Title	The significance of biodiversity in agriculture: relevance, aims and progress of the Ag-Biota Project
Authors(s)	Purvis, Gordon, Bolger, Tom, Breen, John, Connolly, John, Curry, Jim, Kelly-Quinn, Mary, Schmidt, Olaf, Schulte, Rogier, Whelan, John, et al.
Publication date	2005-01-01
Publication information	Purvis, Gordon, Tom Bolger, John Breen, John Connolly, Jim Curry, Mary Kelly-Quinn, Olaf Schmidt, Rogier Schulte, John Whelan, and et al. “The Significance of Biodiversity in Agriculture: Relevance, Aims and Progress of the Ag-Biota Project.” UCD, Faculty of Agriculture, January 1, 2005.
Publisher	UCD, Faculty of Agriculture
Item record/more information	http://hdl.handle.net/10197/10045

Tearmann: Irish journal of agri-environmental research, **4,** 29-50, 2005

The significance of biodiversity in agriculture: relevance, aims and progress of the Ag-Biota Project

*Gordon Purvis[1],Tom Bolger[1], John Breen[2], John Connolly[3], Jim Curry[1], John Finn[4], Mary Kelly-Quinn[1], Tom Kennedy[5], Olaf Schmidt[1], Rogier Schulte[1] and John Whelan[1]

[1]*School of Biology and Environmental Science, University College Dublin, Belfield, Dublin 4*
[2] *Department of Life Sciences, University of Limerick, Limerick*
[3] *School of Mathematical Sciences, University College Dublin, Belfield, Dublin 4*
[4] *Teagasc, Johnstown Castle Research Centre, Wexford*
[5] *Teagasc, Oak Park Research Centre, Carlow*

e-mail: gordon.purvis@ucd.ie

Abstract

We describe and review the scientific and policy background with respect to the impact of agriculture on biodiversity and outline the structure and objectives of the Ag-Biota Project. The latter is a large, multi-institutional study funded by the ERTDI Programme (2000–2006) under the aegis of the National Development Plan. As such, Ag-Biota represents an ongoing commitment to the protection and conservation of biodiversity, and the integration of policy towards the achievement of these goals in all economic sectors. Ag-Biota is addressing directly the practical needs for agri-environmental policy development, such as the need to identify suitable biodiversity indicators for agriculture and begin the development of realistic and practical monitoring and assessment methods; is focussing on the development of ecological understanding concerning the more effective utilisation of beneficial biological populations and processes within the agro-ecosystem; and is asking more fundamental ecological questions concerning the functional role and significance of biological diversity in community structures. The Ag-Biota project represents a suitably policy-focussed response to, and a considerable investment in, the needs of Irish biodiversity research within the context of modern agriculture. As such, we feel that the project is a good model for future biodiversity research, addressing the need for information and an appropriate knowledge base to support practical environmental protection measures.

Key Index Words: Monitoring and managing biodiversity, egroecology, farming practice, beneficial populations, bio-indicators, pollination, predation, soil fertility, plant production, functional redundancy.

Introduction

Agriculture is the single largest form of land use across most regions of Europe and especially in Ireland where the most recently available statistics suggest that approximately 64% of the terrestrial landscape is under agricultural management (71% including 'rough grazing') (Anon., 2005). As a consequence of farming activities over millennia, farmers and the husbandry systems they have developed, have been very largely responsible for creating a varied and until relatively recently often highly biodiverse, 'natural' environment. However in the last 50-60 years, a marked increase in the intensity of agricultural systems across much of Europe has had widely negative impacts on numerous dimensions of environmental quality including air, water and soil quality, biodiversity, the visual appearance of traditional landscapes and cultural heritage. In the case of biodiversity, more recent negative changes have also resulted from the gradual abandonment of traditional, particularly pastoral, farming practices in economically marginal production areas. Such changes show that too little, as well as excessively intensive agricultural management can have strongly detrimental impacts on biodiversity (Council of Europe, 2001).

Such impacts have led to a widespread recognition that agricultural policy and the subsequent actions of farmers profoundly influence the 'natural' environment, and that Europe's farmers are key custodians and potentially some of the most effective managers of terrestrial biodiversity. In Ireland this is particularly true since despite the often very high intensity of certain types of farming such as dairying, the topography of the farmed landscape (i.e. the size and diversity of individual farms and fields) has remained relatively un-intensive. In contrast to many other parts of N.W. Europe, much of Irish agriculture is still practiced within a landscape that retains a relatively high proportion of field boundaries and other important habitats that support a diverse mix of wildlife, which in many areas (particularly in western Ireland) can be considered of 'High Nature Value' (EEA, 2004).

In response to international commitments under the Convention on Biological Diversity (CBD) to integrate biodiversity policy into all economic sectors (UNEP, 1992; see also Usher, this volume), the European Commission (2000, 2001) began to integrate environmental concerns into new agricultural policy and published a Biodiversity Action Plan (BAP) for Agriculture (COM(2001)162 vol. III). The BAP is intended to assist in the Commission's commitment to halt the global decline in biodiversity by 2010, and sought to describe in a holistic way the varying impacts that changing farming practices have on biodiversity. It laid out three major trends of concern relating to ongoing decline in:

1 the genetic diversity of domesticated plants and animals utilised within agricultural systems
2 the 'wild' biodiversity (species and habitats) associated with farmland and farmed landscapes
3 the 'life support systems' (fundamental ecosystem processes) that maintain the ecological integrity of 'agro-ecosystems'.

Recognising that much of the Irish landscape is under the direct influence of agricultural practice, the Ag-Biota project funded by the Irish Environmental Protection Agency under the ERTDI Programme 2000-2006, focuses on the latter two of these concerns. In particular, Ag-Biota seeks to generate knowledge necessary to inform agricultural policy development with respect to the surveillance and management of impacts on biodiversity. To achieve this aim, Ag-Biota has a programme of four well defined, but closely interrelated *Actions*:

Action 1: aiming to assist in the development of monitoring methodologies, which might be used to document and track the changing impact of agriculture and agricultural policy on the status of biodiversity within farmland.

Action 2: aiming to identify the major farm husbandry practices which define or limit biodiversity within existing and alternative farming systems.

Action 3: aiming to study in detail the ecology of functionally important populations within Irish agro-ecosystems, with a view to the more effective utilisation of the benefits of biodiversity within production systems.

Action 4: aiming to address more theoretical and fundamental questions regarding the ecological value and functional significance of diversity in plant and animal communities.

