for assessing the contribution of environmental conditions to determining patterns in pond biodiversity. Therefore, one aim of this study was to develop a rigorous and robust analytical protocol that could be used to analyse pond biodiversity data. Specifically, the aim was to evaluate and compare the potential differences that may arise from the use of a range of analytical procedures, including: 1) differences in the number of variables selected in minimal adequate models; 2) differences in the contribution of each selected variable to the variance in the study response variable; 3) differences in the overall variance explained by the minimal adequate models; and 4) differences generated by the use of species richness versus species composition as the response variable to model. This information is critical to a sound interpretation of any theoretical model and to the development of effective management practices and conservation programmes based on rigorous quantitative information.

2.2 Data Analysis

To perform this modeling exercise, biological and environmental data for a subset of the study ponds were used (21 ponds in R1; 24 ponds in R2; see Gioria et al., 2010b). Vegetation data were modeled using three categorical variables: 1) grazing intensity (grazed, fenced, and ungrazed); 2) pond dominant substratum (clay/mud; gravel); and 3) pond age (2–10 years; >10 years). Nine continuous variables were also measured and used in the analyses for each pond: 1) pond surface area, which was measured at the water level on the sampling dates; 2) maximum pond depth; 3) maximum depth of sampling; 4) conductivity; 5) pH; 6) alkalinity; 7) ammonia (NH_3-N); 8) NO_3-N; and 9) PO_4-P. Conductivity (corrected to 25°C), pH, and dissolved oxygen were measured *in situ* by portable meters. Alkalinity, ammonia, and nutrient concentrations were measured using a photometer chemistry analyser, after filtering the samples (Whatman nylon filters, 0.2 µm pore, see American Public Health Association, 1989). Water samples for laboratory analyses were collected on the same dates as plants and water beetles. Chemical analyses were performed on the day of collection, to avoid considerable changes in the concentrations of the parameters investigated. To model beetle data, two additional continuous explanatory variables were used: plant species richness and plant vegetation cover over the sampled area (see Appendix III for details of the management and environmental variables at the study sites). For details on the models described here see Gioria *et al.* (2011).

2.2.1 Modelling Species Richness

Three statistical procedures were used to investigate the relationship between species richness and the sets of explanatory variables, separately for plants and beetles: 1) a generalized linear model (GLM; McCullagh and Nelder, 1989); 2) a parametric multiple regression model based on a permutational forward selection procedure (FP; Blanchet *et al*., 2008); and 3) non-parametric distance-based multivariate analysis for linear models (DISTML; Anderson, 2001; McArdle and Anderson, 2001).

Species richness and environmental data were log-transformed prior to data analyses. Multi-collinearity between pairs of variables was examined using the Pearson's correlation coefficient (r) as well as variance inflation factor analysis (VIF; Montgomery and Peck, 1982), which was performed on the full models. The maximum correlation between variables was always below 0.60, for both response variables (beetle and plant species richness), and individual VIF values were never above 10. Therefore, all the explanatory variables were retained in the full models.

For the GLM, the Poisson model family was used to model species richness (applying the log-link function to relate the mean value of the response variables to their linear explanatory variables; Crawley, 1993; Guisan and Zimmermann, 2000). The construction of the reduced models was based on a number of steps:

1. The deviance for the null model was computed to calculate the value of the intercept, while a full model was constructed using all the explanatory variables to quantify the total variance in the response variable explained by all the explanatory variables;
2. An iterative stepwise (backward and forward) procedure was then used to identify the variables to be included in the reduced model (see Guisan and Zimmermann, 2000);
3. The Akaike Information Criterion (AIC; Hastie and Pregibon, 1993) and the Bayesian Information Criterion (BIC; Schwarz 1978) as the criteria of model selection were used;
4. The χ^2 statistic was used to test the significance of each variable retained in the reduced model ($\alpha = 0.05$);
5. The goodness-of-fit of the selected model was evaluated using the adjusted deviance D^2 (Guisan and Zimmermann, 2000);
6. The χ^2 test was performed to test whether there were statistically significant differences between the full and the reduced model;
7. A weighted analysis of deviance was conducted to evaluate the performance of the selected model in predicting the relationship between species richness and the predictor variables (McCullagh and Nelder, 1989);
8. A leave-one-out cross-validation was used to calculate the prediction error for the GLMs (Guisan and Zimmermann, 2000);
9. The correlation between the values predicted by the reduced model and the observed values of the response variables was calculated using the coefficient of correlation (r).

A second parametric approach was used to evaluate the contribution of environmental variables to the variation in species richness for plant and beetle assemblages, that is a multiple regression model based on a recently developed forward selection procedure (Blanchet *et al*., 2008). This approach was used since it had been found to prevent two well-known problems associated with the use of classic forward selection, that is 1) the overestimation of the explained variance and 2) an inflated Type I Error. This forward procedure performs a forward selection by permutation of residuals under the reduced model and is based on two stopping criteria: 1) the significance level α (here set at 0.05) and 2) an adjusted coefficient of determination (maximum R^2_{adj}), calculated by constructing a full model inclusive of all explanatory variables.

Finally, a distance-based linear regression model based on permutations, which performs a distance-based analysis on a linear model for any dissimilarity matrix (McArdle and Anderson, 2001), was used to test the significance of biotic–abiotic relationships. A forward selection of the predictor variables, running 9999 tests by permutation ($\alpha = 0.05$), using the Euclidean distance as a measure of dissimilarity between pairs of samples, was applied here. This approach was used to compare the results of parametric and non-parametric approaches since it allows the quantification of differences between pairs of samples (ponds) using any dissimilarity or distance measure of choice.

2.2.2 Modeling Species Composition

Patterns in species composition (sample x species matrices) were modeled using four analytical procedures: 1) traditional canonical correspondence analysis (CCA), using a stepwise variable selection; 2) a forward selection procedure described by Blanchet *et al*. (2008); 3) BIO-ENV (Clarke and Ainsworth 1993); and 4) DISTLM (forward selection of the predictor variables, 9999 permutations).

Two measures of dissimilarity were used to measure the dissimilarity between pairs of samples (ponds) based on species composition data in non-parametric models: 1) the Bray-Curtis dissimilarity measure (d_{BC}; Bray and Curtis, 1957) and 2) a modified Gower distance (d_{MG}), proposed by Legendre and Legendre (1998). Whilst the d_{BC} was selected because of its useful properties in the analysis of community data and its wide use in ecological studies (Clarke, 1993; Legendre and Legendre, 1998), the d_{MG} was used since it has the advantage of being explicit about the contribution of differences in species identity and relative abundances.

Equation 2.1 shows how the Bray-Curtis measure (Bray and Curtis, 1957) can be used to define dissimilarity between two samples as:

$$d_{BC} = \frac{\sum_{i=1}^{p} | y_{i1} - y_{i2} |}{\sum_{i=1}^{p} (y_{i1} - y_{i2})}, \qquad \text{(Eqn 2.1)}$$

where y_{i1} is the abundance of the i^{th} species in sample 1, y_{i2} is the abundance of the i^{th} species in sample 2, and p is the total number of species across both samples. This measure is widely used in ecological studies, since it satisfies a series of desirable criteria, such as:

- It varies between 0 when the samples have no species in common to 1 when two samples are identical;
- A change of one measurement unit to each variable in each sample does not affect its value;
- It ignores species which are absent from two samples (joint absences). This condition, which many coefficients fail to meet, was very important in this study, since each site comprised a different set of species, therefore many vectors (variables) in the matrices were represented by zero values;
- It is flexible in detecting differences in total abundance even when the relative abundances for all species are identical (see Clarke, 1993; Clarke and Warwick, 2001).

The modified Gower dissimilarity (Legendre and Legendre, 1998) is defined in Eqn. 2.2 as:

$$d_{MG} = \frac{\sum_{i=1}^{p} w_i \, | y'_{i1} - y'_{i2} |}{\sum_{i=1}^{p} w_i}, \qquad \text{(Eqn 2.2)}$$

where the weights w_i were used to exclude joint absences by setting $w_i = 0$ when $y_{i1} = y_{i2} = 0$ and $w_i = 1$ elsewhere.

Bray-Curtis dissimilarities have intrinsic sample standardization, which tends to decrease the dispersion among groups that have large abundance values. The Gower distance measure does not have such standardization. Since the choice of the resemblance measure may affect the outcomes of multivariate analyses, the results obtained using these three measures were compared.

Traditional CCA (ter Braak, 1986) was used to identify the variables that best explain patterns in plant and beetle assemblages. Freshwater ecologists have made extensive use of CCA to evaluate biotic-abiotic relationships in ponds and lakes (e.g. Nicolet *et al.*, 2004, Studinski and Grubbs, 2007), despite the fact that using CCA to analyse community data has long been discouraged due to the implicit use of chi-square distances as the measure of dissimilarity between pairs of samples (e.g. Clarke 1993, Legendre and Legendre, 1998). The use of multivariate non-parametric or semi-metric approaches has therefore been strongly recommended (Clarke, 1993; Anderson, 2001). These methods include the BIO-ENV procedure proposed by Clarke and Ainsworth (1993) and permutational multivariate analysis of variance (PERMANOVA) (Anderson, 2001).

BIO-ENV was used here since it allows the identification of the combination of environmental variables that maximize the correlation between a biotic and an environmental data matrix (Clarke and Ainsworth, 1993). The Spearman rank coefficient of correlation (ρ_s) was used as the measure of correlation between biotic and environmental variables, while the Euclidean distance was used as the measure of dissimilarity between pairs of samples for environmental data (9999 random permutations of all combinations of variables).

PERMANOVA (Anderson, 2001) was used prior to performing these analyses to test whether the factor ‘region’ (two levels, R1 and R2) had any significant effects on beetle and plant assemblages. Since the effects of regional differences were not significant, this factor was not included in the analyses.

2.3 Results

Biotic-abiotic relationships, both for the plant and the beetle community, varied largely, from weak to strong, depending on 1) the typology of modeling approach used; 2) parametric *versus* non-parametric models; 3) the response variable (species richness *versus* species composition), 4) the model (three models of species richness, four models of species composition), 5) the measure of dissimilarity used in the non-parametric multivariate approaches, and 6) the model selection criterion in the GLM and the DISTLM (AIC *versus* BIC).

2.3.1 Modelling Species Richness

The GLM and DISTLM models explained a similar amount of variation (70–73% for beetles and 53–56% for plants), despite selecting a different number of variables and the different contribution percentage of each selected variable to the overall explained variance (Tables 2.3.1 and 2.3.2). The FP regression model selected a lower number of variables compared to the GLM and the DISTLM, and explained a lower amount of variation (63% and 48% of variation in beetle and plant species richness, respectively, Table 2.3.3). The model selection criterion in the GLM (AIC *versus* BIC) had only a marginal effect on the overall explained variance, while a substantial difference was evident in the number of variables included in the reduced models. In the DISTLM models, the use of the BIC as compared to the AIC resulted in the selection of a lower number of variables, each of which explained the same amount of variance (Table 2.3.3). A similar result was obtained when species composition data was used as the response variable (Table 2.3.4).

Table 2.3.1. Summary of the results of generalised linear models (GLM) (Poisson family) log-link function, for 1) beetle and 2) plant communities (model selection criterion: a) Akaike Information Criterion [AIC], b) Bayesian Information Criterion [BIC]). The variables grazing intensity and substratum are categorical (from Gioria *et al.*, 2010b).

Variable	Deviance reduction	Coefficient value	VIF**	$p(\chi^2)$
1a) Beetle species richness, AIC, D^2_{adj} = 0.727				
alkalinity	64.31	-	1.94	***
substratum	28.48		1.27	***
max. depth	19.01	+	1.47	***
grazing	8.27		1.87	*
pH	4.71	+	1.10	*
Cross-validation estimate of prediction error: 19.5				
r = 0.8872, difference between full and minimal GLM, $p(\chi^2)$ = 0.969				
1b) Beetle species richness, BIC, D^2_{adj} = 0.697				
alkalinity	64.31	-	1.215	***
max. depth	22.75	+	1.149	***
substratum	27.97		1.170	***
Cross-validation estimate of prediction error: 20.81				
r = 0.8872, difference between full and minimal GLM, $p(\chi^2)$ = 0.518				
2a) Plant species richness, AIC, D^2_{adj} = 0.565				
pond age	29.21		1.63	***
pond area	20.62	+	1.45	***
max. sampling depth	14.93	+	1.97	***
substratum	14.07		1.37	***
alkalinity	13.99	-	3.77	***
grazing	9.64		2.22	**
Cross-validation estimate of prediction error = 37.9				
r = 0.8060, Difference between full and minimal GLM, $p(\chi^2)$ = 0.973				
2b) Plant species richness, BIC, D^2_{adj} = 0.538				
pond area	35.56	+	1.146	***
max. sampling depth	41.58	+	1.075	***
substratum	14.41		1.072	***
Cross-validation estimate of prediction error = 21.39				
r = 0.8060, Difference between full and minimal GLM, $p(\chi^2)$ = 0.218				

* = $p < 0.05$; ** = $p < 0.01$; *** = $p < 0.001$

** variance inflation factor, which was below 10 for all variables, thus there was no issue of multicollinearity among variables

Table 2.3.2. Summary of results of distance-based linear model (DISTLM) analyses used to model the relationship between the explanatory variables and species richness for a) beetle and b) plant communities. Data were log-transformed and the Euclidean distance was applied as a measure of dissimilarity between samples (9999 permutations).

Variable	Variance	Cumulative variance	F**	P**
a) Beetle species richness, $R^2 = 0.7304$				
alkalinity	0.3828	0.3828	26.669	0.001
area	0.1681	0.5509	15.718	0.001
mud	0.1022	0.6530	12.073	0.001
max. depth	0.0430	0.6960	5.656	0.026
pH	0.0343	0.7304	4.967	0.030
b) Plant species richness, $R^2 = 0.5882$				
max. sampling depth	0.4165	0.4165	30.689	0.001
gravel	0.1143	0.5308	10.231	0.005
area	0.0574	0.5882	5.714	0.032

** F and P show the contribution of each significant variable to the variation in the response variables explained by the final models

2.3.2 *Modeling Species Composition*

Models of species composition explained a lower percentage of variation compared to models of species richness (Tables 2.3.3 and 2.3.4). When modeling species composition, the FP regression models, the DISTLM_AIC, and the BIO-ENV models selected a similar number of variables, although the FP reduced models explained a lower percentage of variance, particularly for plant species composition. The use of the AIC *versus* the BIC did affect the results of multivariate analyses. In the DISTLM_BIC model, only one to two variables were selected, compared to up to seven variables included in the DISTLM_AIC.

The choice of the dissimilarity measures (d_{BC} versus d_{MG}) in the non-parametric models (BIO-ENV and DISTLM) affected the overall explained variance in species composition, for both beetle and plant data, but did not affect the identity of the selected variables or their order of contribution to the explained variance (Table 2.3.4).

The CCA models retained only two significant variables in both communities (Table 2.3.4; Fig. 2.3.1). All the models included depth-related variables and type of substratum among the most significant determinants of patterns in species composition, for both plant and beetle communities. The variable ‘maximum sampling depth’ was selected as the most important driver of patterns in beetle and plant species composition by the DISTLM_AIC and the BIO-ENV models, while the FP regression model of beetle species composition and CCA (both for plants and beetles) emphasized the contribution of ‘maximum pond depth’ to the total explained variance.

Table 2.3.3. Summary of the results of multiple regression models based on a modified forward selection procedure (Blanchet *et al*., 2008) for a) beetle and b) plant data (Gioria *et al*., 2010b).

Variable	R^2_{adj}	Cumulative R^2_{adj}	F**	P**
a) Beetle community				
Species richness				
alkalinity	0.3938	0.3938	27.934	0.0001
plant species richness	0.1458	0.5396	13.296	0.0006
gravel	0.0588	0.5690	6.009	0.0193
max. depth	0.0687	0.6338	8.254	0.0067
Species composition				
max. depth	0.1226	0.1226	7.148	0.0001
mud	0.1176	0.2402	7.658	0.0001
plant species richness	0.0373	0.2776	3.171	0.0001
PO_4-P	0.0225	0.3000	2.316	0.0011
alkalinity	0.0156	0.3157	1.914	0.0071
pH	0.0160	0.3317	1.932	0.0066
b) Plant community				
Species richness				
max. sampling depth	0.3667	0.3667	24.901	0.0001
pond area	0.1115	0.4782	8.9763	0.0045
Species composition				
max. sampling depth	0.0913	0.0913	5.423	0.000
mud	0.0361	0.1274	2.777	0.000
pond area	0.0244	0.1518	2.207	0.002
conductivity	0.0193	0.1710	1.953	0.010
age >10	0.0130	0.1840	1.637	0.038

** F and P show the contribution of each significant variable to the variation in the response variables explained by the final models

Table 2.3.4. Summary of results of distance-based linear models (DISTLM), BEST, and canonical correspondence analysis (CCA), used to model the relationship between the explanatory variables and species composition for a) beetle and b) plant communities. The asterisk indicates the variables that were selected using the Bayesian Information Criterion (BIC) criterion of model selection (maxSdepth = maximum sampling depth). Dissimilarity measure: BC = Bray–Curtis dissimilarity; MG = modified Gower distance (from Gioria *et al.*, 2010b).

DISTLM	F**	P**	Cumul. variance	DISTLM	F**	P**	Cumul. variance
a) Beetle species composition							
AIC_BC				AIC_MG			
maxSdepth*	6.202	0.001	0.1236	maxSdepth*	5.150	0.001	0.1048
mud*	5.763	0.001	0.2271	mud*	4.899	0.001	0.1963
PO$_4$-P	2.485	0.008	0.2703	PO$_4$-P	2.059	0.002	0.2339
conductivity	2.694	0.002	0.3153	max. depth	2.130	0.001	0.2717
alkalinity	1.972	0.020	0.3475	conductivity	1.924	0.004	0.3051
max. depth	1.944	0.031	0.3784				
pH	1.895	0.036	0.4080				
b) Plant species composition							
AIC_BC				AIC_MG			
maxSdepth*	7.903	0.001	0.1523	maxSdepth*	4.442	0.001	0.0917
mud	3.618	0.002	0.2180	mud	2.510	0.001	0.1418
area	2.560	0.004	0.2630	area	1.877	0.012	0.1785
conductivity	2.507	0.003	0.3054	conductivity	1.919	0.006	0.2152
BEST analysis				CCA	χ^2	F	P
a) Beetle species composition				a) Beetle species composition			
$\rho_{BC} = 0.381$		$\rho_{MG} = 0.481$		$\chi^2 = 0.3842$			
maxSdepth		maxSdepth		max. depth	0.191	3.388	<0.005
conductivity		conductivity		mud	0.193	3.427	<0.005
PO$_4$-P		PO$_4$-P					
plant species richness		plant species richness					
mud		Mud					
b) Plant species composition				b) Plant species composition			
$\rho_{BC} = 0.288$		$\rho_{MG} = 0.276$		$\chi^2 = 0.4690$			
maxSdepth		max. sam. depth		maxSdepth	0.276	2.899	<0.005
pH		pH		mud	0.192	2.017	<0.005
mud		conductivity					

** F and P show the contribution of each significant variable to the variation in the response variables explained by the final models

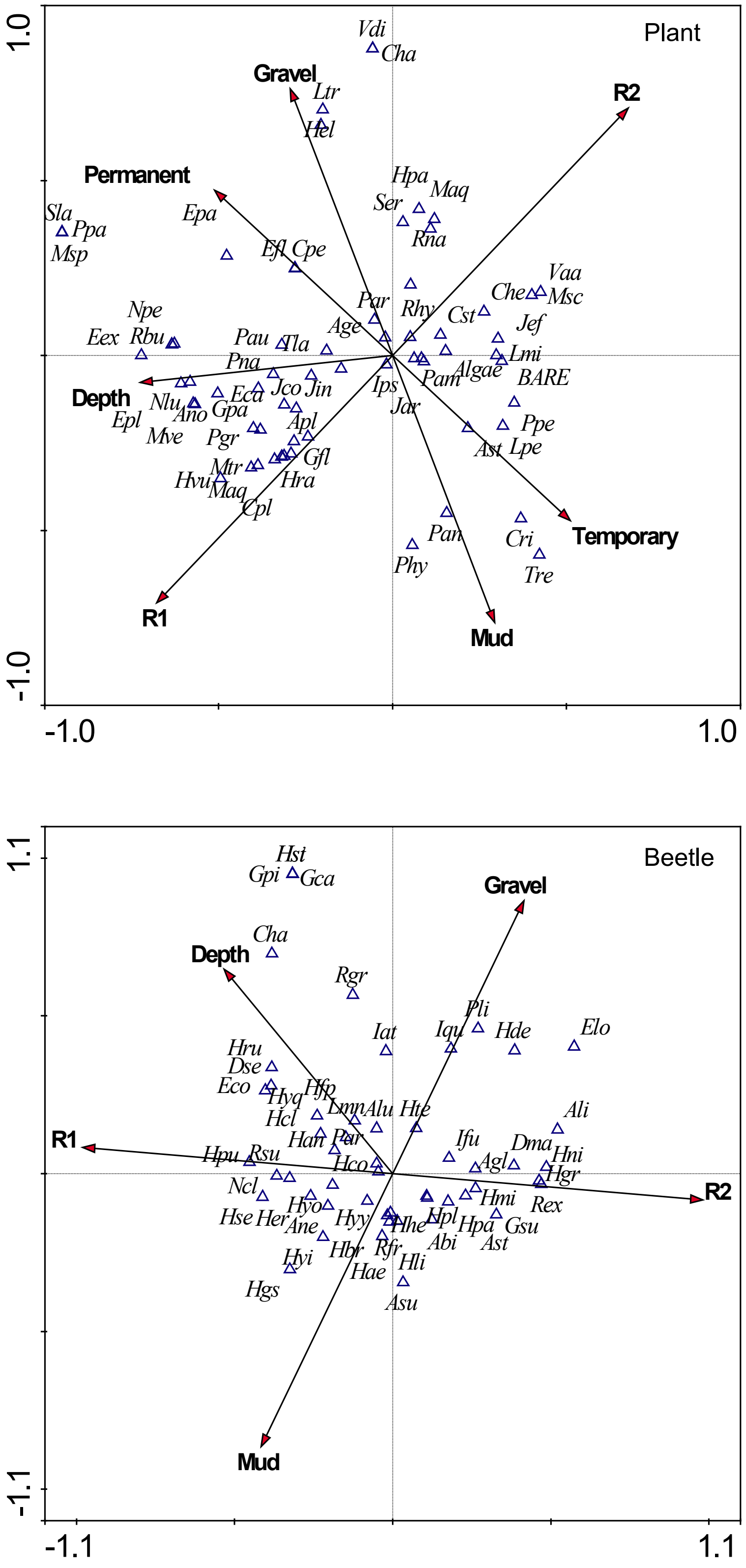

Figure 2.3.1. Canonical correspondence analysis (CCA) biplots for explorative reduced models (after stepwise environmental variable selection based on the Akaike Information Criterion [AIC]), based on beetle and plant community composition data. Full names of plant and beetle species in Appendix IV (from Gioria *et al.*, 2010a).

2.4 Discussion

The relationship between plant and beetle assemblages with a range of environmental local and regional variables ranged from weak to strong. Indeed, the use of different analytical approaches and response variables (species richness versus species composition) affected the overall variance explained by the reduced models, the identity of the variables selected in the reduced models, the total number of retained variables, as well as the contribution of the selected environmental variables to the explained variance (see Gioria *et al.*, 2010b, 2011).

Models of species richness explained a larger percentage of variation compared to models of species composition. These models showed a strong relationship between beetle species richness and environmental variables, while a moderate relationship was found for plant species richness. The lower percentage of variation explained by the models of species composition was expected, due to the higher complexity of multispecies abundance, consistent with previous investigations in terrestrial systems (e.g. Guisan *et al.*, 1999; Su *et al.*, 2004). The higher variability of species composition data was reflected in the fact that only two explanatory variables were never selected by any of the models of species of composition. Some variables were significant determinants of patterns in species composition (e.g. pond age) but not of those in species richness, providing additional evidence that species richness may not summarize community patterns exhaustively, as suggested previously (Clarke, 1993; Su *et al.*, 2004; Gioria and Osborne, 2009).

When modeling species richness, the forward procedure proposed by Blanchet *et al.* (2008) resulted in the selection of a lower number of significant variables compared to GLM and explained a lower percentage of variation, indicating that this method does indeed avoid calculating an over-inflated variance (see Gioria et al., 2010b). The differences/advantages of these procedures were not, however, evident when modelling patterns in species composition.

The use of parametric versus non-parametric approaches to model species richness (GLM and FP versus DISTLM) also resulted in differences in the number, identity, and contribution of the significant variables to the overall explained variance. The GLM and the DISTLM explained virtually the same amount of variation, probably because species richness was normalized prior to the data analyses, allowing the use of a parametric measure of dissimilarity (Euclidean distance) in the DISTLM.

All the models selected pond-depth related variables and substratum among the significant determinants of plant and beetle community patterns, consistent with previous studies on plant and invertebrate assemblages in ponds and lakes (e.g. Rodwell, 1995; Fairchild *et al.*, 2003; Pakulnicka, 2008). CCA was the multivariate method that selected the lowest number of significant variables (pond-depth related measures and substratum) and did not include ‘pond surface area’ among the significant drivers of vegetation patterns. Some of the study models emphasized the role of maximum pond depth as a determinant of biodiversity patterns, while other models selected maximum depth of sampling amongst the most important determinants of such patterns. Despite being depth-related, these measures encompass different information, as indicated by their weak relationship in permanent ponds. While maximum sampling depth provides information on the aquatic-terrestrial transition zones of a pond, maximum pond depth affects the presence of emergent and submerged plant species. Thus, the selection of only one pond depth-related measure may have important implications for the development of conservation strategies. The contribution of chemical variables to patterns in beetle and plant species composition was not always evident in the reduced models, consistent with previous investigations in ponds (Gee *et al.*, 1997; Nicolet *et al.*, 2004; Heino, 2000; Jeffries, 2008). This is likely due to the absence, in the ponds used for this study, of large chemical gradients and extreme conditions, as well as to a broad tolerance of the recorded species to physico-chemical variables (Gioria *et al.*, 2010a).

The differences in the measure of dissimilarity used in the non-parametric models and in the model selection criterion (AIC *versus* BIC) affected both the overall explained variance and the number of variables included in the reduced models. Although the use of multiple dissimilarity measures and/or criteria of model selection is not suggested, this study showed that ecologists must be aware that their choice in relation to the use of different analytical methods, measures of dissimilarity between samples, and criteria of model selection may affect substantially the outcomes of any model and must be based on considerations on the type of data they intend to analyse. The same applies to the choice of the transformation applied to both biotic and abiotic data prior to the analysis.

A number of recommendations can be drawn from this study. First, the interpretation of the ecological significance of the results of only one theoretical model must be based on field observations, and multiple analytical approaches should be used in the analysis of biodiversity patterns. Each method has advantages and drawbacks in comparison to others. In a comparison of CCA and GLM models of plant species distribution, Guisan *et al*. (1999) concluded that while CCA gives a broader view of ecological gradients in an area (particularly in the presence of rare species), GLM provides better species-specific models, although both approaches showed a similar ranking of model quality. In a study on littoral ascidians, Naranjo *et al*. (1996) compared the results of CCA and BIO-ENV and suggested a combined use of these methods, despite their being conceptually different. Since the BIO-ENV procedure includes a stopping rule when ρ_s decreases with the inclusion of unimportant variables, these authors recommended the use of BIO-ENV to select the variables that would be subsequently used as the explanatory variables in a CCA.

The use of multiple taxonomic groups may aid in the interpretation of the results of theoretical models aimed at assessing the effects of environmental variables on biotic communities. Here, the combined use of plant and beetle data provided useful information on the direct as well as the potential indirect effect of the environmental variables on the ecological quality of the study ponds. The role of vegetation in providing food, shelter from predators, and a physical structure to invertebrate communities has been reported in both aquatic and terrestrial systems (e.g. Foster *et al*., 1990, 1992; Schaffers *et al*., 2008), and the strength of vegetation patterns in predicting water beetle species composition has been recently quantified (Gioria *et al*., 2010a).

Information on species identity is central to a correct interpretation of the results of any theoretical model. Here, the variable ‘pond surface area’ was not always selected among the determinants of plant or beetle community patterns. The role of species identity could be included in theoretical models by using weights accounting for a species’ rarity and conservation value. This information could already be available for certain regions and taxonomic groups (Foster *et al*., 1990).

3 Cross-taxon Congruence Analysis

3.1 Introduction

3.1.1 *Identifying Surrogate Taxa for Biodiversity*

The collection of large-scale biodiversity data in freshwater systems is essential for the conservation of biodiversity. Ideally, a systematic programme of pond biodiversity data collection should be developed and implemented, should we wish to maintain and protect freshwater biodiversity. This is particularly true for Ireland, where the majority of its surface area is farmed. The collection of extensive information on pond biodiversity is hampered by the complexity of spatio-temporal dimensions of this variable, and by financial constraints and by often limited taxonomic expertise (see Gioria *et al*., 2010a, 2011 and references therein). Surveying biodiversity presents difficulties associated with limited time and financial resources (Gaston and Williams, 1996; Howard *et al*., 1998), and often requires a large degree of expertise, particularly for certain taxonomic groups. The large number of ponds compared to that of freshwater ecosystem types (De Meester *et al*., 2005) makes the task of surveying pond biodiversity even more complicated. In recent years, the need to make biodiversity monitoring and conservation planning more efficient has led to an increasing interest in the identification of correlates for biodiversity (Noss, 1990; Margules and Pressey, 2000), using cross-taxon congruence analysis (Prendergast *et al*., 1993; Howard *et al*., 1998; Su *et al*., 2004). Recent evidence shows, in fact, that the use of surrogate taxa in conservation planning is substantially more effective than that of surrogates based on environmental data only (Rodrigues and Brooks, 2007).

The effectiveness of the use of one taxon to predict community patterns for other taxonomic groups ultimately depends upon the strength of the relationship among such groups and on its underlying mechanisms (e.g. Prendergast *et al*., 1993; Sætersdal *et al*., 2003). The factors that affect cross-taxa relationships include 1) a similar but independent response of two taxonomic groups to the same set of environmental conditions; 2) trophic interactions or functional interdependence; 3) a shared bio-geographical and evolutionary history at a large/global scale; and 4) species-energy relationships (Gaston and Williams, 1996; Heino, 2010).

Cross-taxon congruence analysis can be expressed as the correlation in patterns of species richness and/or diversity (Pearson and Carroll, 1999) between multiple taxonomic groups, or, in a multi-species context, as community concordance, i.e. the relationship among composition (abundance) patterns of multiple taxonomic groups across sites (Jackson and Harvey, 1993; ter Braak and Schaffers, 2004). To date, cross-taxa relationships have been characterized using groups belonging to the same taxonomic rank, such as between different invertebrate groups (Briers and Biggs, 2003; Bilton *et al*., 2006). Relationships between multiple taxonomic or functional ranks have been evaluated, particularly in an attempt to evaluate whether vascular plants can be used as surrogates for other groups such as invertebrates, birds, and bryophytes (Prendergast *et al*., 1993; Sætersdal *et al*., 2003; Su *et al*., 2004). A quantification of congruence in species richness or composition between different taxa has been proved to be useful in the identification of hotspots of biodiversity (Prendergast *et al*., 1993; Howard *et al*., 1998), although evidence of consistent patterns that can justify the use of one taxon to predict patterns for other taxonomic groups effectively is rather weak (Rodrigues and Brooks, 2007; Heino, 2010).

The difficulties in finding predictive patterns of congruence between different taxonomic groups have often been attributed to the spatial scale at which these relationships have been investigated (Pearson and Carroll, 1999). While strong cross-taxa relationships have been identified for certain groups at the regional scale (Prendergast *et al*., 1993; Howard *et al*., 1998; Gaston, 2000), contrasting results have been identified at the local/community level (e.g. Bilton *et al*., 2006). These differences depend upon the choice of the variable(s) used to evaluate such relationships, such as species richness versus abundance or species occurrence data, with patterns in species composition often stronger than those in species richness, both in terrestrial (Su *et al*., 2004; Schaffers *et al*., 2008) and aquatic ecosystems (Bilton *et al*., 2006).