In this regard, Ag-Biota's aims are closely aligned with three of five specific research priorities that were recently identified as priorities for the Agriculture BAP at a meeting of the European Platform for Biodiversity Research Strategy (EPBRS) held under the Irish Presidency of the EU in Killarney, (EPBRS, 2004). These recommendations (current authors' italics) are to:

Recommendation 14: Assess the performance of the reformed CAP in achieving the target of halting biodiversity loss by developing a harmonised framework for evaluation, *and urgently supporting the development of monitoring systems using agreed indicators*.

Recommendation 16: *Improve the design, implementation, monitoring and evaluation of agri-environmental instruments* at the scales at which they most effectively deliver on the 2010 biodiversity targets.

Recommendation: 17: *Develop ecologically-based agricultural and food supply systems that enhance biodiversity and utilise its benefits*, starting with research for conservation of the most vulnerable and potentially useful species.

In this paper, we present an overview of the main issues and a brief description of the goals of the Ag-Biota work programme with respect to its four Actions listed above. A number of specific 'work in progress' papers, which were presented along with a number of posters illustrating the work of Ag-Biota at a conference held in University College Dublin on 30th March 2005, follow in this volume.

In Actions 1 and 2, Ag-Biota is seeking to develop practical methods to monitor, assess the current status of, and potentially track trends of change in biodiversity within Irish farmland. This work concentrates on grassland farming, largely comprising Habitat GA 1 – Improved Grassland, as defined by Fossitt (2000), since this is the predominant form of Irish agriculture. In this work, the project is seeking to answer *apparently* simple questions, such as what should be monitored: i.e. what are the most appropriate biological indicators of biodiversity impacts in Irish farmland; and how can these impacts be most easily monitored in practical terms: i.e. what relatively simple (surrogate) farm system parameters reliably predict impacts on the chosen biological indicators?

I. Selection of biodiversity indicators for agriculture

A large literature has grown on the topic of biodiversity indicators since Europe adopted the principles of the CBD, guided by inputs from agencies such as UNEP (1997, 1999) and the EEA (2003). The OECD assisted considerably by publishing an Agri-biodiversity Indicators Framework (Figure 1) to guide the development of specific indicators for the agriculture sector (OECD 2001).

This holistic framework provides a basis for developing a hierarchy of indicators relevant to different biodiversity dimensions (genes to habitats), recognising the complex interactions between the different elements of the agro-ecosystem itself, and its interactions with other terrestrial and aquatic ecosystems

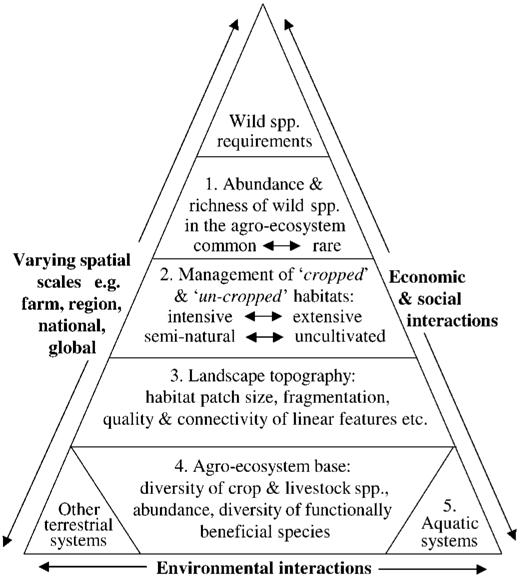

Figure 1. Conceptual agri-environmental indicator framework (modified after OECD 2001). The framework provides a hierarchical overview of the multiple spatial and temporal scales and environmental and socio-economic interactions that indicators need to address. Numbers within the framework have been added to indicate areas of direct relevance to Ag-Biota.

at different spatial scales and within different ecological and socio-economic contexts.

The efforts of Ag-Biota are concentrated on particular dimensions (numbered 1-5 in Figure 1) within this indicator framework, namely:

1. Selection of wild (uncultivated) biological indicator populations which best reflect the wider impact of farming within agro-ecosystems.
2. Selection of 'surrogate' farm husbandry indicators that accurately predict these likely impacts
3. Selection of 'surrogate' landscape structure indicators that accurately predict the effects of wider farm management impacts.
4. Selection of functionally relevant (beneficial) bio-indicator populations that best reflect the *ecological integrity* of the agro-ecosystem.
5. Selection of indicators for interactions between the agro-ecosystem and the quality of fresh water aquatic ecosystems.

Ag-Biota is currently *not* looking at indicators relating to the specific ecological requirements of particular 'flagship' wild species – indicators that link habitat quality and quantity at the apex of the OECD framework (Figure 1). Neither is the project looking at agro-ecosystem base indicators relating to the diversity of cultivated crops and livestock, or indicators of interactions between the agro-ecosystem and other *terrestrial* ecosystems.

In selecting indicators representative of 'wild biodiversity' within agro-ecosystems, the OECD (2001, Annex 5) provided a number of specific guidelines (current authors' italics):

1. Select a *minimum set* of wild species collectively representing a wide range of habitat types across agricultural land
2. Select a range of wild species that require different types of agricultural land and from *various species groups*
3. Select rare, endangered and *widespread* species
4. Select wild species *relevant to policy issues* at different scales from the local to global level.

Selection of bio-indicators for Irish agriculture

Ag-Biota has begun an initial monitoring programme at a range of ten potentially long-term monitoring locations in south-east Ireland. These sites are paired, consisting of four Teagasc Research Centres paired with nearby commercial farms of similar enterprise, and two separate locations of differing management type on the UCD farm at Celbridge, Co. Kildare (Table 1). A common programme of monitoring in early and late summer (May/June and July/August, respectively) for a wide range of taxonomic groups at appropriate field, farm or landscape scales was begun in 2002 and continued in 2003 (Table 2). At all sites, within field groups were assessed within grassland fields.

Following initial analysis of the data collected

Table 1: Location and principal farming types at Action 1 monitoring sites

Region	Location	Farming type
Wexford	Teagasc, Johnstown Castle	Intensive dairy grassland (250-300 kg.ha^{-1})
	Paired commercial farm	Intensive dairy grassland (300 kg.ha^{-1})
Tipperary	Teagasc, Solohead	Intensive dairy grassland (250 kg.ha^{-1})
	Paired commercial farm	Intensive dairy grassland (300 kg.ha^{-1})
Meath	Teagasc, Grange	REP-type beef grassland (100 kg.ha^{-1})
	Paired commercial farm	REPS drystock grassland (100 kg.ha^{-1})
Carlow	Teagasc, Oak Park	Mixed arable/drystock park grassland (120 kg.ha^{-1})
	Paired commercial farm	Mixed arable/drystock park grassland (120 kg.ha^{-1})
Kildare	UCD farm	Intensive dairy grassland (220 kg.ha^{-1})
	UCD farm	Extensive drystock grassland (80 kg.ha^{-1})

Table 2: Summary of sampled taxa and sampling methods used at Action 1 monitoring sites

Taxa	Sampling scale	Sampling method	Groups identified to species level (unless indicated otherwise)
Field vegetation	Field	3dm^2 quadrats	All sward flora
Vegetation arthropods	Field	*Vortis* suction samples	Total arthropods including: Coleoptera, Hemiptera, Diptera**, Hymenoptera (Parasitica)***
Predatory arthropods	Field	Pitfalls/field edge sod samples*	Carabidae, Staphylinidae, Araneae
Earthworms	Field	Formalin quadrats	Lumbricidae
'*Aesthetic*' arthropods	Farm	Field margin transects, yellow pan & light traps	Hymenoptera (Apidae), Lepidoptera, Diptera (Syrphidae)
Birds	Farm/ landscape	Farm transects	All species
Aquatic invertebrates	Landscape	Multi-habitat kick sampling	All macroinvertebrates

* sampled in mid winter 2002, ** identified to family level, *** identified to genus level

in this monitoring programme, a 'basket' of the following four potential indicator groups have been provisionally identified:

1 Grassland arthropods, particularly parasitoid wasps (Hymenoptera: Parasitica)
2 Bees (Hymenoptera: Bombini (bumblebees)
3 Birds
4 Aquatic macroinvertebrates.

This selection of groups covers a wide range of taxa associated with different parts and dimensions of the Irish farmed landscape from individual fields (parasitoid wasps) to the wider terrestrial landscape (bees and birds) and adjacent environments strongly influenced by farming practice (rivers and streams). Our provisional list of bio-indicators for agricultural land does not include any aspect of plant diversity *per se*, which has often been viewed as a primary driver of other forms of biological diversity at higher trophic levels. This is certainly not because the project lacks 'plant interest', but is a consequence of the fact that within the context of moderately to intensively managed farm landscapes, botanical diversity is highly constrained within the crop ecosystem by husbandry practice, and in such a context, floristic diversity probably only has such relevance at a habitat scale in regions where appreciable areas of 'semi-natural' vegetation still survive within the agricultural landscape (Waldhardt *et al.*, 2003).

Parasitoid wasps

Theoretically, because of their high trophic position, highly specialised biologies and very close ecological relationships with practically all other insect groups (Godfray, 1994; Quicke, 1997; Hochberg and Ives, 2000), the Hymenoptera: Parasitica have a particularly good theoretical potential as bio-indicators within agro-ecosystems. Additionally, they have particular functional relevance as natural enemies of many pest species, a role that has lead to their being widely studied as highly specific biological control agents (e.g. Wharton, 1993). However, their potential as indicators of wider arthropod diversity within agro-ecosystems has received little or no attention from ecologists, very largely because of the comparative inaccessibility of their taxonomy. Our initial monitoring at Action 1 sites, and increasingly information from Action 2 (see below), suggest however that even when studied at the comparatively crude taxonomic levels of Family or Genus, the community structure of parasitic Hymenoptera has a quite novel potential to reflect the wider ecological impacts of changing farming practice on the abundance and diversity of their host insect groups (Anderson *et al.,* this volume).

Bumblebees

Bumblebees are large, conspicuous insects that are relatively easy to identify, and the number and relative incidence of the eighteen species known to occur in Ireland would be entirely feasible to monitor as an indicator of environmental change. Furthermore, along with birds, some mammals, and butterflies, members of the general public frequently observe, and are generally sympathetic to the conservation of bumblebees. As well as being of conservation interest, it can be argued that bumblebees are also potentially good indicators of functional biodiversity since wild bees, including bumblebees, play a pivotal role as pollinators both of wild plants and of crops in agricultural ecosystems. The presence and abundance of pollinators in agro-ecosystems have been proposed and actually used as indicators of nature protection status (Roots and Talkop, 1997). Because of the distances that bumblebees routinely forage from their nests (e.g. Knight *et al.,* 2005), they can be considered as likely indicators of change at the landscape, rather than field level. Hence landscape structure and the management of non-crop vegetation, such as in hedgerows, woodland margins, areas of semi-natural meadows etc., might all act as influences on bumblebee community structure. Such influences, and the potential of bumblebee populations as indicators of wider biodiversity changes at the landscape level, are the subject of both recent (Kleijn *et al.*, 2001; Sepp *et al.,* 2004) and ongoing research within the Ag-Biota project (Santorum and Breen, present volume).

Birds

Bird populations have been widely studied as indicators of environmental quality and specifically proposed as a ‘headline’ indicator group for ecosystem health within the context of agricultural land (CEC, 2000). Bird populations can be less sensitive to immediate environmental change than other groups such as plants, but they are generally considered valuable indicators of biodiversity at landscape level over an extended period of time. Birds are relatively easy to identify and study, are high in the food chain, highly mobile (enabling broad spatial monitoring) and there are often large amounts of historical data available for assessment of temporal change. However bird populations are regulated by density-dependent processes, so their populations are often buffered against environmental changes (Furness and Greenwood, 1993).

As previously stated, agriculture represents the most extensive form of land use in Ireland and much of Europe, and decline in bird populations and a contraction in the range of a number of farmland species have been widely reported over the past 20-30 years in northern Europe (Marchant *et al.*, 1990; Fuller *et al.*, 1995; Siriwandera *et al.*, 1998). Fuller *et al.* (1995) have shown that the majority of species associated with farmland in the UK have undergone a contraction in range and a number of species have declined significantly between 1960 and 1990. These declines have been particularly associated with species inhabiting arable, mixed and grass farmland, more so than species associated with other habitats such as woodland, uplands and wetlands over the same time period (Chamberlain and Fuller, 2002).