To date, a range of methods has been used to assess cross-taxa relationships (Gioria *et al*., 2011). The correlation between two or more taxonomic groups has been typically quantified using simple regression analysis applied to univariate indices of diversity, such as species richness, traditional diversity indices (Shannon Diversity or Simpson Dominance), or taxonomic distinctness indices (e.g. Prendergast *et al*., 1993; Howard *et al*., 1998; Su *et al*., 2004), using either the Pearson or the Spearman rank correlation, depending on the error distribution of the data to analyse. Rarefaction (Colwell and Coddington, 1994) has also been used to compare species richness among multiple taxonomic groups across different habitats (see Chiarucci *et al*., 2008). With regard to species composition, a recent review by Heino (2010) showed that Mantel tests (Mantel, 1967; Manly, 1997) and Procrustes analysis (Gower, 1971) were the methods most frequently used for evaluating cross-taxon congruence in freshwater ecosystems.

The Mantel approach has been widely used by ecologists to test hypotheses concerning the degree of association between two or more data matrices (Legendre and Legendre, 1998; Peres-Neto and Jackson, 2001). This is a robust and flexible method that calculates the concordance between two datasets by correlating dissimilarity or distance matrices calculated using any resemblance measure of choice. However, the use of resemblance matrices rather than the original ones as the starting point of the correlation calculations has been much criticized (Dutilleul *et al*., 2000; Peres-Neto and Jackson, 2001; Legendre and Fortin, 2010). To address this issue, Jackson and Harvey (1993) suggested the use of Procrustes analysis (Gower, 1971) as an alternative approach to calculate the degree of association between ecological data matrices. This is a family of multivariate ordination techniques that aims to calculate the correlation among data matrices using a rotational-fit algorithm that attempts to minimize the total sum-of-squared residuals between data values in two matrices using a series of steps, including centering, scaling, reflection, rotation, and dilation (see Jackson, 1995; Legendre and Legendre, 1998). Randomization tests such as the Procrustean Randomization tests (PROTEST; Jackson, 1995) have then been used to test the significance of the Procrustean fit, i.e. the measure of matrix concordance (m^2). One of the advantages of this method over the Mantel approach is the possibility of using the results of multivariate ordination analyses as the input data matrices, thus allowing a reduction in the dimensionality of the original data (Peres-Neto and Jackson, 2001). The majority of applications of this approach to the analysis of cross-taxa relationships have used the resulting axes of gradient analyses, including principal components analysis, canonical analysis, detrended correspondence analysis, or those of non-parametric ordination methods such as non-metric multidimensional scaling, as the input data matrices (see Gioria *et al*., 2011 and references therein).

An ordination method used specifically to predict one biological community from another – co-correspondence analysis (Co-CA) – was recently developed by ter Braak and Schaffers (2004). This method relates two community data matrices directly by maximizing the weighted covariance between weighted averages species scores of one community and weighted averages species scores of the other community. Two versions of Co-CA are available: (i) a symmetric explanatory version and (ii) a predictive one, based on weighted averaging and partial least squares regression (ter Braak and Schaffers, 2004). The predictive version of Co-CA has been used to assess the value of vascular plants in predicting invertebrate community patterns in terrestrial (Schaffers *et al*., 2008) and in freshwater systems (Gioria *et al*., 2010a).

A number of indirect approaches have also been used to analyse cross-taxa relationships. For instance, congruence in composition has been evaluated by calculating pair-wise correlations of the resulting ordination axes of indirect gradient analyses or reciprocal averaging (e.g. Maccherini *et al*., 2009). Other approaches have included the calculation of the correlation between the results of DCA performed on community data and a set of environmental variables (Sætersdal *et al*., 2003).

3.1.2 Quantifying the Relationship between Different Taxonomic Groups

To evaluate whether wetland plants can be used as a surrogate group for pond biodiversity in the agricultural landscape, a rigorous evaluation of the degree of association in community patterns between wetland plants and water beetles was made. If the composition of water beetles was consistently correlated to that of wetland plants, it would be possible to assess the potential conservation of ponds by using data on plants only (Gioria *et al*., 2010a). Since plants are generally identified more rapidly than invertebrates and tend to respond more rapidly to ecological disturbance due to their sessile nature (e.g. McIntyre *et al*., 1999), they could be used in preliminary and cost-effective biodiversity assessments, allowing the collection of large-scale data sets on the distribution and biodiversity of ponds within a relatively short period of time compared to that required to survey and identify water beetles and/or other invertebrate groups. The predictive strength of the vegetation was evaluated at three levels of detail: 1) plant species composition; 2) plant community type composition; and 3) plant species richness and evenness, expressing a decreasing order of variability in the data and of complexity in the data collection, taxonomic identification, and analysis.

Specifically, this investigation aimed at: 1) evaluating the degree of correlation in community patterns between water beetles and wetland plants; 2) quantifying and comparing the capacity of plant community data and of environmental variables to predict water beetle species composition; and 3) evaluating and comparing the response of plant and beetle assemblages to the same set of environmental conditions. This information is critical in order to identify suitable surrogate taxa for biodiversity and to develop cost-effective management and conservation strategies that allow sampling one or more taxonomic groups to infer information on another taxonomic group and on the ecological quality of ponds. It is also central to improving our understanding of cross-taxon congruence patterns and plant–animal relationships.

3.2 Data Analysis

3.2.1 *Species Richness and Univariate Patterns*

A detailed description of the methods and procedures used in this study can be found in Gioria *et al.* (2010a) and in Gioria *et al*. (2011). One-way ANOVA was used to test the differences in species richness between plant and beetle assemblages in the study ponds. The Pearson correlation coefficient (*r*) was used to measure congruence in species richness between plants and beetles, after performing a Shapiro-Wilk test of normality. An indirect comparison of patterns in species richness for the two groups was also made using rarefaction (Sanders, 1968; Colwell and Coddington, 1994). Pond-based rarefaction curves were calculated as the mean of 9999 accumulation curves obtained with different random sequences of ponds, for plants and beetles, both collectively and separately for each region. The formula proposed by Kobayashi (1974) was used to calculate the rarefaction curves (see Chiarucci *et al*., 2008) (Eqn. 3.3):

$$\mathrm{E}(S_i) = S_n - \binom{n}{i}^{-1} \sum_{k \in \mathrm{G}} \binom{n-n_k}{i} \quad i = 1, \ldots, n \qquad \text{(Eqn 3.3)}$$

where G is the set of species observed in the collection of *n* samples (ponds), S_n is the total number of observed species and n_k is the number of samples containing at least one individual of species $k \in$ G, and S_i is the expected species richness for the sub-sample *i* out of the total number of samples *N*. The ratio between the species rarefaction curve obtained for plants and those calculated for water beetles was also calculated, to compare patterns of rarefaction for the two communities, both separately and collectively for the two regions.

3.2.2 *Predicting Community Patterns*

The strength of plant community data and environmental variables in predicting patterns in water beetle species composition (response variable, unimodal) was evaluated using two ordination techniques (ter Braak and Schaffers, 2004 and Schaffers *et al*., 2008): 1) predictive Co-CA and 2) predictive canonical correspondence analysis (partial least squares extension; CCA-PLS), depending on the predictive data set (vegetation or environmental data).

Co-CA directly relates two community compositions by maximizing the weighted covariance between weighted averages (WA) species scores of one community (plant) and WA species scores of the other community (water beetle), attempting to identifying the ecological gradients that are common to both communities (ter Braak and Schaffers, 2004; Schaffers *et al*., 2008). In this study, both the symmetric (explanatory) version and asymmetric (predictive) versions of Co-CA were used. Predictive Co-CA in particular combines two approaches, i.e. weighted averaging and partial least squares (PLS; see Martens and Naes, 1992). The species scores of the water beetle community were obtained as WA of the site scores of the plant set, and the species scores of the plant set as WA of the site scores of the beetle community, while all site scores were WA of the species scores of their own set. A leave-one-out cross-validatory procedure was performed to obtain the cross-validatory fit for a different number of axes solutions and to select the minimal adequate predictive models. For this, a Co-CA model was calculated *n* times (with n = number of ponds), leaving out one site at a time. Each time, the composition of the left-out site is calculated (predicted) using the predictor community composition and the obtained model parameters. This way a predicted composition was obtained for all sites, which was subsequently compared to the actual composition. The cross-validatory fit (%) used to assess the accuracy of the prediction was calculated as: 100 x (1 - ssp_a/ssp_0), where ssp_a represents the sum of squared prediction errors of the model and ssp_0 is the sum of squared prediction errors under the null model of no relationship (see ter Braak and Schaffers, 2004). Schaffers *et al*. (2008) pointed out that, due to its predictive nature, any cross-validatory fit > 0 implicitly validates the model, indicating that prediction is better than that obtained under the null model.

The advantage of predictive Co-CA is that it can be used to compare its results with those of predictive CCA directly (i.e. CCA–PLS; ter Braak and Schaffers, 2004), thus allowing the evaluation of whether data for one community or environmental data best predict community patterns for another taxonomic group. Here, CCA-PLS was performed to assess the strength of 1) environmental variables only and that of 2) the combined environmental variables and plant diversity (species richness (*S*) and evenness [*J'*, Pielou, 1975], in predicting beetle species composition. CCA-PLS uses PLS routines, allowing the use of many predictive variables (ter Braak and Schaffers, 2004). These CCA-PLS analyses were also performed using a leave-one-out cross-validation (ter Braak and Schaffers, 2004), allowing the comparison of the results of Co-CA and CCA-PLS based on the same cross-validatory fit measure (Schaffers *et al*., 2008). To assess whether the differences between predictive Co-CA and CCA-PLS models were significant, two-tailed randomization tests were performed. Specifically, the significance of the differences between the mean squared prediction errors of pairs of models was tested. Site prediction errors were rearranged by swapping the prediction errors for random sites between the two models (a different random set at a time) using 9999 permutations.

Co-CA and CCA-PLS analyses were also performed removing singletons (i.e. species for which only one individual was recorded from only one site) from both the beetle and plant datasets, to indirectly assess the contribution of rare species in determining the predictive patterns. Since the removal of singletons from these data sets had only a minimal effect on all predictions and the general patterns remained similar, the results of these analyses are not presented here. It should be borne in mind that cross-validatory fit measures obtained for predictive methods are usually considerably lower than those obtained using explained variation in explanatory methods.

Table 3.2.1. Variables, transformations, and analytical methods used to predict water beetle species composition. CoCA = predictive co-correspondence analysis; CCA-PLS = predictive canonical correspondence analysis-partial least squares. Superscripts refer to the number of continuous variables (a) or categorical (dummy) variables (b).

Predictive variables	No. variables	Transformation	Method
Plant community composition			
Plant species composition	67	log(x+1)	CoCA
Plant community type composition	28	log(x+1)	CoCA
Plant diversity			
Species richness (*S*) and evenness (*J'*)	2	log(x+1)	CCA-PLS
Environmental variables	14 ($9^a + 5^b$)	log(x+1)	CCA-PLS
Environmental variables + plant diversity	20	log(x+1)	CCA-PLS

Plant species composition (percentage cover), the composition of plant community types (percentage cover), and beetle species composition (counts) were $\log_e(x + 1)$-transformed prior to analyses. Continuous environmental variables were also $\log_e (x + 1)$-transformed, while categorical variables (permanency, grazing intensity, substratum, and region) were used as dummy variables (each level of a factor = a dummy 0/1 variable). In total, there were 14 environmental variables (9 continuous and effectively 5 dummies resulting from the 4 categorical variables). Table 3.2.1 presents a summary of the predictive data sets used, of the transformation applied to data prior to analyses, and of the methods used to predict water beetle species composition.

3.2.3 Explaining Community Patterns

Congruence in species composition was evaluated using 1) Mantel tests; 2) Procrustes analysis; and 3) symmetric Co-CA. Mantel tests were performed using 1) the standard parametric approach, based on the Pearson correlation and 2) a non-parametric approach based on the Spearman rank correlation (Clarke, 1993). For both approaches, Monte Carlo randomizations (see Jackson, 1995; Manly, 1997; 9999 permutations) were used to test the significance of the correlation between the two resemblance matrices, which were calculated using the Bray–Curtis dissimilarity (Bray and Curtis, 1957).

The input data matrices used in Procrustes analyses are presented in Table 3.2.1. Non-metric multidimensional scaling (NMDS; Kruskal and Wish, 1978; 100 iterations) was used to summarize patterns in the plant and beetle datasets. Information on the stress value, that is, the degree of correspondence between the distances among points in NMDS configurations and the dissimilarity matrix, was used to determine the number of axes to be employed as the input data of Procrustes analyses.

Principal components analysis (PCA; Chatfield and Collins, 1980) was used to reduce the dimensionality of the environmental data into a few principal components representing the main environmental gradients (Peres-Neto and Jackson, 2001). The correlation between the scores of PCAs and the NMDS scores derived for the plant and the beetle species composition datasets, separately, were then calculated using Procrustes analyses. Finally, the Procrustean approach was used to calculate the correlation between the results of CCA (ter Braak, 1986), performed separately on the plant and beetle data. This allowed a further evaluation of the degree of association between plant and beetle assemblages based on their response to the same set of environmental conditions. In Procrustes analyses, congruence between plants and beetles was calculated using plant data as the independent variable (reference configuration) and beetle data as the dependent (response) variable. The measure of matrix concordance is the Procrustean fit m^2, that is the sum-of-square residuals between matrices in their optimal superimposition (Gower, 1971). The statistical significance of m^2 for each pair of matrices was assessed using PROTEST (PROcrustean randomization TEST; Jackson, 1995). For a full description of the analytical procedures and references used in this study see Gioria *et al*. (2011).

3.3 Results

3.3.1 Species Diversity Patterns

A moderate correlation in species richness (S_{plant} = 66; S_{beetle} = 76) was found in the study ponds ($R^2 = 0.3692$ $F_{1,52}$ = 34.3, $P < 0.001$; Fig. 3.3.1). Results of one-way ANOVA showed that the number of beetle species in the study ponds was significantly higher than that of plant species ($F_{1,52}$ = 21.1, $P < 0.001$). The rarefaction curves calculated for each region are presented in Fig. 3.3.2. Given the same number of ponds sampled, patterns of rarefaction were similar in both regions. The ratio of the results of rarefaction (S_{plant}/S_{beetle}) stabilized after approximately 10–15 ponds, in both regions, indicating that plant and beetle species richness tend to have similar rarefaction patterns after a minimum threshold of ponds (Fig. 3.3.3).

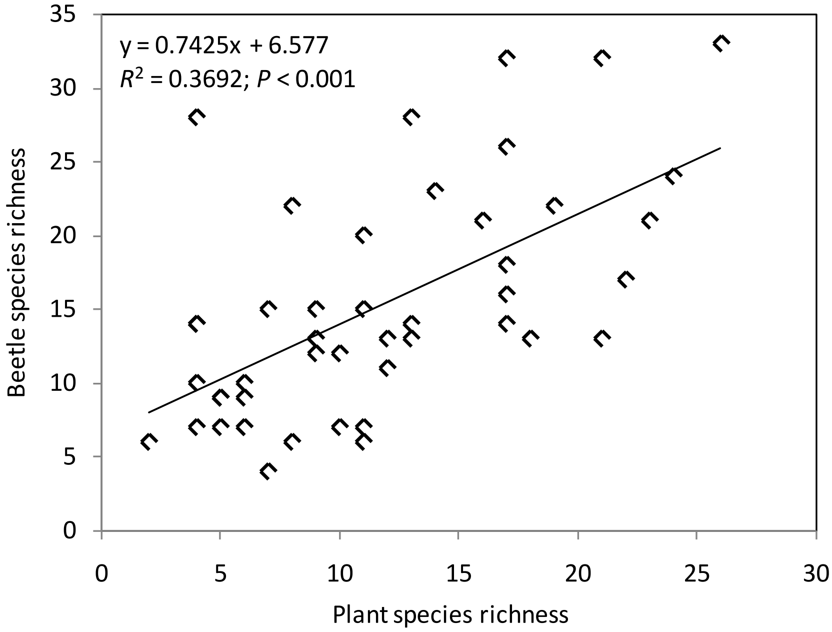

Figure 3.3.1. Relationship between water beetle and wetland plant species richness, based on the Pearson correlation, for data recorded from 54 farmland ponds in Ireland.