In Europe, declines in mean farmland bird population density between 1970 and 1990 have been greater in Western European countries than in the former Eastern-block countries (Donald *et al.*, 2001) These declines have been associated with increased agricultural productivity in Western Europe at a time of decreasing productivity in Eastern Europe (Donald *et al.*, 2001). The principal changes in farming associated with the decline in bird species have been the loss of mixed farming, the drainage of wet grasslands, loss of rough grazing, intensification of grassland management including increased use of fertilizers and pesticides, and the overgrazing of upland areas (Donald *et al.*, 2002).

In Ireland, a decline in populations and a decrease in the range of some farmland species were recorded in the latter quarter of the 20th century (Newton *et al.*, 1999). The breeding populations of farmland species such as corncrake, grey partridge, lapwing, yellowhammer and chough declined by more than 50% during this period. The Countrywide Bird Survey, which began in 1998, has shown that these declines have not necessarily continued in recent years (Coombes, 2004). Trend analysis of data for 55 species from 1998 to 2003 suggest that only robin, stock dove, grasshopper warbler, cuckoo, wheatear and skylark declined during this time, while a further twelve species actually increased and the remaining species showed stable population densities. Why some insectivorous and granivorous species continue to decline, while others are apparently stable or increasing is difficult to answer. However, the Countrywide Bird Survey has taken place over a period when the Rural Environmental Protection Scheme (REPS) and other protection measures are hopefully beginning to have an impact on the quality and management of farm habitats. The situation for migrant species is also complicated by coincident changes in the environments beyond Europe where they overwinter.

Aquatic macroinvertebrates

Macroinvertebrates have been extensively used as indicators in water quality monitoring and assessment and a variety of biotic indices have been derived particularly for rivers (Rosenberg and Resh, 1993). The indicator potential of aquatic fauna stems from the fact that their various components exhibit a wide range of environmental preferences, and a graded response to a wide range of anthropogenic pressures. Data on aquatic community composition are often reported as biotic indices, such as the Irish EPA Q-value system (Mc Garrigle *et al.,* 2002), and various indices have been developed to communicate information on prevailing water quality or ecological state. In recent years, comparative analyses have become more complex and have taken on board the concepts of 'reference conditions,' i.e. conditions of minimal human impact on biological, hydrochemical and hydromorphological elements (Bailey *et al.*, 2004), biological integrity (Karr, 1991; Karr and Chu, 2000) and aquatic ecosystem health (Reynoldson and Metcalfe-Smith, 1992). Comparisons with reference state, as incorporated in the **R**iver **I**n**V**ertebrate **P**rediction **A**nd **C**lassification **S**ystem (RIVPACS) (Wright *et al.*, 2000), permit evaluation of the extent of biodiversity change or community restructuring. Invertebrate data, however, can often also provide information on the drivers and pressures causing a shift from reference condition at scales ranging from reach-level to catchment or landscape-level. It is likely that the abiotic and biotic factors structuring invertebrate communities operate at different physical and temporal scales (Sponseller *et al.*, 2001), and thus detailed knowledge of the river itself and its linkages to the catchment are required to elucidate the reasons for any observed change. These linkages have been well illustrated by many studies since first proposed by Hynes (1960). Recent studies have elaborated on this concept and emphasise four dimensions relating to the connectivity between the river and landscape: longitudinal flow from source to sea, lateral links with the flood plain, vertical connections to groundwater and finally the temporal components (Pringle, 2003). In terms of the current Ag-Biota studies, the main objectives are to provide information of the state of macroinvertebrate biodiversity within catchments influenced by intensive agriculture, and to compare community composition and structure with that found at appropriate reference condition sites.

Further information from Ag-Biota's studies justifying the provisional selection of all the above-mentioned groups as biological indicators is provided by Anderson *et al.*, Santorum and Breen, McMahon *et al.* and Baars and Kelly-Quinn within this volume.

II. Identifying the major pressures on biodiversity within Irish farming systems

Ag-Biota seeks to support and inform agri-environmental policy, such as the Rural Environmental Protection scheme (REPS), by assisting in the identification of those farm system and wider landscape management practices that most strongly impact on biodiversity. In this aim, we are essentially seeking to better understand the main farming 'pressures' causing significant biodiversity loss within the Irish countryside. This will assist policy making so that REPS and other policy tools can better address the underlining 'drivers' that create such pressures in the first place (EEA, 1999). Actions 1 and 2 are closely linked in achieving a better understanding of the relationship between what farmers do and subsequent effects on biodiversity (Figure 2).

In achieving this understanding of causality, we also (in theory at least) assist in the identification of 'surrogate' indicators that are *likely* to be indicative of actual trends of change in biodiversity. Such indirect indicators are of great practical significance, since they can be tracked with less demand for technical expertise and resources than would be required to *routinely* survey a wide range of biological indicators (see Rath *et al.*, this volume).

Impacts of grassland husbandry

Ag-Biota is making use of a number of already existing grassland management experiments being conducted at Teagasc Research Centres

These are all conventional replicated plot experiments contrasting the intensity of grassland management systems in terms of nitrogen use and stocking rate. None of these experiments were specifically designed to investigate effects on biodiversity, and both high and low input level treatments are being managed in a largely standardised pattern of short cycle rotational paddock grazing. This *may* explain why, in our analysis of results to-date, we have found relatively few effects of input levels on the diversity of either the sward flora or its arthropod fauna. Even when such effects have been found, this has required relatively high levels of sampling effort. Publication of these findings awaits completion of an ongoing and fuller analysis of all our data. However, our suspicions that the relatively small treatment effects so far documented in these experiments may be a consequence of the over-riding effect of the intensive rotational grazing of all treatments is supported by a much older study (Purvis and Curry, 1981), which showed that sward structure as influenced by the manner of grazing and conservation was the primary determinant of arthropod abundance and diversity in agricultural

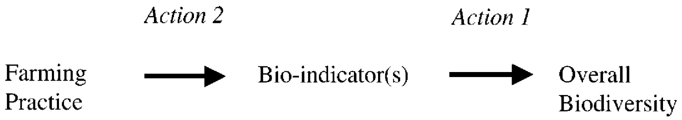

Figure 2: The relationship between Actions 1 and 2: understanding the influences between farming practices (potential indirect surrogate indicators), biological indicator groups and wider impacts on biological diversity, arrows represent the underlying relationships being studied.