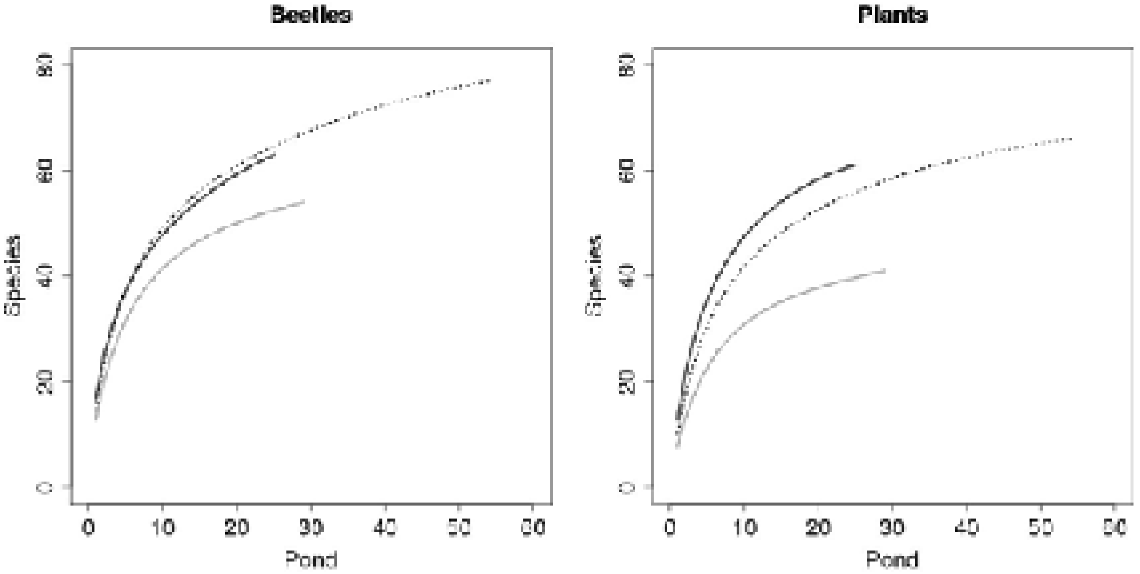

Figure 3.3.2. Pond-based rarefaction curves calculated for plant and beetle assemblages recorded from 54 farmland ponds in Ireland, separately for two regions, showing the expected number of species E(*S*) as a function of the number of ponds (R1 = Region 1, 25 ponds, black line; R2 = Region 2, 29 ponds, grey line). The pond-based rarefaction curve for the pooled sample of all 54 ponds is also shown (dashed line) (from Gioria *et al.*, 2011).

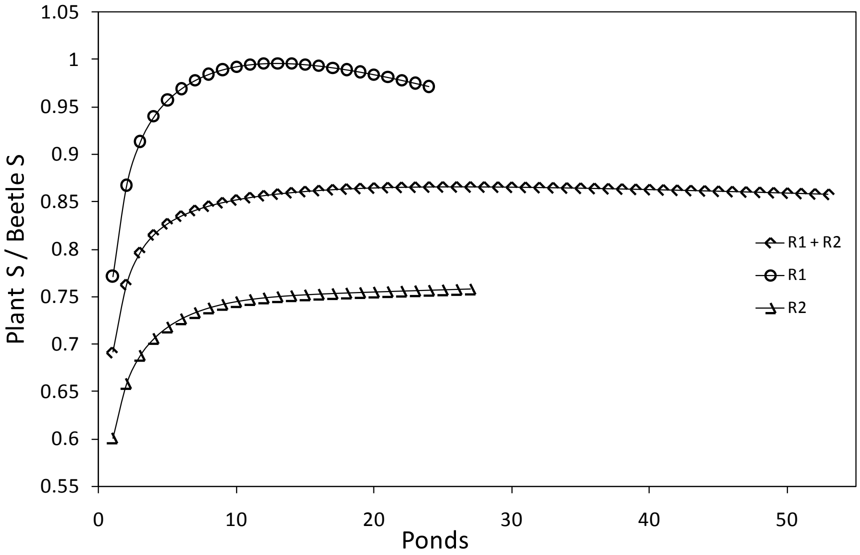

Figure 3.3.3. Ratios of water beetle and wetland plant rarefaction curves, for assemblages collected from 54 farmland ponds (R1 = Region 1, 25 ponds; R2 = Region 2, 29 ponds) (from Gioria *et al.*, 2011).

3.3.2 *Predicting Water Beetle Species Composition*

The prediction accuracy (cross-validatory fit %) of plant species composition, plant community type composition, and environmental variables were all positive (Table 3.3.1), indicating that predictions of water beetle species composition based on these predictive variables were better than those expected under the null model of no relationship (Gioria *et al.*, 2010a). Plant species composition predicted 5.34% of the variation in water beetle species composition using the two-axis solution (local maximum; global maximum was 5.41% at four axes). The first axis of Co-CA was significant ($P < 0.0002$), while P was greater than 0.08 for the second. Plant species composition performed significantly better than plant community type composition ($P < 0.04$, Table 3.3.1). The model based on the plant community type composition had a predictive strength of 3.36%, with the first axes significant ($P < 0.02$), while $P = 0.07$ for the second axis.

Plotting the cross-validatory fit percentage for all the compared predictive datasets against the number of axes showed a pattern that was common to all the data sets, with the maximum prediction level usually obtained at two axes (Fig. 3.3.4). A bi-modal pattern was observed only for plant species composition, but the global maximum was only marginally higher than the local maximum achieved at two axes. In all selected models, therefore, only the first two axes were retained.

Environmental variables predicted 7.53% of the variation in beetle species composition, with the first two axes significant ($P < 0.005$). Despite the fact that predictive accuracy of environmental variables was higher than that obtained using plant species composition data, the differences between the two models were not significant ($P = 0.134$, **Table 3.3.1**). The predictive strength of plant diversity, measured by species richness (S) and plant evenness (J'), was minimal. Environmental variables and diversity indices together predicted a value very similar to that obtained for environmental variables only (7.69%, $P < 0.005$ for the first two axes, the difference between the two CCA-PLS models was not significant).

Table 3.3.1. Results of predictive co-correspondence analysis (CoCA) and canonical correspondence analysis (partial least squares extension) (CCA-PLS) analyses showing the value of plant community composition, plant structure, environmental variables, and environmental variables combined with plant diversity (richness *S* and the Pielou evenness index *J'*), as predictors of water beetle species composition.

	Cross-validatory fit (%)	No. of axes retained
Plant species composition (Co-CA)	5.34 a	2
Plant community type composition (Co-CA)	3.36 b	2
Environmental variables (CCA-PLS)	7.53 a	2
Environmental variables + plant diversity (CCA-PLS)	7.69 a	2

Prediction levels are expressed as % cross-validatory fit. The significance of differences (*P* = 0.05) between pairs of models (two-tailed) was obtained using simple randomization tests (9999 random site permutations). The same letter indicates that the two values do not differ significantly.

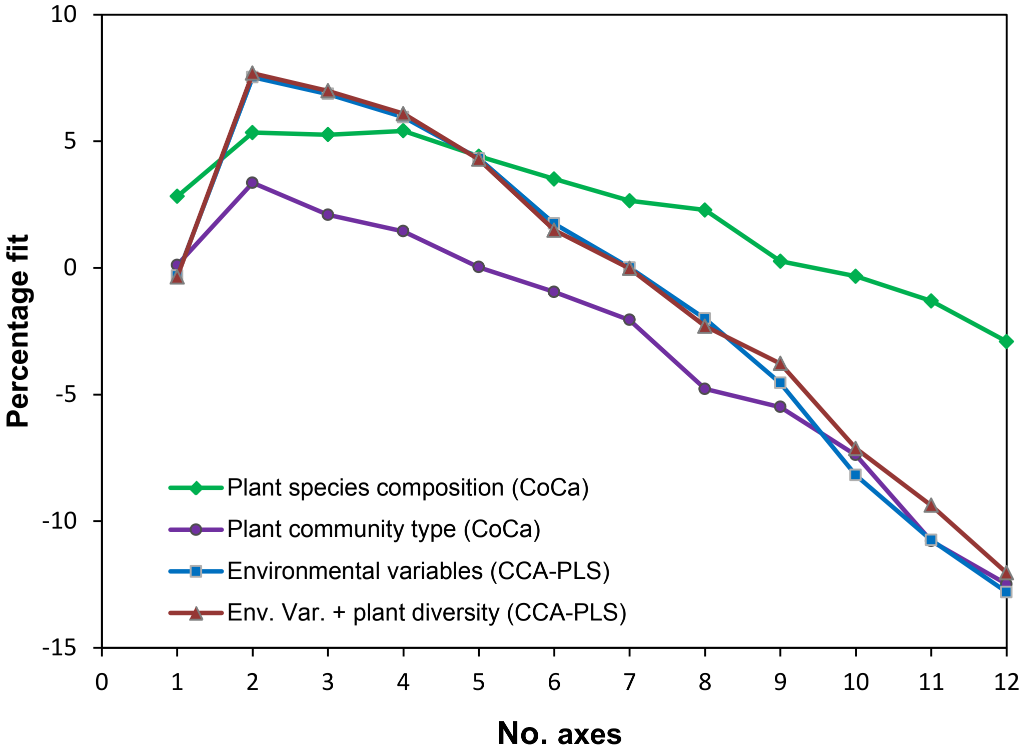

Figure 3.3.4. Cross-validatory fit of the water beetle data set against the number of ordination axes for different predictor data sets. Positive values indicate that prediction by the model, using that particular number of axes, is better than the null model prediction (the overall average species composition) (Gioria *et al*., 2010a).

The ordination diagrams of Co-CA analysis are presented in Fig. 3.3.5a and b. In the plots, temporary ponds were all grouped together (top right), while a gradient of grazing intensity could be identified along Axis 2. This allowed the identification of those beetle species that are consistently associated with certain plant species. Here, a number of beetle species, such as *Enochrus testaceus*, *Hygrotus versicolor*, *Porhydrus lineatus*, and *Donacia marginata*, were consistently associated with species such as *Phalaris arundinacea*, *Myriophyllum aquaticum*, and *Typha latifolia* in ungrazed permanent ponds. Conversely, temporary and grazed ponds were dominated by agricultural weeds, such as *Lolium perenne*, *Juncus effusus*, and *Polygonum persicaria*, and supported beetle assemblages dominated by herbivores of the genus *Helophorus.* These results were consistent with those found separately for each pond category, for both beetle and plant assemblages, using SIMPER analyses (Table 3.3.1, Appendices I and II).

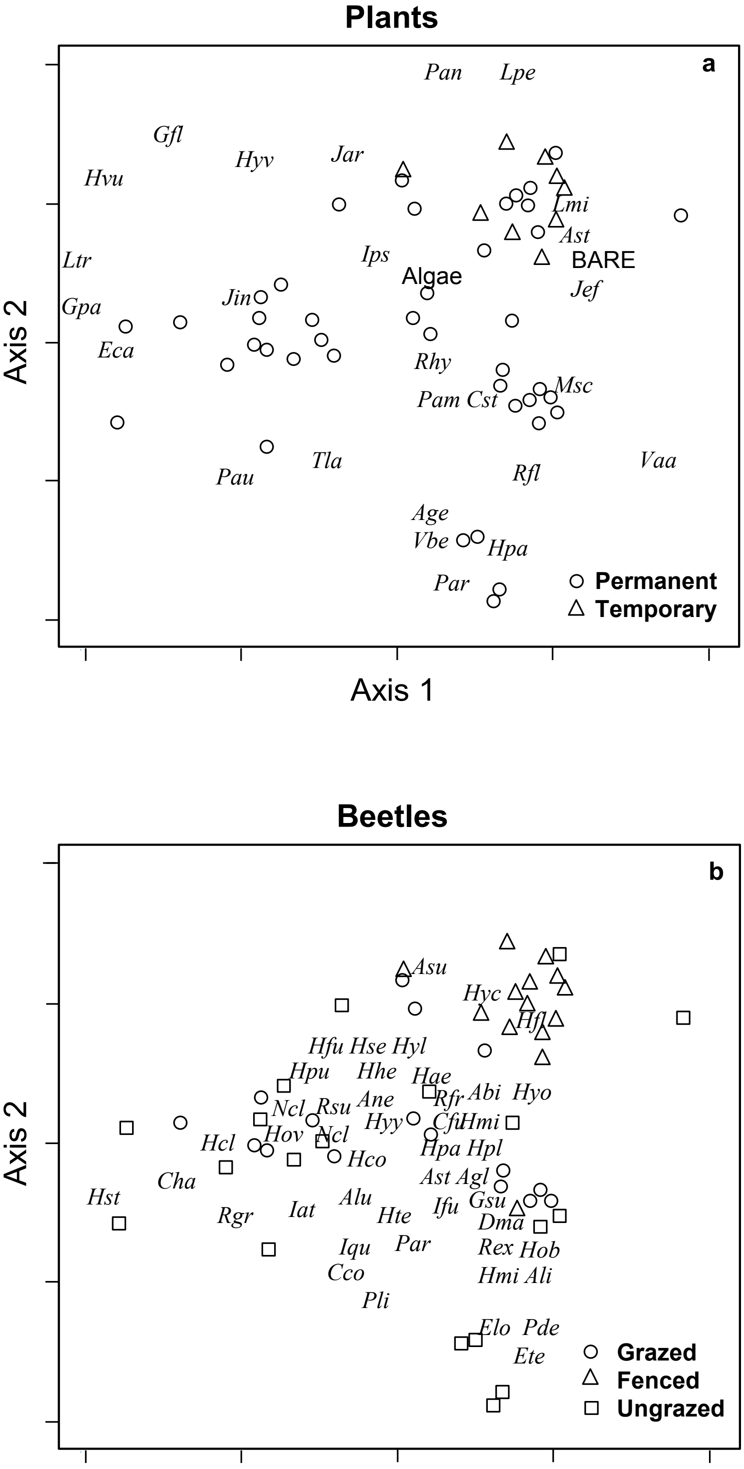

Figure 3.3.5. Predictive co-correspondence analysis (Co-CA) biplot of plant and beetle species composition. In each plot, species are positioned according to their loadings with respect to normalized pond scores derived from the plant composition data. The ponds were assigned a symbol based on their permanency (a, plant plot) and grazing intensity (b, beetle plot). The axes were rescaled to the same ranges so that sites (ponds) occupy the same position in both plots (Gioria *et al.*, 2010a). The full names of plant and beetle species are given in Appendix IV.

3.3.3 Cross-taxa Explanatory Relationships

The results of multivariate analyses are presented in Table 3.3.2. The Mantel test showed a moderate correlation between plant and water beetle species composition ($r = 0.302$, $P = 0.001$), a value that is similar to that obtained using the permutational version based on the Spearman rank correlation ($\rho = 0.337$; $P < 0.05$). A similar degree of congruence was also found using symmetric Co-CA, with the first two plant-derived (significant) axes representing approximately 28% of the variance in beetle species composition. A stronger relationship in species composition was, conversely, found using the Procrustean approach based on the first three axes of NMDS ordinations ($m^2 = 0.4253$, $P = 0.001$, Fig. 3.3.6). In general, stronger relationships were identified when the correlations among data matrices were calculated separately for each region, although inconsistencies were identified (Table 3.3.2).

The first three axes from the PCA of environmental variables summarized 71.4% of the total variance. Procrustes analysis showed that patterns in plant and beetle species composition were strongly correlated to the main environmental gradients summarized by the first three axes resulting from PCA ($m^2_{\text{NMDSbeetle-PCA}} = 0.6467$, $P < 0.001$; $m^2_{\text{NMDSplant-PCA}} = 0.4941$, $P < 0.001$, Table 3.3.2). A strong correlation between the results of CCA analyses for plant and beetle communities (first two significant axes), summarizing the environmental patterns driving the composition of plant and beetle communities, was also found ($m^2_{\text{CCAplant-CCAbeetle}} = 0.8552$, $P < 0.001$; Table 3.3.2). This was indicative of a similar response of both communities to the same set of environmental variables.

Table 3.3.2. Results of multivariate analyses aimed at evaluating cross-taxon congruence in species composition between wetland plants and water beetles in 54 farmland ponds in Ireland from two regions (R1 = Region 1; R2 = Region 2; NMDS = non-multidimensional scaling; PCA = principal components analysis; CCA = canonical correspondence analysis) (from Gioria *et al.*, 2010a, 2011).

Data matrices	Matrix dimension	Measure of correlation	P
Parametric Mantel test		*r*	
p-Mantel test R1+R2	(54 x 66) *vs* (54 x 76)	0.302	0.001
p-Mantel test R1	(25 x 63) *vs* (25 x 61)	0.212	0.001
p-Mantel test R2	(29 x 54) *vs* (29 x 41)	0.426	0.001
Non-parametric Mantel test		*ρ*	
n-p-Mantel test R1+R2	(54 x 66) *vs* (54 x 76)	0.348	0.05
n-p-Mantel test R1	(25 x 63) *vs* (25 x 61)	0.396	0.05
n-p-Mantel test R2	(29 x 54) *vs* (29 x 41)	0.432	0.05
Symmetric Co-CA		*λ*s	
Co-CA symmetric R1+R2	(54 x 66) *vs* (54 x 76)	0.281	*
Co-CA symmetric R1	(25 x 63) *vs* (25 x 61)	0.160	*
Co-CA symmetric R2	(29 x 54) *vs* (29 x 41)	0.318	*
Predictive Co-CA		Cross-validatory fit %	
Co-CA predictive R1+R2	(54 x 66) *vs* (54 x 76)	0.053	*
Co-CA predictive R1	(25 x 63) *vs* (25 x 61)	0.195	*
Co-CA predictive R2	(29 x 54) *vs* (29 x 41)	0.171	*
PROTEST cross-taxon congruence		m^2	
NMDSplant *vs* NMDSbeetle R1+R2	(54 x 3) *vs* (54 x 3)	0.425	0.001
NMDSplant R1 *vs* NMDSbeetle R1	(25 x 3) *vs* (25 x 3)	0.531	0.002
NMDSplant R2 *vs* NMDSbeetle R2	(29 x 3) *vs* (29 x 3)	0.625	0.001
PROTEST correlation biotic-abiotic data		m^2	
NMDSbeetle *vs* PCA	(54 x 3) *vs* (54 x 3)	0.647	0.001
NMDSplant *vs* PCA	(54 x 3) *vs* (54 x 3)	0.494	0.001
CCAplant *vs* CCAbeetle	(54 x 3) *vs* (54 x 3)	0.855	0.001

[*] first two significant axes

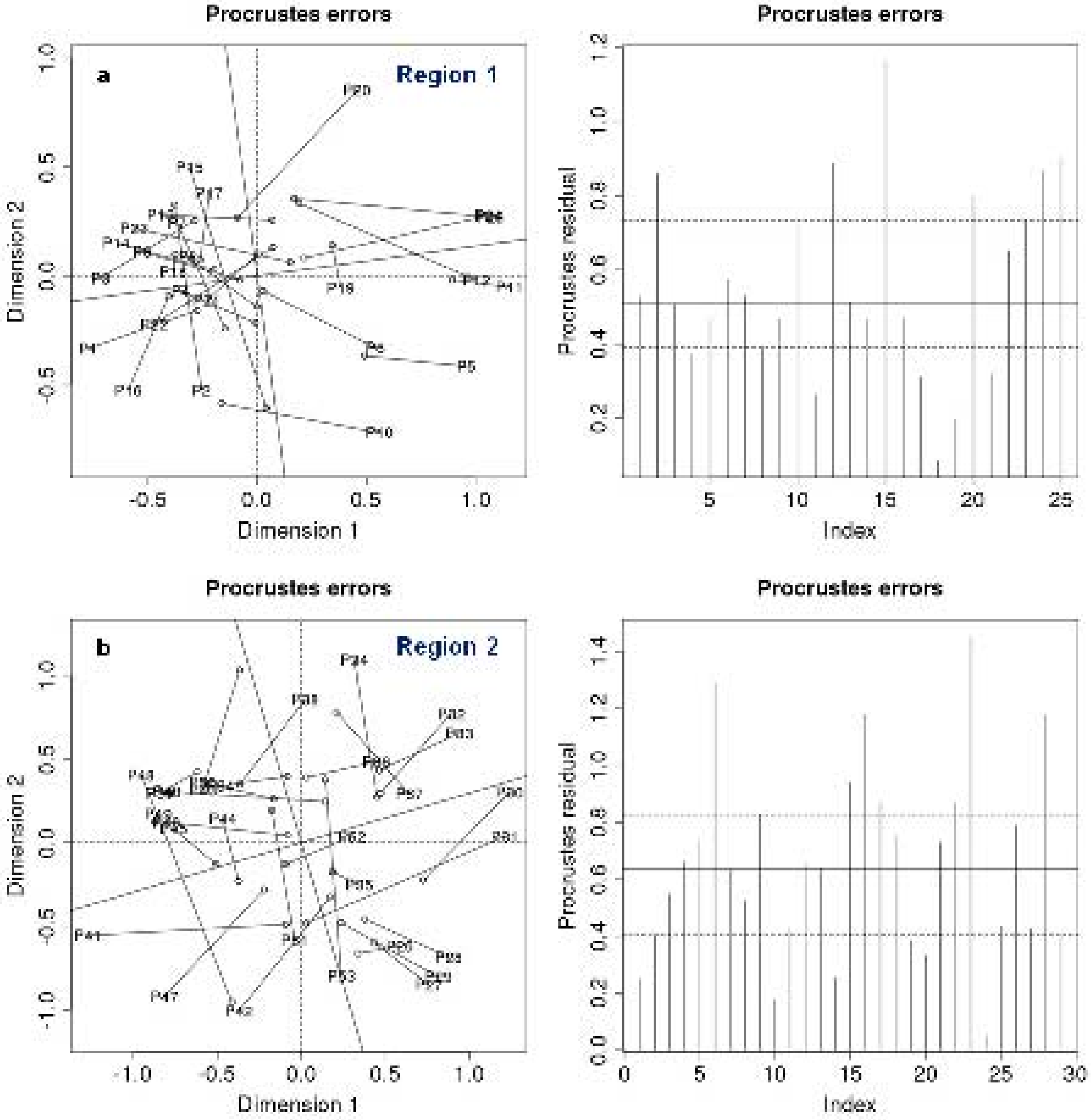

Figure 3.3.6. Procrustean configurations and Procrustean errors based on wetland plant and water beetle species composition data summarized by the first three axes of non-multidimensional scaling ordinations (100 iterations, Bray–Curtis dissimilarity), separately for (a) Region 1 (Wexford) and (b) Region 2 (Mullingar).

3.4 Discussion

3.4.1 Identifying Surrogate Taxa for Biodiversity

The use of surrogate taxa in conservation planning ultimately depends upon the strength of the congruence between ‘predictive’ and ‘predicted’ taxonomic groups and on their response to environmental conditions (Prendergast *et al*., 1993; Su *et al*., 2004; Gioria *et al*., 2011). A number of issues are, however, associated with the analysis of cross-taxa relationships, reflecting the complexity of biodiversity (multispecies) data, which are typically highly heterogeneous in space and time (Clarke, 1993; Anderson, 2001). The mechanisms underlying patterns of congruence, such as trophic relationships, competition and facilitation, are rather difficult to quantify (e.g. Gaston and Williams, 1996; Gaston, 2000), particularly where only species occurrence data are available, further complicating the search for consistent patterns of species distribution and relationships between different taxonomic groups.

The use of multiple statistical approaches indicated the existence of a consistent relationship between these taxonomic groups in Irish farmland ponds. Such a relationship is a function of species-specific associations, the role of plants in providing a range of functions to invertebrate communities, the response of plants and beetles to local environmental conditions, as well as regional environmental differences (Gioria *et al*., 2011). In general, species-poor plant assemblages supported species-poor beetle assemblages. Grazed and temporary ponds, which were characterized by the presence of only a few plant species – predominantly grasses and rushes (*Juncus* species) – supported species-poor beetle assemblages dominated by herbivores (*Helophorus* species) and algivores (*Haliplus* species; see Gioria *et al*., 2010a for a list of species). Conversely, ponds characterized by species-rich plant assemblages also supported a large number of beetle species.

The overall correlation between plant and beetle species richness was, however, only moderate. This reflects the inability of this measure of diversity to provide any information on the identity of the species recorded. For instance, certain species-poor plant communities, such as swamps dominated by the bulrush *Typha latifolia* L. or the bur-reed *Sparganium erectum* L., supported species-rich beetle assemblages. Thus, whilst plant species richness is easy to understand and useful for identifying highly degraded ponds or ponds of high conservation value, it should not be used as an indicator of water beetle species richness unless information at the plant community type or on the identity of the dominant plant species is accounted for. A study on the relationship between multiple taxonomic groups in ponds, aimed at identifying potential surrogates for biodiversity, should first be based on considerations on the composition of the vegetation recorded in a pond. The analysis of patterns in species richness is, however, useful for a rapid identification of 1) potentially degraded sites or sites at risk of biodiversity loss; 2) priority conservation areas; and 3) the presence of species that may play a major role in determining biodiversity patterns for other groups.

Species composition (abundance) was a better predictor of cross-taxa relationships and biodiversity patterns than species richness. This is because composition data incorporates information on the functional role and on the abundance of a species in a community. This is particularly important for vegetation, due to its role as a primary producer and as a provider of a physical structure to invertebrates, including oviposition site and shelter from predators (Painter, 1999). Water beetles include primary and secondary consumers. Only few beetle species are known to be associated with particular plant species or plant community types (e.g. Foster *et al*., 2009; Gioria *et al*., 2010a). The moderate congruence in species composition between plants and beetles probably reflects the characteristics of the habitat provided by the vegetation. The strength of the relationship in species composition was generally higher when the analysis of community data was conducted separately for each sampling region. This is because some species were unique to one region and the heterogeneity of the environmental conditions was higher when the data for the two regions were combined – most likely reflecting regional differences in land-use history, soil characteristics and climatic conditions.

3.4.2 Challenges in Identifying Surrogate Taxa

This study showed that multiple approaches and response variables should be used to evaluate relationships across multiple taxonomic groups and to identify surrogate taxa for biodiversity. A comprehensive evaluation of cross-taxon congruence should be based on the evaluation of both patterns in species richness and composition, as previously suggested (Su *et al*., 2004; Warman *et al*., 2004). Preliminary assessments of the potential ecological and conservation value of a pond based on correlations in species richness should be followed by evaluations of congruence in species composition providing information on species-specific associations. When faced with the choice of selecting the most appropriate analytical approach to calculate relationships in species composition, the possibility of evaluating the degree of association for each dimension (species) in the data matrices between each taxonomic group should be accounted for (Gioria *et al*., 2011). Such a feature, in fact, aids in the interpretation of the structure of the data and in the identification of important sources of variation, particularly when accompanied by the possibility of visualizing such patterns, such as when using Procrustean analysis and Co-CA.

Methods that account for information on the response of predictive and predicted taxa to the same set of environmental conditions should be employed, since stronger relationships are likely to be detected when the noise generated by the heterogeneity of local and regional environmental conditions is reduced. This can be achieved by calculating the correlation between the results of ordination techniques aimed at identifying the main environmental gradients or factors underlying community patterns for each study taxon (see Peres-Neto and Jackson, 2001).

An analytical framework designed to rigorously quantify cross-taxon relationships using multiple methods and multiple measures of community structure is central to improving our understanding of biodiversity patterns. Such a framework should encompass a series of steps to deal with the pitfalls associated with the analysis of community data. The use of a common framework would at least partially address the issues associated with the dependence of the strength of cross-taxa relationships upon the characteristics of the analytical approaches used in different studies and upon the nature of the input variables.

The use of a common analytical framework is important for improving our ability to make predictions about biodiversity patterns in ponds regionally and, potentially, globally, by allowing for a direct comparison of the results of multiple studies, across different spatial scales. Such a framework could also be used to evaluate changes in biodiversity over time as well as the effects of climate on species distribution.

A number of invasive alien species were recorded and locally abundant, such as the water fern *Azolla filiculoides*, the floating pennywort *Hydrocotyle ranunculoides*, the pigmyweed *Crassula helmsii* and the pondweeds *Elodea canadensis*, and *Myriophyllum aquaticum*. These species are listed among established threats and the ‘most unwanted’ species in Ireland and their rapid spread of these species in Irish rivers and lakes has been recently highlighted (Maguire *et al.,* 2011). This suggests that ponds may act as a source of seeds or plant material for invasive alien species. The role of ponds in the spread of alien species should be further investigated. As for the impact of invasive weeds on beetle diversity, species-rich beetle assemblages were recorded in alien or invasive vegetation, suggesting that the ecological functions provided by invasive plant species are similar to those of native species. Experimental work is required to evaluate whether the effect of invasive plant species on water beetles in negative, positive, or neutral. Only the water fern *Azolla filiculoides* had a clear detrimental impact of water beetles.

4 Conclusions

This study generated findings that are central to the conservation of freshwater biodiversity in Ireland, showing that ponds play a central role in maintaining freshwater biodiversity. A random sample of only over 50 ponds in two relatively small study areas located in Co. Wexford and Co. Westmeath supported over 30% of the Irish water beetle fauna and included uncommon beetle and plant species. These values are particularly significant when accounting for a number of local and regional factors. In this study, the majority of ponds were characterized by high nitrates and phosphates, and two ponds were hypertrophic. The study ponds were rather uniform in their chemical composition, with all ponds being either circum-neutral or slightly alkaline at the time of sampling. The duration of the hydroperiod (i.e., whether ponds possess a permanent or a temporary nature over a year), naturally affects the level of nutrients and the plant and animal communities. This means that even if the study ponds had not been impaired, species that have a requirement for more acid or lotic conditions would have been missed anyway.

The presence of IUCN Red List water beetle species clearly indicates farmland ponds play an important role in maintaining freshwater biodiversity not only regionally but also at the national level. The presence of 18 beetle species that have some form of Red List status in other European countries (Foster *et al*., 2009) indicates that farmland ponds may be a valuable habitat for the conservation of biodiversity even at a more global scale (Gioria *et al*., 2010a).

A number of environmental variables affected plant and beetle assemblages significantly. The duration of the hydroperiod is a major factor affecting the biodiversity of farmland ponds. Permanent ponds supported more species as well as more diverse plant and beetle assemblages compared to temporary ones, probably due to a lower habitat complexity and high level of disturbance in temporary ponds. Although temporary ponds did not support any unique plant or beetle species and overall species richness was low, these ecosystems could play an important role in the maintenance of biodiversity in the agricultural matrix by increasing pond connectivity, as previously reported in other landscape types and for different geographic regions (e.g. Nicolet *et al*., 2004; Della Bella *et al*., 2005). The potentially important contribution of temporary ponds to regional biodiversity was indicated by the high number of beetle species that were recorded in two newly created ponds, whose conditions were similar to those of temporary ponds. This suggests that where high habitat connectivity and landscape structure are present, newly created or temporary ponds have a potential to be rapidly colonized by active flyers such as dytiscid species (see Gee *et al*., 1997). The presence of the dytiscid *Acilius sulcatus* (L.), a species that has been found to sense the presence of fish by using chemical cues (Åbjörnsson *et al*., 1997) in these ponds suggests that these habitat types may be central to supporting species that avoid predators restricted to permanent waters. The presence of a swamp zone within permanent ponds where stands of the bulrush *Typha latifolia* and the bur-reed (*Sparganium erectum*) were dominant was consistently associated with species-rich beetle assemblages. *Sparganium erectum* was particularly important for supporting leaf beetles of the genus *Donacia* species, which tend to be uncommon and for which more information is required at the national level (Cox, 2007; Nelson *et al*., 2007; Foster *et al*., 2009).