Table 3: Summary of grassland management experiments being monitored for biodiversity effects at Teagasc Research Centres

Location	Experimental comparison	Taxa sampled
Johnstown Castle	Dairy grassland management: intensive (390 kg.ha^{-1}) vs semi-intensive (225 kg.ha^{-1}) vs extensive (0 kg.ha^{-1})	Sward flora Vegetation arthropods Predatory arthropods Earthworms
Grange	Suckler beef grassland management: semi-intensive (225 kg.ha^{-1}, 0.81 ha. cow^{-1}) vs extensive (100 kg.ha^{-1}, 1.0 ha. cow^{-1})	
Solohead	Dairy grassland management: semi-intensive (215 kg.ha^{-1}) vs extensive (90 kg.ha^{-1})	

grassland. Such a conclusion is also supported by ongoing Ag-Biota work that is demonstrating the biodiversity benefits of creating ungrazed conservation margins at the edges of rotationally grazed dairy grassland paddocks at Johnstown Castle.

This paddock margin study was initiated as a Teagasc-funded Walsh Fellowship investigation of botanical and hoverfly (Syrphidae) diversity in variously managed fenced marginal strips from 1.5 to 3m wide (Sheridan, 2005). Ongoing Ag-biota monitoring of the experiment has extended observations to a much wider range of vegetation arthropods. Results after just one season of these faunal observations in 2004 are showing a clear increase in arthropod diversity in all fenced margin treatments compared to the grazed paddock. Furthermore, a marked influence on the relative abundance of different parasitoid wasp families in the fenced margin strips compared with the grazed pasture suggests that the composition of the parasitoid wasp fauna (perhaps reflecting the relative abundance of different wasp host groups) may be a very sensitive indicator of the influence of grassland management (Purvis *et al.,* 2005; Anderson *et al.*, this volume). These results, if borne out in further monitoring, have important implications for the development of REPS measures, which currently seek to limit the intensity of grassland inputs and actual stocking rates, but do not address at all the possibility that grazing patterns and the intensity and frequency of grass defoliation may have a greater impact on biodiversity. The paddock margin experiment at Johnstown Castle will continue to be monitored in 2005 in order to determine whether faunal differences between the different fenced margin treatments develop over time, as might be expected.

Impacts of farming on adjacent ecosystems

Aquatic macroinvertebrates have been provisionally identified by Ag-Biota as potential indicators of farming's impact on biodiversity beyond the agro-ecosystem itself. As previously discussed aquatic macroinvertebrates are routinely used as bio-indicators of water quality and the state of freshwater ecosystems. Due to the integral link between rivers and the adjacent landscape, changes induced by intensive farming are likely to be reflected in the faunal biodiversity of freshwater ecosystems at various spatial scales. To date, in comparison with streams in reference condition, freshwater catchments at Action 1 sites show a notable difference in the faunal diversity and community composition (Baars and Kelly-Quinn, this volume). The observed reduction in biodiversity, and the significant changes in the invertebrate community structure, suggest a negative impact induced by surrounding land use practice, which in these catchments is very largely farming.

Impacts of landscape management

Bird and bee populations have been provisionally identified as likely bio-indicators for the influence of farming on the wider landscape. Most notably, studies across Action 1 sites indicate a significant relationship between field boundary attributes, as quantified by the Field Boundary Evaluation and Grading System (FBEGS) (Collier and Feehan, 2003), and the diversity of both breeding and over-wintering bird populations (McMahon *et al.*, this volume). Should this relationship stand up to further scrutiny, the FBEGS method may prove to be a most useful surrogate for monitoring ongoing change across the farmed landscape.

The next step in developing indicator methods

Having identified a very provisional set of potential bio-indicators and uncovered 'hints' regarding some of the more important influences of farm management on these groups, our next step is to extend our monitoring to a larger number of sites, with the following primary aims:

- To provide a more rigorous and comprehensive test of our initial findings with respect to the selection of indicator groups and refinement of sampling protocols.
- To provide an alternative approach to identifying the major husbandry/farm management pressures on biodiversity using multivariate analyses of the relationships between farm system and landscape management characteristics and the diversity of appropriate bio-indicator groups.

In this next phase of monitoring, Ag-Biota will test the insights gained from our earlier work by collection of a larger, independent data set relating to the diversity of sward flora and fauna and bird populations at approximately 50 randomly selected commercial farms located in south-east Ireland, along with relevant data relating to farm system and landscape characteristics. The resulting comprehensive data set will help to address many questions relating to the selection of indicators and the development of assessment systems to track the ongoing impacts of agriculture on biodiversity.

III. Utilising the ecological and agronomic benefits of biodiversity in agricultural systems

That many biological populations within agro-ecosystems have an indispensable ecological role and an intrinsic agronomic value is not a new revelation, but warrants restatement in the light of the current, and arguably quite justified preoccupation with wider conservation issues in biodiversity research. Farm production systems essentially depend upon the effective management and exploitation of complex natural biological processes. Although there has been an increasing tendency to forget or even ignore this fact in the development of more intensive farm systems, it nevertheless remains a truism that many biological processes and the populations associated with them remain essential to ecologically sustainable agriculture (Altieri, 1999). As a consequence, a large part of Ag-Biota's research effort is focussed on such functionally important populations within the agro-ecosystem.

Biodiversity and primary production

The benefits of botanical diversity in agricultural grasslands depend on the environmental and managerial conditions in which the plants are grown. In controlled experiments in which growth resources (e.g. nutrients, trace elements and water) are not limiting and external stresses (e.g. water stress, selective grazing and sward damage) are largely absent, the plants in a sward effectively compete for light.

Competition for light is a positive feedback process, eventually favouring the dominance of species that convert light energy into above-ground biomass most efficiently (Schulte *et al*., 2003). Such conditions are commonly maintained in the herbage variety trials that determine recommended seed lists; this has resulted in the adoption of recommended seed mixtures comprised exclusively of varieties of perennial ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*).

However in actual farming practice, resources for plant growth are rarely uniformly distributed and unlimited, and meteorological and managerial stresses are the norm rather than the exception. In such conditions, experiments that have manipulated the diversity of plant species and/or functional groups in plant communities have generally (but not always) found positive relationships between diversity and above ground biomass (Hector *et al*., 1999).

There are multiple mechanisms by which such positive effects may come about. Increased resource utilisation is one such mechanism, whereby more diverse mixtures of species more fully use available resources due to niche differentiation e.g. more diverse plant communities may have greater root or canopy volume for capturing more soil nutrients and light, respectively. As an example, soil nitrate concentration reduced as plant species richness increased in experimental manipulations ranging from 1 to 24 species (Tilman *et al.* 1996). Positive species interactions are another mechanism, and include the role of nitrogen-fixing legumes in increasing the availability of nitrogen to other plant species. Another is the sampling effect, which refers to the greater probability that a mixture of species will contain a dominant productive species than a particular monoculture (Bolger, 2001).