Grazing had a detrimental effect on the ecological quality and biodiversity value of ponds. The direct accessibility of cattle to ponds was typically associated with high nutrient levels and low plant and beetle species richness. Grazing suppressed the growth of marginal and emergent plants, leading to plant assemblages dominated by species such as *Lolium perenne* and/or *Juncus effusus*, and resulting in species-poor beetle assemblages dominated by *Helophorus* species and dytiscids that have shown some tolerance to eutrophic conditions, such as *Hyphydrus ovatus* (L.) (see, e.g., Foster *et al*., 1992; Foster and Eyre, 1992). The detrimental effect of grazing on the ecological quality of ponds was confirmed by the fact that fenced ponds supported richer and more diverse beetle assemblages compared to grazed ponds. However, the positive effect of a fence system was dependent upon the distance of the fence from the pond margin and, thus, upon the presence and abundance of marginal and emergent vegetation.

In addition to providing information on the plant and beetle assemblages of farmland ponds and on their ecological quality and potential conservation value, this study provides information that is crucial for making predictions on the drivers of ponds biodiversity. This is in fact the first study to rigorously characterize the contribution of information on the vegetation in predicting community patterns for water beetles in ponds. One of the major implications of these findings is that wetland plants represent a suitable surrogate group for pond biodiversity in rapid pond biodiversity assessments. The species composition of wetland plants in farmland ponds in the study displayed a positive predictive capacity, being consistently correlated to that of water beetles, which was similar to that of environmental variables.

Trophic relationships and biotic interactions played an important role in determining the degree of correlation between plant and beetle assemblages, as indicated by the higher value of plant species composition compared to that of plant community type composition in predicting water beetle species composition. The loss of predictive capacity resulting from grouping information on plant species composition into information at the level of plant community type indicates that data at the species level are important for explaining and predicting the presence and abundance of host-specific beetle genera or species (Gioria *et al*., 2010a; 2011), such as that of leaf beetles of the genus *Donacia*, but not for the majority of other species.

Environmental variables showed a higher capacity in predicting patterns in water beetle species composition compared to that of plant species composition data, although the differences were not significant. This suggests that some information on the physic-chemical conditions at a pond and on the management regime is important to make considerations on the biodiversity of a pond. Regional differences were among the most important determinants of patterns in species composition of plant and beetle assemblages (Gioria *et al*., 2010a). This reflects the importance of the factor ‘region’ in encompassing and synthesising information such as climatic conditions, topography and geology, degree of habitat patchiness, as well as processes of dispersal and speciation (see Gioria *et al*., 2010a and references therein).

Maximum pond depth and the nature of the dominant substratum were also major determinants of the composition of both plant and beetle assemblages (Gioria *et al*., 2010b). The effect of pond depth on the vegetation was expected, since a certain pond depth is a requirement for submerged or floating-leaved plant species, such as *Myriophyllum* spp., *Elodea canadensis*, or *Potamogeton* species (Rodwell, 1995), which are therefore absent in temporary ponds. Differences in the dominant pond substratum also affected plant and beetle assemblages, with the majority of beetle species of higher conservation value (see Foster *et al*., 1992) being found in ponds characterized by a gravel substratum (Gioria *et al*., 2010a). The dominance of a clay or mud substratum inevitably results in increases in turbidity and available nutrients in case of disturbance events such as trampling by cattle or episodes of heavy rain in shallow waters. Gravel ponds, conversely, are characterized by a more stable substratum that supports richer beetle assemblages as well as uncommon species. This is an important result that must be accounted for when developing guidelines for the creation of ponds specifically targeted at maintaining or increasing biodiversity.

Plant and beetle assemblages responded in a similar way to the same set of environmental conditions and management practices. This important finding represents an additional argument in favour of the use of wetland plants as a surrogate group to be used to make predictions on the composition of water beetle assemblages in farmland ponds and for identifying priority conservation ponds. An important finding for the development of biodiversity conservation strategies is that pond surface area was correlated weakly with plant and beetle species richness in permanent ponds. This indicates that even the creation of small temporary ponds could play an important role in maintaining local biodiversity, also contributing to an increase in habitat connectivity.

Despite a higher contribution of environmental variables in predicting the composition of water beetle assemblages, the difference between models using information on plant species composition only and on environmental variables only in predicting biodiversity patterns for water beetles was not statistically significant. This is a major finding with important practical implications, indicating that even in the absence of any information on the environmental conditions at a pond, such as water chemistry, rigorous predictions on the composition of pond water beetle assemblages based on information on the vegetation only could be made, which is relatively easier to collect and is less time consuming and labour intensive.

Contrary to species-composition data, information on plant species richness and/or evenness, summarized by a univariate index for each pond, was a poor predictor of water beetle community patterns (Gioria *et al*., 2010a, 2011). The non-suitability of plant species richness or other univariate indices of diversity in predicting patterns in water beetle assemblages is consistent with previous findings (e.g. Su *et al*., 2004; Bilton *et al*., 2006; Schaffers *et al*., 2008) and in setting targets in conservation planning (Fleishman *et al*., 2006). While univariate measures such as species richness may provide a measure of the overall biodiversity value of a system (e.g. Gaston, 2000), they do not contain any information on species identity (Su *et al*., 2004; Gioria and Osborne, 2009; see Gioria et al., 2010a and references therein). This was evident in a number of ponds, where the relatively species-poor *Typha latifolia* community supported a large number of beetle species. Also, small ponds, characterized by one or two plant species or community types, also supported diverse beetle assemblages as well as uncommon species. However, information on plant species richness can only still be valuable for identifying highly degraded ponds, where only a few plant and beetle species are found, or ponds of potentially high conservation value, where both groups are characterized by a large number of species.

Since data on plant species composition are good predictors of composition patterns for water beetles in ponds, they can be used to identify ponds of potentially high conservation value and represent a cost-effective way of making preliminary pond biodiversity assessments. This would allow the collection of large-scale data within a relatively short period of time compared to that required to collect and identify water beetles or other invertebrates, due the fact that plants are in general easier to identify (Gioria *et al*., 2010a). The similar accuracy of environmental variables to predict water beetle species composition, combined with the similarity in the response of plants and beetles to the same environmental variables, indicates that information on plant species composition can be used as an effective indicator of disturbance in ponds (see also Foster *et al*., 1990; 1992; Menetrey *et al*., 2005).

The identification of the factors that are more important in determining patterns for both plant and beetle assemblages, such as permanency and grazing, pond depth and substratum, is critical to the development of conservation strategies aimed at maximizing the biodiversity value of farmland ponds. The significance of plant species composition in determining patterns in water beetle assemblages indicates that programmes aimed at conserving freshwater biodiversity will benefit from considering the vegetation first, as previously recommended for the conservation of arthropods in terrestrial systems (Schaffers *et al*., 2008; see Gioria *et al*., 2010a).

Ultimately, this study provides essential information for the production of documentation that is required due to Ireland's obligations to conserve freshwater biodiversity deriving from national and international agreements. These include the Convention on Biological Diversity (CBD 1992), the Ramsar Convention on Wetlands (1971), the Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora (EC Habitats Directive, CEC, 1992), and the Water Framework Directive (Directive 2000/60/EC; WFD, 2000).

This information is critical to the development of conservation programmes and of cost-effective biodiversity surveys. It is also central to predicting changes in biodiversity and the ecological quality of freshwater ecosystems associated with changes in land use and management practices.

5 Recommendations

This study highlighted the importance of ponds in the Irish agricultural landscape in maintaining freshwater biodiversity at the local, regional, and national level. The conservation of freshwater biodiversity requires a better understanding of the distribution of ponds and the diversity they support, as well as the development of management strategies based on a solid understanding of the contribution of environmental conditions to pond biodiversity. This requires: 1) the development of a mapping system of ponds, to monitor changes in the distribution of ponds associated with agricultural activities, other anthropogenic factors, as well as climate change and the distribution of invasive alien species; 2) a classification system of Irish ponds based on their ecological quality and the biodiversity they support, to monitor changes in freshwater biodiversity associated with the above-mentioned environmental changes; 3) a database of pristine and near-pristine ponds to identify potential high-quality reservoirs of biodiversity.

The conservation of biodiversity requires the collection of large datasets on the distribution of species and ecological quality of freshwater system. This requires the use of new approaches to identify surrogate taxa to be used in conservation planning and monitoring. It was shown that recently developed ordination techniques proved to be effective in assessing relationships between multiple taxonomic groups in freshwater systems, allowing a direct comparison of the strength of community data and abiotic conditions in predicting community patterns for other taxa. To collect large-scale information on pond biodiversity and their potential conservation value or degradations status, it is recommended that the vegetation of ponds is surveyed, possibly at the species level, although information on the composition of plant community type can provide some information useful for identifying conservation priority ponds or highly degraded ponds.

The use of common sampling protocols and analytical procedures to sample the biotic and abiotic conditions in Irish ponds should be encouraged. This is particularly important for ponds, since these are highly heterogeneous systems and the use of standardized sampling and analytical protocols would allow a reduction in the variability intrinsic in this kind of data and the identification of common drivers of biodiversity patterns and the comparison of the results of multiple studies.

This information is critical in the identification of the principal causal factors of community patterns for different taxa and for identifying surrogate groups. A wider use of predictive ordination techniques is recommended; this would allow the identification of surrogate taxa and a comparison of the role of vascular plants in predicting invertebrate assemblages in other aquatic and terrestrial systems (e.g. Schaffers *et al.*, 2008). This study showed that a sound ecological interpretation of biodiversity patterns in ponds should not depend solely on the value of a single statistic. Conversely, multiple approaches should be used to evaluate cross-taxa relationships, due to their dependence on biotic relationships and trophic interactions, and on the response of each taxon to a range of local and regional environmental conditions.

Finally, sampling designs and analytical approaches that allow for a synthesis of the results of a multiplicity of studies should be used. This would be best achieved by progressively shifting from purely observational studies to experimental investigations and to the use of hypothesis testing analytical procedures.

A critical approach is required when attempting to evaluate and discuss biotic–abiotic relationships based on the results of theoretical models. Based on the results of this study, some general recommendations can be made. Both species richness and species abundance data should be used in models of biodiversity. At least two modeling approaches should be used for each response variable, to increase the likelihood of detecting the major contributors of biodiversity patterns and to account for the highly variable nature of species data. Non-parametric modeling approaches should be included in studies of biotic–abiotic relationships, since these methods are flexible, robust, and have long been shown to be more appropriate for the analysis of community data, not being based on any assumption of multivariate normality (see Clarke, 1993). Moreover, the use of non-parametric methods would not require a strong data transformation, such as that applied in this study for comparative reasons, precluding the need to reduce the contribution of species abundance data in the models to information similar to that provided by presence/absence data (see Gioria *et al*., 2010b; 2011).

As the field of pond ecology rapidly increases, there is a growing need to adopt common analytical approaches that allow a synthesis of the results of multiple studies. This could be best achieved by progressively combining information from purely observational investigations with results obtained from experimental studies. In the long term, the use of balanced experimental designs and hypothesis-testing procedures would allow a more rigorous quantification of the effects of categorical variables and of the interactions among variables. It would also allow the conducting of meta-analysis studies and the drawing of more general conclusions about the effects of specific variables on communities and ecosystems (Gioria *et al*., 2010b). The use of a common analytical framework to analyse biodiversity patterns in ponds and to identify the main determinants of pond biodiversity would be particularly useful in an Irish context, as it would allow the comparison of the results of multiple research public and private research groups.

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Acronyms and Annotations

CBD	Convention on Biological Diversity
DISTNL	Distance-based multivariate analysis for a linear model
EPCN	European Pond Conservation Network
GLM	Generalized linear model
IUCN	International Union for Conservation of Nature
NBDC	National Biodiversity Data Centre
VIF	Variance inflation factor analysis
WFD	Water Framework Directive

Statistical Terms

CCA	Canonical correspondence analysis
CoCA	Co-correspondence analysis
DCA	Detrended correspondence analysis
NMDS	Non-parametric multidimensional scaling
PCA	Principal components analysis
PERMANOVA	Permutational multivariate analysis of variance
PLS	Partial least squares regression
PROTEST	Procrustean randomization test
SIMPER	Similarities percentage analysis

Appendix I

Appendix Ia. Results of SIMPER analysis showing the contribution of each plant species in determining 90% of the average similarity percentage within ungrazed, fenced, and grazed ponds.

Species	Av. Abu.	Av. Sim.	Sim./SD	Contr. %	Cum. Contr. %
Ungrazed ponds					
Typha latifolia	3.13	6.29	0.74	26.27	26.3
Rorippa nasturtium-aquaticum	1.98	2.37	0.42	9.89	36.2
Juncus effusus	1.18	1.90	0.49	7.93	44.1
Lemna minor	1.05	1.41	0.40	5.89	50.0
Lolium perenne	1.32	1.33	0.38	5.54	55.5
Callitriche stagnalis	1.05	1.24	0.50	5.17	60.7
Agrostis stolonifera	0.64	0.92	0.40	3.83	64.5
Potamogeton natans	0.96	0.82	0.39	3.41	67.9
Elodea canadensis	1.05	0.70	0.33	2.92	70.9
Phalaris arudinacea	0.74	0.60	0.34	2.51	73.4
Juncus conglomeratus	0.61	0.51	0.40	2.13	75.5
Mentha aquatica	0.52	0.47	0.35	1.95	77.4
Alisma plantago-aquatica	0.51	0.42	0.36	1.74	79.2
Menyanthes trifoliata	0.87	0.39	0.21	1.64	80.8
Iris pseudacous	0.54	0.36	0.30	1.51	82.3
Equisetum fluviatile	0.63	0.36	0.26	1.48	83.8
Juncus inflexus	0.55	0.35	0.28	1.48	85.3
Lemna trisulca	0.60	0.35	0.29	1.48	86.8
Phragmites australis	0.55	0.33	0.25	1.38	88.2
Apium nodiflorum	0.79	0.33	0.22	1.37	89.5
Nuphar lutaea	0.68	0.31	0.21	1.30	90.8
Fenced ponds					
Callitriche stagnalis	2.02	3.62	0.63	16.19	16.2
Lolium perenne	2.05	3.15	0.50	14.10	30.3
Typha latifolia	2.28	2.73	0.52	12.20	42.5
Juncus effusus	1.41	2.62	0.58	11.71	54.2
Iris pseudacorus	1.45	1.46	0.44	6.53	60.7
Potagemon natans	1.18	0.94	0.36	4.20	64.9
Algae	1.10	0.89	0.22	3.99	68.9
Juncus conglomeratus	0.76	0.64	0.37	2.84	71.8
Phalaris arudinacea	0.79	0.55	0.34	2.45	74.2
Juncus inflexus	0.54	0.54	0.37	2.41	76.6
Equisetum fluviatile	0.60	0.51	0.30	2.27	78.9
Agrostis stolonifera	0.54	0.48	0.37	2.13	81.0
Rorippa nasturtium-aquaticum	0.93	0.47	0.22	2.10	83.1
Glyceria fluitans	0.64	0.45	0.27	2.03	85.1
Sparganium erectum	0.54	0.41	0.27	1.82	87.0
Mentha aquatica	0.46	0.39	0.22	1.73	88.7
Lemna minor	0.56	0.36	0.16	1.60	90.3

Species	Av. Abu.	Av. Sim.	Sim./SD	Contr. %	Cum. Contr. %
Grazed ponds					
Lolium perenne	5.04	16.98	1.00	54.16	54.2
Lemna minor	1.55	4.17	0.62	13.30	67.5
Juncus effusus	1.76	3.53	0.52	11.25	78.7
Agrostis stolonifera	0.93	1.71	0.42	5.45	84.2
Callitriche stagnalis	1.05	1.31	0.37	4.18	88.3
Juncus conglomeratus	0.59	0.57	0.27	1.81	90.2

Av. Abu. = Average Abundance; Av. Sim. = Average Similarity; Sim./SD = Similarity/Standard Deviation; Contr. % = contribution to average similarity percentage; Cum. Contr. % = Cumulative contribution to average similarity percentage

Appendix Ib. Results of SIMPER analysis showing the contribution of each plant species in determining 90% of the average similarity percentage within permanent and temporary ponds.

Species	Av. Abu.	Av. Sim.	Sim./SD	Contr. %	Cum. Contr. %
Permanent ponds					
Typha latifolia	2.64	4.16	0.60	18.14	18.1
Lolium perenne	1.81	2.60	0.44	11.31	29.5
Juncus effusus	1.31	2.41	0.52	10.50	40.0
Callitriche stagnalis	1.47	2.19	0.53	9.56	49.5
Rorippa nasturtium-aquaticum	1.34	1.24	0.30	5.41	54.9
Lemna minor	0.91	1.19	0.34	5.20	60.1
Potamogeton natans	0.99	0.85	0.35	3.69	63.8
Juncus conglomeratus	0.74	0.77	0.40	3.34	67.2
Algae	0.87	0.62	0.19	2.72	69.9
Iris pseudacorus	0.81	0.60	0.31	2.61	72.5
Menyanthes trifoliata	0.96	0.58	0.24	2.54	75.0
Agrostis stolonifera	0.52	0.57	0.34	2.47	77.5
Juncus inflexus	0.58	0.51	0.33	2.21	79.7
Phragmites australis	0.74	0.45	0.28	1.98	81.7
Equisetum fluviatile	0.61	0.44	0.28	1.92	83.6
Alisma plantago-aquatica	0.49	0.42	0.35	1.84	85.4
Elodea canadensis	0.70	0.41	0.28	1.81	87.2
Phalaris arudinacea	0.55	0.40	0.28	1.76	89.0
Sparganium erectum	0.69	0.34	0.23	1.50	90.5
Temporary ponds					
Lolium perenne	7.37	37.54	3.94	66.46	66.5
Agrostis stolonifera	1.73	6.38	1.00	11.29	77.8
Lemna minor	1.86	6.27	0.72	11.10	88.9
Juncus effusus	2.07	4.28	0.51	7.59	96.4

Av. Abu. = Average Abundance; Av. Sim. = Average Similarity; Sim./SD = Similarity/Standard Deviation; Contr. % = contribution to average similarity percentage; Cum. Contr. % = Cumulative contribution to average similarity percentage

Appendix Ic. Results of SIMPER analysis showing the contribution of each plant species in determining 90% of the average similarity percentage within ponds with either a mud/clay or a gravel substratum.

Species	Av. Abu.	Av. Sim.	Sim./SD	Contr. %	Cum. Contr. %
Mud/clay substratum					
Lolium perenne	3.05	6.59	0.58	27.50	27.5
Juncus effusus	1.52	3.11	0.53	12.99	40.5
Typha latifolia	2.06	2.70	0.44	11.24	51.7
Lemna minor	1.15	2.13	0.42	8.90	60.6
Callitriche stagnalis	1.38	1.96	0.47	8.19	68.8
Agrostis stolonifera	0.75	1.19	0.40	4.95	73.8
Potamogeton natans	0.87	0.64	0.29	2.66	76.4
Algae	0.87	0.62	0.19	2.60	79.0
Juncus conglomeratus	0.61	0.54	0.31	2.25	81.3
Iris pseudacorus	0.74	0.51	0.27	2.14	83.4
Rorippa nasturtium-aquaticum	0.76	0.48	0.20	2.02	85.4
Menyanthes trifoliata	0.85	0.48	0.22	1.98	87.4
Juncus inflexus	0.48	0.35	0.25	1.46	88.9
Alisma plantago-aquatica	0.42	0.29	0.27	1.22	90.1
Gravel substratum					
Rorippa nasturtium-aquaticum	3.32	5.19	0.57	14.57	14.6
Typha latifolia	3.33	4.74	0.72	13.30	27.9
Lemna trisulca	1.74	3.53	1.24	9.92	37.8
Equisetum fluviatile	1.79	3.51	1.03	9.85	47.6
Mentha aquatica	1.22	2.52	0.91	7.09	54.7
Phragmites australis	1.24	1.79	0.69	5.02	59.8
Sparganium erectum	1.64	1.68	0.50	4.71	64.5
Callitriche stagnalis	1.05	1.62	0.67	4.55	69.0
Phalaris arundinacea	1.12	1.34	0.51	3.77	72.8
Hypericum elodes	0.78	1.08	0.70	3.05	75.8
Alopecurus geniculatus	0.63	1.03	0.72	2.89	78.7
Juncus effusus	0.83	0.99	0.72	2.79	81.5
Juncus inflexus	0.95	0.99	0.72	2.79	84.3
Juncus conglomeratus	0.87	0.99	0.71	2.77	87.1
Ranunculus flammula	0.52	0.45	0.32	1.27	88.3
Elodea canadensis	0.85	0.44	0.33	1.24	89.6
Potamogeton natans	0.68	0.38	0.31	1.06	90.6
Iris pseudacorus	0.68	0.37	0.31	1.04	91.7

Av. Abu. = Average Abundance; Av. Sim. = Average Similarity; Sim./SD = Similarity/Standard Deviation; Contr. % = contribution to average similarity percentage; Cum. Contr. % = Cumulative contribution to average similarity percentage

Appendix II

Appendix IIa. Results of SIMPER analysis showing the contribution of each beetle species in determining 90% of the average similarity percentage within ungrazed, fenced, and grazed ponds.

Species	Av. Abu.	Av. Sim.	Sim./SD	Contr. %	Cum. Contr. %
Ungrazed ponds					
Helophorus brevipalpis	8.64	5.80	1.43	17.19	17.2
Hygrotus inequalis	8.86	4.90	1.39	14.55	31.7
Hydroporus planus	5.23	4.08	1.57	12.10	43.8
Hydroporus palustris	5.05	3.69	1.27	10.95	54.8
Haliplus confinis	6.41	3.43	1.00	10.19	65.0
Anacaena globosus	4.41	1.57	0.63	4.66	69.6
Laccophilus minutus	4.18	1.28	0.50	3.78	73.4
Agabus nebulosus	2.45	1.04	0.69	3.08	76.5
Anacaena lutescens	1.95	0.93	0.71	2.76	79.3
Ilybius fuliginosus	2.86	0.79	0.45	2.35	81.6
Helophorus aequalis	1.91	0.62	0.43	1.83	83.4
Rhantus suturalis	3.32	0.60	0.35	1.79	85.2
Noterus clavicornis	4.55	0.54	0.22	1.59	86.8
Hydroporus tessellatus	1.23	0.49	0.50	1.46	88.3
Agabus bipustulatus	1.86	0.48	0.43	1.43	89.7
Helophorus minutus	1.82	0.46	0.39	1.38	91.1
Fenced ponds					
Helophorus brevipalpis	6.19	8.91	1.20	27.89	27.9
Hygrotus inequalis	7.00	6.66	1.35	20.86	48.8
Hydroporus planus	2.31	4.31	1.34	13.49	62.2
Hydroporus palustris	1.63	2.58	1.17	8.08	70.3
Agabus nebulosus	1.94	1.57	0.62	4.92	75.2
Agabus bipustulatus	1.69	1.06	0.32	3.33	78.6
Anacaena globosus	1.69	0.99	0.33	3.08	81.7
Haliplus confinis	1.31	0.92	0.47	2.88	84.5
Helophorus minutus	1.13	0.78	0.32	2.44	87.0
Helophorus grandis	1.56	0.74	0.32	2.33	89.3
Ilybius fuliginosus	1.88	0.70	0.28	2.20	91.5
Grazed ponds					
Helophorus brevipalpis	9.19	6.96	0.79	25.41	25.4
Hydroporus palustris	2.44	4.54	1.07	16.57	42.0
Hygrotus inequalis	3.94	4.16	0.87	15.18	57.2
Hydroporus planus	1.88	2.59	1.01	9.46	66.6
Agabus nebulosus	2.56	2.21	0.75	8.05	74.7
Noterus clavicornis	2.50	1.44	0.41	5.26	79.9
Haliplus confinis	1.44	1.06	0.33	3.85	83.8
Helophorus minutus	4.69	0.97	0.25	3.54	87.3
Rhantus suturalis	2.38	0.88	0.36	3.21	90.5

Av. Abu. = Average Abundance; Av. Sim. = Average Similarity; Sim./SD = Similarity/Standard Deviation; Contr. % = contribution to average similarity percentage; Cum. Contr. % = Cumulative contribution to average similarity percentage

Appendix IIb. Results of SIMPER analysis showing the contribution of each beetle species in determining 90% of the average similarity percentage within permanent and temporary ponds.

Species	Av. Abu.	Av. Sim.	Sim./SD	Contr. %	Cum. Contr. %
Permanent ponds					
Hygrotus inequalis	1.43	6.30	1.86	15.36	15.4
Helophorus brevipalpis	1.41	6.15	1.55	15.00	30.4
Hydroporus planus	1.24	5.63	1.77	13.73	44.1
Hydroporus palustris	1.19	5.18	1.59	12.64	56.7
Haliplus confinis	1.00	3.01	0.84	7.34	64.1
Agabus nebulosus	0.83	2.23	0.76	5.45	69.5
Anacaena globosus	0.78	1.80	0.60	4.40	73.9
Ilybius fuliginosus	0.61	0.89	0.45	2.17	76.1
Agabus bipustulatus	0.50	0.85	0.37	2.08	78.2
Noterus clavicornis	0.63	0.84	0.38	2.04	80.2
Anacaena lutescens	0.55	0.81	0.44	1.98	82.2
Helophorus aequalis	0.49	0.81	0.37	1.97	84.2
Helophorus minutes	0.50	0.72	0.37	1.75	85.9
Rhantus suturalis	0.48	0.53	0.31	1.29	87.2
Hyphydrus ovatus	0.40	0.52	0.35	1.27	88.5
Helophorus grandis	0.38	0.46	0.26	1.13	89.6
Hydroporus angustatus	0.38	0.40	0.33	0.97	90.6
Temporary ponds					
Helophorus brevipalpis	1.72	8.51	1.69	19.58	19.6
Hydroporus palustris	1.24	8.02	3.09	18.46	38.0
Hydroporus planus	1.13	5.68	1.67	13.09	51.1
Agabus nebulosus	1.1	5.1	1.05	11.75	62.9
Agabus bipustulatus	1.11	4.17	1.08	9.61	72.5
Hygrotus inequalis	1.06	3.42	0.8	7.87	80.4
Helophorus minutes	1.12	3.09	0.75	7.12	87.5
Rhantus suturalis	0.78	1.28	0.44	2.94	90.4

Av. Abu. = Average Abundance; Av. Sim. = Average Similarity; Sim./SD = Similarity/Standard Deviation; Contr. % = contribution to average similarity percentage; Cum. Contr. % = Cumulative contribution to average similarity percentage

Appendix IIc. Results of SIMPER analysis showing the contribution of each beetle species in determining 90% of the average similarity percentage within ponds with either a mud/clay or a gravel substratum.

Species	Av. Abu.	Av. Sim.	Sim./SD	Contr. %	Cum. Contr. %
Mud/clay substratum					
Helophorus brevipalpis	1.47	7.26	1.59	17.41	17.4
Hygrotus inequalis	1.34	6.48	1.62	15.54	32.9
Hydroporus planus	1.15	6.05	1.72	14.52	47.5
Hydroporus palustris	1.13	6.04	1.65	14.48	61.9
Agabus nebulosus	0.89	3.03	0.83	7.28	69.2
Haliplus confinis	0.78	2.21	0.63	5.30	74.5
Agabus bipustulatus	0.56	1.27	0.42	3.05	77.6
Anacaena globosus	0.54	1.22	0.42	2.94	80.5
Helophorus aequalis	0.54	1.06	0.42	2.54	83.0
Noterus clavicornis	0.61	0.96	0.40	2.29	85.3
Helophorus minutus	0.56	0.95	0.39	2.27	87.6
Rhantus suturalis	0.54	0.76	0.35	1.83	89.4
Ilybius fuliginosus	0.40	0.54	0.32	1.29	90.7
Gravel substratum					
Hydroporus planus	1.67	4.73	3.85	8.53	8.5
Hydroporus palustris	1.62	4.38	5.94	7.90	16.4
Anacaena globosus	1.58	3.94	4.18	7.11	23.5
Hygrotus decoratus	1.41	3.70	5.67	6.67	30.2
Helophorus brevipalpis	1.42	3.46	6.32	6.25	36.5
Hygrotus inequalis	1.53	3.31	1.66	5.98	42.4
Anacaena lutescens	1.22	2.89	1.59	5.21	47.6
Ilybius fuliginosus	1.34	2.78	1.46	5.01	52.7
Haliplus confinis	1.35	2.45	0.98	4.42	57.1
Hydroporus angustatus	0.94	2.20	1.58	3.97	61.0
Porhydrus lineatus	1.18	1.99	1.02	3.60	64.6
Hydroporus tessellatus	0.88	1.88	0.99	3.38	68.0
Hydroporus nigrita	0.75	1.80	1.02	3.25	71.3
Helophorus obscurus	0.75	1.62	1.01	2.93	74.2
Anacaena limbata	0.89	1.62	0.71	2.92	77.1
Helophorus minutus	0.85	1.34	0.71	2.41	79.5
Enochrus testaceus	0.67	1.32	0.72	2.38	81.9
Helophorus grandis	0.89	1.31	0.72	2.37	84.3
Laccophilus minutus	0.99	1.26	0.73	2.27	86.5
Agabus bipustulatus	0.82	1.17	0.72	2.12	88.7
Agabus nebulosus	0.77	0.97	0.72	1.76	90.4

Av. Abu. = Average Abundance; Av. Sim. = Average Similarity; Sim./SD = Similarity/Standard Deviation; Contr. % = contribution to average similarity percentage; Cum. Contr. % = Cumulative contribution to average similarity percentage

Appendix III

Appendix III. Environmental variables. Grazing (U = ungrazed; F = fenced; G = grazed); Permanency (Perm: P = permanent; T = temporary); Substratum (Sub: M = mud/clay; Gr = gravel); Region (R1 = P1–P25; R2 = P26–P54); Cond = Conductivity, expressed in microSiemens per centimetre at 25° Celsius.