There appears to be a widespread generality to the finding that diversity supports more effective ecosystem functioning. For example, more diverse agricultural seed mixtures were found to increase the productivity of hay meadows established on ex-arable land (Bullock *et al.,* 2001). Species and genetic diversity of plants in European grasslands may bring benefits to agriculture through effects on biomass production, retention of soil nutrients and water use (Minns *et al*., 2001, Joshi *et al.,* 2001). In an experiment replicated across sites in Europe to assess the benefits of grass/legume mixtures, results from the first year of sampling suggest that mixing grass and legume forage species enhanced above-ground biomass and decreased the proportion of unsown species (Kirwan *et al.,* 2005). In addition, plant species mixtures facilitate a greater diversity of interactions at both higher (e.g. pollinators, herbivores, predators and parasitoids) and lower (e.g. mycorrhiza) trophic levels. In one of the largest agricultural studies ever conducted, thousands of farmers in Yunnan Province in China planted mixtures of two rice varieties, instead of the traditional planting of a single variety. Resulting yields increased by 89% and the severity of rice blast (a severe fungal disease of rice) was reduced by 94% (Zhu *et al*., 2000).

Biodiversity and soil processes

The linkages between soil organisms and the key processes that influence soil quality, fertility and plant growth are widely recognised but still poorly understood. To date, no clear relationship between soil biodiversity and ecosystem function has been established, and most experimental studies tend to support the 'redundant species' hypothesis, i.e. they do not demonstrate improved functionality at higher levels of species richness (Wolters, 2001; Bolger, 2001). This is perhaps not surprising given the enormous diversity of the soil biota, the complexity of soil food web interactions and the multiplicity of functions in which soil organisms participate. As pointed out by Bengtsson (1998), there is no direct mechanistic relationship between diversity and ecosystem function; the crucial issue is what those species that are present actually do. Thus, in the context of sustainability, there is a particular need to understand the linkages between key species or functional groups and ecosystem processes.

Defining functional groups that are relevant to key ecosystem processes, and refining the criteria for assigning species or higher taxa to these functional groups are important areas for research. Currently taxa are often assigned to functional groups on the basis of questionable assumptions about trophic interactions, which are frequently inferred from gut content analysis or laboratory studies on model species. These studies may not reliably reflect the full range and strength of trophic links in the field and take no account of potentially important non-trophic interactions. Novel techniques, such as the use of stable isotopes and molecular PCR methods (see below), provide new and powerful tools for investigating trophic relationships and matter fluxes in ecosystems.

Soil biodiversity can sometimes be negatively correlated with increasing levels of fertility, at least in the case of invertebrates in habitats managed for agriculture (Curry, 1986, 1994). The invertebrate communities of semi-natural grasslands tend to be complex and species-rich, reflecting the varied habitats and resources that are provided by diverse vegetation and a well-developed litter layer. By contrast, micro-arthropods tend to be less abundant and less species-rich in ryegrass-dominated, intensively managed swards where soil organic matter is rapidly decomposed and mineralized. In such conditions some groups, notably lumbricid earthworms, can benefit greatly from the increased fertility and high quality plant litter, and their biomass tends to be positively correlated with management intensity (Muldowney *et al.,* 2004). Such relationships are the subject of ongoing Ag-biota investigations. Earthworms in turn exert a profound influence on soil fertility and structure and on the nature of the soil as a habitat for other biota, particularly micro-organisms. These effects are seen clearly when earthworms become established in soils from which they were previously absent (Hoogerkamp *et al.,* 1983; Stockdill, 1982; Vimmerstedt, 1983; Boyle *et al.*, 1997). However, our current understanding of how earthworm activity influences the diversity of other soil organisms is particularly limited (Brown and Doube, 2004; Parkinson *et al.,* 2004).

Biodiversity and trophic interactions

Linking biodiversity with ecosystem functions in grassland systems requires detailed knowledge of soil–plant–animal trophic relationships at species or functional group level (Jones and Bradford, 2001). For example, detritivorous soil invertebrates such as earthworms, which were previously regarded as simply ‘omnivores’, comprise different feeding guilds that play quite dissimilar functional roles in processes such as organic matter decomposition (Briones and Schmidt, 2004). Until recently, traditional research methods have provided limited information concerning these subtleties. However, stable isotope ratio analysis of biologically relevant light elements offers a powerful research tool to reveal and quantify trophic linkages in food webs. The strengths of these techniques are that they can be applied to study real, undisturbed food webs in the field, they reflect assimilated rather than ingested dietary components, and they can be used to quantify matter fluxes between compartments or organisms as functional processes.

Stable isotope ratio techniques measure the ratios of stable isotope pairs, such as $^{13}C/^{12}C$ (carbon), $^{15}N/^{14}N$ (nitrogen) and $^{34}S/^{32}S$ (sulphur), usually by isotope ratio mass spectrometry (IRMS) (Scrimgeour and Robinson, 2004). Two broad approaches have been found useful in ecological studies, namely natural abundance and tracer addition methods. Natural abundance methods measure the ratios of naturally occurring stable isotopes and thus reflect undisturbed, *in situ*, processes. In animal ecology, this approach relies on known isotopic changes (fractionations) that occur along food chains to infer food sources from C and S isotope ratios, which undergo little fractionation, and the trophic level of animals from their N isotope ratios, which increase along food chains in a step-wise fashion (Schmidt *et al.*, 2004). For example, known C isotopic differences between C_3/C_4 photosynthetic plants can be exploited to quantify host plants in the diets of insect pests (Ponsard *et al.*, 2004). Also, N isotope ratios have recently been used to show that 36 coexisting species of minute soil mites, which traditionally have been difficult to study by any other method, occupy up to four different trophic levels within food chains (Schneider *et al.*, 2004).

Deliberate introductions of isotopically enriched tracers can provide increased resolution, which facilitates tracing elements through food chains at smaller scales, e.g in micro-plot experiments. Grass biomass, for instance, can be isotopically labelled to trace the fate of C and N from decomposing plant residues into different compartments of soil food webs (Schmidt and Scrimgeour, 2001).