Pond	Area	Max. depth	Max. sampling depth	pH	Cond	Alk	NH$_3$-N	NO$_3$-N	PO$_4$-P	Graz	Perm	Sub
	(m^2)	(m)	(m)			(mgL^{-1})	(µgL^{-1})	(µgL^{-1})	(µgL^{-1})			
P1	89	100	50	7.56	248	36	31	641	28	U	P	M
P2	198	200	50	7.19	474	46	46	397	31	U	P	Gv
P3	189	200	50	7.30	422	51	41	406	37	U	P	Gv
P4	123	60	60	7.47	547	80	97	1112	61	F	P	M
P5	77	30	30	7.12	450	120	180	1492	89	G	T	M
P6	99	50	50	7.34	564	26	24	423	47	U	P	M
P7	69	50	50	7.56	452	51	114	924	77	G	P	M
P8	65	55	55	7.00	508	49	11	1165	80	F	P	M
P9	64	70	70	7.81	364	39	60	408	66	G	P	M
P10	99	40	40	7.50	544	92	201	1321	99	G	P	M
P11	25	25	25	7.70	389	120	245	1491	82	G	T	M
P12	59	50	50	7.01	314	51	99	713	57	G	T	M
P13	89	70	70	7.12	340	62	192	408	66	U	P	M
P14	201	200	70	7.01	260	52	56	901	55	U	P	M
P15	89	80	50	7.22	253	91	258	1356	101	G	P	M
P16	128	70	50	7.45	683	111	301	1521	99	U	P	M
P17	301	200	50	7.65	559	52	109	913	61	U	P	M
P18	114	200	50	7.01	572	39	56	589	35	U	P	M
P19	59	60	60	7.13	231	45	45	623	41	F	P	M
P20	25	40	40	7.77	289	90	84	941	59	U	T	M
P21	99	100	50	7.25	378	81	102	1042	65	F	P	M
P22	108	100	50	7.01	243	65	89	920	55	F	P	M
P23	49	40	40	7.22	298	55	101	867	56	F	P	M
P24	69	25	25	7.35	417	135	289	1345	79	G	T	M
P25	189	25	25	7.54	453	129	278	1249	69	G	T	M
P26	106	50	50	7.33	755	41	39	556	16	U	P	Gv
P27	117	50	50	7.19	780	51	35	509	18	U	P	Gv
P28	154	50	50	7.30	758	48	33	488	16	U	P	Gv
P29	145	50	50	7.47	783	45	36	479	12	U	P	Gv
P30	39	30	30	6.88	446	51	41	601	41	U	P	Gv
P31	30	50	50	6.98	999	67	67	789	56	F	P	Gv
P32	78	30	30	7.99	909	81	81	1002	67	F	P	M
P33	67	30	30	8.00	1029	91	122	1309	71	G	P	M
P34	25	50	50	7.81	937	81	123	1201	78	F	P	M
P35	89	30	30	8.01	1003	95	131	1304	71	F	P	M
P36	61	30	30	8.08	896	99	144	1346	73	F	P	M

Pond	**Area**	**Max. depth**	**Max. sampling depth**	**pH**	**Cond**	**Alk**	**NH$_3$-N**	**NO$_3$-N**	**PO$_4$-P**	**Graz**	**Perm**	**Sub**
	(m^2)	**(m)**	**(m)**			**(mgL^{-1})**	**(μgL^{-1})**	**(μgL^{-1})**	**(μgL^{-1})**			
P37	49	40	40	6.83	759	45	35	501	31	U	P	M
P38	89	100	50	7.01	595	77	45	789	46	F	P	M
P39	108	30	30	7.45	756	91	81	1003	81	G	P	M
P40	99	30	30	7.66	830	86	78	1009	83	G	P	M
P41	89	50	50	9.05	105	41	28	405	22	U	P	M
P42	39	40	40	7.00	398	43	34	309	35	U	P	M
P43	89	25	25	7.16	876	101	98	1130	68	G	T	M
P44	41	30	30	7.13	810	111	123	1356	82	G	T	M
P45	29	25	25	7.24	746	121	167	1461	81	G	T	Gv
P46	98	35	35	7.00	457	78	98	761	45	U	P	M
P47	59	150	50	7.00	597	67	71	651	41	U	P	M
P48	28	35	35	7.02	308	55	51	451	32	U	P	M
P49	89	30	30	7.46	753	115	101	1108	81	G	P	M
P50	84	35	35	7.03	619	71	61	691	43	G	P	M
P51	44	40	40	7.32	114	81	72	864	56	F	P	M
P52	32	35	35	7.01	399	73	67	671	51	U	P	M
P53	49	30	30	7.46	279	74	71	751	54	U	P	M
P54	148	30	30	7.44	605	117	109	1221	78	G	P	M

Appendix IV

Appendix IV. Plant and beetle species abbreviations.

Plants					
Species	**Code**	**Species**	**Code**	**Species**	**Code**
Agrostis stolonifera	Ast	*Hydrocotyle ranunculoides*	Hra	*Phragmites australis*	Pau
Alisma plantago-aquatica	Apl	*Hypericum elodes*	Hel	*Polygonum amphibium*	Pam
Alopecurus geniculatus	Age	*Iris pseudacorus*	Ips	*Potamogeton natans*	Pna
Apium nodiflorum	Ano	*Juncus articulatus*	Jar	*Potamogeton obtusifolius*	Pob
Azolla filiculoides	Afi	*Juncus bufonius*	Jbu	*Potamogeton polygonifolius*	Ppo
Butomus umbellatus	Bum	*Juncus conglomeratus*	Jco	*Potamogeton gramineus*	Pgr
Callitriche platycarpa	Cpl	*Juncus effusus*	Jef	*Potentilla anserina*	Pan
Callitriche stagnalis	Cst	*Juncus inflexus*	Jin	*Potentilla palustris*	Ppa
Carex pendula	Cpe	*Lemna minor*	Lmi	*Ranunculus bulbosus*	Rbu
Carex riparia	Cri	*Lemna minuta*	Lmn	*Ranunculus flammula*	Rfl
Caltha palustris	Cpa	*Lemna trisulca*	Ltr	*Ranunculus hederacea*	Rhe
Chara denudate	Cde	*Lolium perenne*	Lpe	*Rorippa nasturtium-aquaticum*	Rna
Crassula helmsii	Che	*Mentha aquatica*	Maq	*Rumex hydrolapathum*	Rhy
Elatine hexandra	Eex	*Menyanthes trifoliata*	Mtr	*Scirpus lacustris*	Sla
Eleocharis palustris	Epa	*Myosotis scorpioides*	Msc	*Sparganium erectum*	Ser
Elodea Canadensis	Eca	*Myriophyllum aquaticum*	Maq	*Trifolium repens*	Tre
Equisetum fluviatile	Efl	*Myriophyllum spicatum*	Msp	*Typha latifolia*	Tla
Equisetum palustre	Epl	*Myriophyllum verticillatum*	Mve	*Utricularia vulgaris*	Uvu
Galium palustre	Gpa	*Nuphar lutea*	Nlu	*Valeriana dioica*	Vdi
Glyceria fluitans	Gfl	*Nymphoides peltata*	Npe	*Veronica anagallis-aquatica*	Vaa
Hottonia palustris	Hpa	*Persicaria hydropiper*	Phy	*Veronica beccabunga*	Vbe
Hippuris vulgaris	Hvu	*Polygonum persicaria*	Ppe		
Hydrocotyle vulgaris	Hyv	*Phalaris arundinacea*	Par		
Beetles					
Acilius sulcatus	Asu	*Haliplus fulvus*	Hfu	*Hygrobia hermanni*	Hhe
Agabus bipustulatus	Abi	*Haliplus immaculatus*	Him	*Hygrotus confluens*	Hyc
Agabus nebulosus	Ane	*Haliplus lineatocollis*	Hli	*Hygrotus impressopunctatus*	Hyi
Agabus sturmii	Ast	*Haliplus flavicollis*	Hfl	*Hygrotus inequalis*	Hyy
Anacaena globosus	Agl	*Haliplus ruficollis*	Hru	*Hygrotus quinquelinatus*	Hyq
Anacaena limbata	Ali	*Haliplus sibiricus*	Hsi	*Hygrotus versicolor*	Hyv
Anacaena lutescens	Alu	*Helophorus aequalis*	Hae	*Hyphydrus ovatus*	Hyo
Cercyon convexiusculus	Cco	*Helophorus brevipalpis*	Hbr	*Ilybius ater*	Iat
Colymbetes fuscus	Cfu	*Helophorus grandis*	Hgr	*Ilybius fuliginosus*	Ifu
Copelatus haemorroidalis	Cha	*Helophorus griseus*	Hgs	*Ilybius quadriguttatus*	Iqu
Donacia bicolora	Dbi	*Helophorus minutus*	Hmi	*Laccobius minutus*	Lmi
Donacia crassipes	Dcr	*Helophorus obscurus*	Hob	*Laccophilus minutus*	Lmn
Donacia marginata	Dmr	*Hydaticus seminiger*	Hse	*Lesteva longoelytrata*	Llo
Donacia simplex	Dsi	*Hydrobius fuscipes*	Hfp	*Limnebius truncatellus*	Ltr
Donacia versicolorea	Dve	*Hydroporus angustatus*	Han	*Noterus clavicornis*	Ncl
Dryops luridus	Dlu	*Hydroporus erytrocephalus*	Her	*Phaedon armoraciae*	Par

Beetles					
Species	**Code**	**Species**	**Code**	**Species**	**Code**
Dytiscus marginalis	Dma	*Hydroporus nigrita*	Hni	*Porhydrus lineatus*	Pli
Dytiscus semisulcatus	Dse	*Hydroporus palustris*	Hpa	*Prasocuris phellandrii*	Pph
Enochrus coartatus	Eco	*Hydroporus obsoletus*	Hyo	*Rhantus exoletus*	Rex
Enochrus melanocephalus	Eme	*Hydroporus planus*	Hpl	*Rhantus frontalis*	Rfr
Enochrus testaceus	Ete	*Hydroporus pubescens*	Hpu	*Rhantus grapii*	Rgr
Graptodytes pictus	Gpi	*Hydroporus striola*	Hst	*Rhantus suturalis*	Rsu
Gyrinus caspius	Gca	*Hydroporus tessellatus*	Hte	*Stictonectes lepidus*	Sle
Gyrinus substriatus	Gsu	*Hydroporus umbrosus*	Hum	*Suphrodites dorsalis*	Sdo
Haliplus confinis	Hco	*Hydrovatus clypealis*	Hcl		

An Ghníomhaireacht um Chaomhnú Comhshaoil

Is í an Gníomhaireacht um Chaomhnú Comhshaoil (EPA) comhlachta reachtúil a chosnaíonn an comhshaol do mhuintir na tíre go léir. Rialaímid agus déanaimid maoirsiú ar ghníomhaíochtaí a d'fhéadfadh truailliú a chruthú murach sin. Cinntímid go bhfuil eolas cruinn ann ar threochtaí comhshaoil ionas go nglactar aon chéim is gá. Is iad na príomhnithe a bhfuilimid gníomhach leo ná comhshaol na hÉireann a chosaint agus cinntiú go bhfuil forbairt inbhuanaithe.

Is comhlacht poiblí neamhspleách í an Ghníomhaireacht um Chaomhnú Comhshaoil (EPA) a bunaíodh i mí Iúil 1993 faoin Acht fán nGníomhaireacht um Chaomhnú Comhshaoil 1992. Ó thaobh an Rialtais, is í an Roinn Comhshaoil, Pobal agus Rialtais Áitiúil.

ÁR bhFREAGRACHTAÍ

CEADÚNÚ

Bíonn ceadúnais á n-eisiúint againn i gcomhair na nithe seo a leanas chun a chinntiú nach mbíonn astuithe uathu ag cur sláinte an phobail ná an comhshaol i mbaol:

- áiseanna dramhaíola (m.sh., líonadh talún, loisceoirí, stáisiúin aistrithe dramhaíola);
- gníomhaíochtaí tionsclaíocha ar scála mór (m.sh., déantúsaíocht cógaisíochta, déantúsaíocht stroighne, stáisiúin chumhachta);
- diantalmhaíocht;
- úsáid faoi shrian agus scaoileadh smachtaithe Orgánach Géinathraithe (GMO);
- mór-áiseanna stórais peitreail;
- scardadh dramhuisce.

FEIDHMIÚ COMHSHAOIL NÁISIÚNTA

- Stiúradh os cionn 2,000 iniúchadh agus cigireacht de áiseanna a fuair ceadúnas ón nGníomhaireacht gach bliain.
- Maoirsiú freagrachtaí cosanta comhshaoil údarás áitiúla thar sé earnáil - aer, fuaim, dramhaíl, dramhuisce agus caighdeán uisce.
- Obair le húdaráis áitiúla agus leis na Gardaí chun stop a chur le gníomhaíocht mhídhleathach dramhaíola trí comhordú a dhéanamh ar líonra forfheidhmithe náisiúnta, díriú isteach ar chiontóirí, stiúradh fiosrúcháin agus maoirsiú leigheas na bhfadhbanna.
- An dlí a chur orthu siúd a bhriseann dlí comhshaoil agus a dhéanann dochar don chomhshaol mar thoradh ar a ngníomhaíochtaí.

MONATÓIREACHT, ANAILÍS AGUS TUAIRISCIÚ AR AN GCOMHSHAOL

- Monatóireacht ar chaighdeán aeir agus caighdeáin aibhneacha, locha, uiscí taoide agus uiscí talaimh; leibhéil agus sruth aibhneacha a thomhas.
- Tuairisciú neamhspleách chun cabhrú le rialtais náisiúnta agus áitiúla cinntí a dhéanamh.

RIALÚ ASTUITHE GÁIS CEAPTHA TEASA NA HÉIREANN

- Cainníochtú astuithe gáis ceaptha teasa na hÉireann i gcomhthéacs ár dtiomantas Kyoto.
- Cur i bhfeidhm na Treorach um Thrádáil Astuithe, a bhfuil baint aige le hos cionn 100 cuideachta atá ina mór-ghineadóirí dé-ocsaíd charbóin in Éirinn.

TAIGHDE AGUS FORBAIRT COMHSHAOIL

- Taighde ar shaincheisteanna comhshaoil a chomhordú (cosúil le caighdéan aeir agus uisce, athrú aeráide, bithéagsúlacht, teicneolaíochtaí comhshaoil).

MEASÚNÚ STRAITÉISEACH COMHSHAOIL

- Ag déanamh measúnú ar thionchar phleananna agus chláracha ar chomhshaol na hÉireann (cosúil le pleananna bainistíochta dramhaíola agus forbartha).

PLEANÁIL, OIDEACHAS AGUS TREOIR CHOMHSHAOIL

- Treoir a thabhairt don phobal agus do thionscal ar cheisteanna comhshaoil éagsúla (m.sh., iarratais ar cheadúnais, seachaint dramhaíola agus rialacháin chomhshaoil).
- Eolas níos fearr ar an gcomhshaol a scaipeadh (trí cláracha teilifíse comhshaoil agus pacáistí acmhainne do bhunscoileanna agus do mheánscoileanna).

BAINISTÍOCHT DRAMHAÍOLA FHORGHNÍOMHACH

- Cur chun cinn seachaint agus laghdú dramhaíola trí chomhordú An Chláir Náisiúnta um Chosc Dramhaíola, lena n-áirítear cur i bhfeidhm na dTionscnamh Freagrachta Táirgeoirí.
- Cur i bhfeidhm Rialachán ar nós na treoracha maidir le Trealamh Leictreach agus Leictreonach Caite agus le Srianadh Substaintí Guaiseacha agus substaintí a dhéanann ídiú ar an gcrios ózóin.
- Plean Náisiúnta Bainistíochta um Dramhaíl Ghuaiseach a fhorbairt chun dramhaíl ghuaiseach a sheachaint agus a bhainistiú.

STRUCHTÚR NA GNÍOMHAIREACHTA

Bunaíodh an Ghníomhaireacht i 1993 chun comhshaol na hÉireann a chosaint. Tá an eagraíocht á bhainistiú ag Bord lánaimseartha, ar a bhfuil Príomhstiúrthóir agus ceithre Stiúrthóir.

Tá obair na Gníomhaireachta ar siúl trí ceithre Oifig:

- An Oifig Aeráide, Ceadúnaithe agus Úsáide Acmhainní
- An Oifig um Fhorfheidhmiúchán Comhshaoil
- An Oifig um Measúnacht Comhshaoil
- An Oifig Cumarsáide agus Seirbhísí Corparáide

Tá Coiste Comhairleach ag an nGníomhaireacht le cabhrú léi. Tá dáréag ball air agus tagann siad le chéile cúpla uair in aghaidh na bliana le plé a dhéanamh ar cheisteanna ar ábhar imní iad agus le comhairle a thabhairt don Bhord.

Science, Technology, Research and Innovation for the Environment (STRIVE) 2007-2013

The Science, Technology, Research and Innovation for the Environment (STRIVE) programme covers the period 2007 to 2013.

The programme comprises three key measures: Sustainable Development, Cleaner Production and Environmental Technologies, and A Healthy Environment; together with two supporting measures: EPA Environmental Research Centre (ERC) and Capacity & Capability Building. The seven principal thematic areas for the programme are Climate Change; Waste, Resource Management and Chemicals; Water Quality and the Aquatic Environment; Air Quality, Atmospheric Deposition and Noise; Impacts on Biodiversity; Soils and Land-use; and Socio-economic Considerations. In addition, other emerging issues will be addressed as the need arises.

The funding for the programme (approximately €100 million) comes from the Environmental Research Sub-Programme of the National Development Plan (NDP), the Inter-Departmental Committee for the Strategy for Science, Technology and Innovation (IDC-SSTI); and EPA core funding and co-funding by economic sectors.

The EPA has a statutory role to co-ordinate environmental research in Ireland and is organising and administering the STRIVE programme on behalf of the Department of the Environment, Heritage and Local Government.

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An Ghníomhaireacht um Chaomhnú Comhshaoil

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