The research within Ag-Biota on stable isotope techniques (Action 3 in part), ideally complements the main focus of the wider project, namely the conservation and utilisation of biodiversity in grassland ecosystems. One specific objective of this research is to develop stable isotope techniques for the investigation of trophic, and thus functional, diversity of ground beetle (carabid) populations. These beetles are generalist predators or omnivores, important in pest control (see below). The natural abundance of C and N isotope ratios are being used in Ag-Biota to investigate their feeding ecology. Of particular interest is the extent of polyphagy in these predatory species, some of which feed on both plant pests (e.g. aphids, slugs) and other detritivorous soil animals such as earthworms and springtails (McNabb *et al.*, 2001), and the significance of the abundance and diversity of alternative prey to polyphagous predator populations and pest regulation. Such knowledge will facilitate the development of crop management practices that better utilise natural mechanisms of pest control and maximise the functional exploitation of biodiversity for pest management. Another objective of this research is to quantify the role of different coexisting earthworm species belonging to different functional groups in the incorporation and decomposition of cattle dung triple-labelled with ^{13}C, ^{15}N and ^{34}S. This work will facilitate the quantification of dung-feeding activity in undisturbed field populations of individual earthworm species based on measured isotopic enrichment over time, and will strongly complement more theoretical studies of the functional significance of diversity in earthworm community structures (Action 4 – see Section IV below).

Biodiversity and pest control

Following a period of heavy reliance on chemical pest control methods in an attempt to ‘override’ and ‘control’ nature, increasing attention is now being paid to a much broader range of pest management methods (Metcalf and Luckmann, 1994). These include the implementation of an increasingly sophisticated range of biological pest control methods, including the achievement of natural pest regulation through the conservation of native natural enemy populations (Van Driesche and Bellows, 1996). The management of biological diversity and agricultural pest ecology are therefore closely inter-linked (Price and Waldbauer, 1994).

A large body of both theoretical and practical research has shown the parasitoid wasps (Hymenoptera: Parasitica) to be particularly effective natural enemies of insect pests (e.g. Waage and Greathead, 1986; Waage, 1990). It is therefore, particularly relevant that the diversity of this particular insect group should be showing such promise in our Action 1 and 2 studies as a potential bio-indicator of wider insect diversity and grassland husbandry in agro-ecosystems. Less obviously, many generalised (non-specific) predatory insects, including beetles in the families Carabidae and Staphylinidae and spiders of the family Linyphiidae are potentially very abundant and important predators of a wide range of pest groups, especially aphids. However, this fact has only been fully appreciated since the advent of molecular methods of predation diagnosis in predators that might not ingest solid recognisable fragments of their prey (Sunderland *et al.,* 1987; Symondson, 2002). Such findings have generated a considerable literature relating to the population ecology and ecological significance of such *polyphagous* predator populations in agro-ecosystems (Lovei and Sunderland, 1996; Kromp, 1999; Holland and Luff, 2000; Nyffeler and Sunderland, 2003). Recent improvements in the sophistication of molecular methods of prey detection have resulted in the possibility of detecting the presence of multiple prey species within a single individual predator (Agusti *et al.,* 2003). This technology now rivals the use of stable isotopes as a basic investigative tool for studying both the theoretical and practical significance of both predator and prey biodiversity within agro-ecosystems (Symondson *et al.,* 2002) and is being used specifically in Ag-Biota studies investigating the ecology of aphid vectors of barley yellow dwarf virus (BYDV).

Biodiversity and pollination

Bees are among the most important pollinators of crops and wild plants. Other insect groups may collect pollen and nectar for their own consumption. However, bees also collect pollen and nectar to provide for the next generation. Hence they make more flower visits than other insects and are well known to show constancy to individual plant species while on foraging trips (e.g. Corbet *et al.*, 1991). Bees with different tongue lengths visit flowers with different corolla lengths – this results in an effective partitioning of the available forage, but also demands the presence of different bee species for the effective pollination of a range of different plant species. The farmland flora is effectively divided amongst different pollinating species (Proctor and Yeo, 1973) and plants and their pollinators are inter-dependent. Many crops of national importance depend to a large extent on pollination by bees, including bumblebees. The seed set of forage species such as red clover, crops such as oilseed rape and beans, and fruits such as apple, raspberry and strawberry either depends on, or is enhanced by, bee pollination (see Corbet *et al.,* 1991 for a comprehensive listing). Sometimes natural pollination can be supplemented by providing honeybee colonies. However, bumblebees can be more effective pollinators of some crop types, such as red clover, and the keeping of honeybees is currently under economic pressure from a major pest problem in the form of *Varroa* mite infestation of bee hives.

Ag-Biota studies on the ecology of beneficial populations in the agro-ecosystem

Recognising the need for greater information concerning the ecology of specific populations and more effective utilisation of the ecological benefits of biodiversity within agro-ecosystems, the Ag-Biota project includes nine individual postgraduate studies addressing a wide range of the above issues (Table 4).

IV. Alternative experimental approaches to studying the functional significance of biodiversity

Central to resolving current controversies in biodiversity/function research is a growing appreciation of the roles of different experimental designs in biasing the results and interpretation, i.e. experimental design may confound the determination of the relative contribution of different facets of biodiversity

Table 4: Summary of individual Ag-Biota studies on the ecology and agronomic significance of floral and faunal populations in farmland.

Topic	Student/ Supervisor	Location
Management strategies to integrate botanical diversity and grassland productivity	Ilse Geijzendorffer/ Rogier Schulte	Johnstown Castle
Studies on the status and conservation of aesthetic arthropod populations in farmland	Veronica Santorum/ John Breen	University of Limerick
The population ecology of predatory carabid beetles in Irish crop systems	Brigitte Henatsch/ Gordon Purvis	UCD
Functional significance of biodiversity: the links between soil cultivation, predatory arthropods and the ecology of cereal aphids and barley yellow dwarf virus (BYDV)	John McDonald/ Tom Kennedy	Oak Park
The ecology and diversity of bird populations in farmland	Barry McMahon/ John Whelan	UCD
The use of stable isotopes in field studies of functional ecology	Dirk Miksche/ Olaf Schmidt	UCD
The abundance and diversity of earthworm populations in agro-ecosystems	Paul Doherty/ Jim Curry	UCD
Development of novel Simplex designs and analytical methods to study biodiversity and ecological function	Laura Kirwan/ John Connolly	UCD
Simplex studies of the effect of earthworm biodiversity on primary productivity and decomposition	Michael Doyle/ Tom Bolger	UCD

(Connolly *et al.*, 2001a; Allison, 1999). In most previous experiments, the separate effects of community structure and ecosystem function and the manner in which they interrelate have not been addressed, nor has this interaction generally been studied across climatic and management regimes. Importantly, most experiments have used replacement designs, which have been criticised for containing confounded species density treatments and size-bias; few have used more than two species (e.g. Gibson *et al.*, 1999; Connolly *et al.*, 2001b).

A new system of design and ecological modelling is being developed to provide a framework to investigate questions of function at community level and of structure and competition at the level of species. It provides a set of statistical models that allow the separate assessment of initial overall abundance, species richness, species evenness and environment. The proposed experimental designs are based on the simplex (Cornell, 1990) at each of at least two overall densities. For *s* species, each experimental community will consist of up to *s* species and can be represented as a point in an *s-1* dimensional simplex. For three species (Figure 3), the simplex vertices represent monocultures of each species and the central point a mixture in which the initial abundance of all species is equal. A design consists of stands defined by two simplexes, each at a different levels of total initial abundance The simplex methodology provides a simple framework for selecting communities. The design selected will depend on *s*, the number of species, on whether one is primarily interested in questions of function or structure, the complexity of the models to be fitted and considerations of design power. Designs for larger numbers of species, which are not excessively large in number of experimental stands required, can be constructed.

Statistical models are proposed for the analysis of data from these experiments. For functional responses such as yield, the models contain terms that assess the effects of species identity, overall initial abundance, environ-

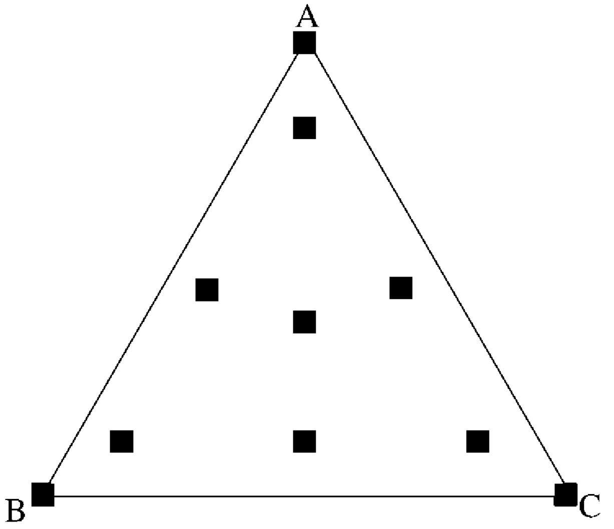

Figure 3: A Simplex design for investigation of the functional influence of community structure in a community of three species; marked points represent the proportional incidence of each of the three species in a limited number of possible experimental combinations, e.g. points at the apices A, B and C represent 'pure stands' of each individual species, whilst the central point represents an experimental community structure in which all three species are equally represented.

ment, species interaction and evenness. The interaction terms are interpretable as synergistic and antagonistic inter-specific relationships whose impact on the response depends on stand evenness. For structural responses, the RGRD (relative growth rate difference) models proposed by Connolly and Wayne (2005) and Ramseier *et al.* (2005) allow assessment of the effects of species identity, initial species abundance and environment as determinants of change in community biomass composition.

Based on this approach, separate experiments have been made on community function and structure in mixtures of four plant species (two grass and two clover species) and on the functional diversity of earthworms. The former experiment is part of an international experiment with identical design repeated at more than 40 sites, mainly in Europe under the aegis of COST852. This study has now been closely integrated with the work of Ag-Biota, allowing the monitoring of effects on sward arthropod populations at the Irish site of this originally botanical experiment at the Teagasc Johnstown Castle Research Centre. Results from the first year of botanical observations have shown that there is a positive effect of increased species richness on yield and on the ability of the sward community to resist invasion by unsown species. The advantage of this experimental structure is that we can also see an effect of evenness. The more even the community structure, the more pronounced the effect of species richness. A further advantage of this experimental structure is that we can test for functional group effects. There was no effect on yield or resistance to invasion from mixing two grasses or mixing two legumes. There was however, a marked effect due to mixing a grass with a legume (regardless of species).

Our earthworm studies, also based on the novel simplex experimental design, investigated the effects of earthworm functional group diversity on nitrogen dynamics in soils. The experiments have been carried out at two overall earthworm densities, with varying community structures and at two levels of food supply and variable temperature regime. The mesocosm experiments use Plexiglas cylinders (15cm diameter x 30cm depth) containing 2.65 litres of soil as experimental units and leachate is being collected and analysed weekly. Soil nitrogen content and microbial activity are measured when the units are sampled destructively at the conclusion of the experiments. Results available show a differential effect of different earthworm functional groups on N dynamics. There are also obvious effects of population density, resource availability and temperature regime. A significant synergy exists between worms that live deeper in the soil and those that burrow vertically in terms of nitrate in soil. Differential effects of food supply were seen for the three functional groups. For example, there were reduced concentrations of nitrate-N in the leachate from the monocultures containing species that burrow vertically at low levels of food supply, while increased amounts were leached from units containing surface-dwelling species under the same conditions.

Food supply was the major driver leading to increased microbial activity and biomass. Increased earthworm biomass had the opposite effect. In the lower soil layer, the presence of surface feeders which produced vertical burrows had the greatest effect on microbial populations and activity. This effect was enhanced with increased food supply.

Overall, the relationship between earthworm functional group diversity and microbial functional diversity depends on the composition of the earthworm assemblage and the effect of the earthworms varies with soil layer, initial biomass and food supply. It is also apparent that nitrogen transformations and amounts available in soil water are dependent on the composition of the earthworm community.

V. Conclusions

Ag-Biota has made a significant start in undertaking research targeted at policy development specifically within the context of protecting and utilising biodiversity in agricultural systems. This is important considering the potential influence that agriculture has within the Irish landscape. In addressing concerns about biodiversity loss, the conservation of endangered species and protection of the declining number of 'special' habitats we still have is critically important and quite correctly receives widespread attention. From both a conservation and functional perspective, however, the protection of biodiversity within the variously managed agro-ecosystem, which represents the greater part of our natural heritage, is also extremely important. Without due attention and left unchecked, the potential contribution of changes in traditional agricultural systems to the loss of Irish biodiversity, and the possible ecological consequences of this loss for key agro-ecosystem processes are likely to be profound. Ag-Biota's work programme covers a necessarily very broad range of topics – but even so, this covers only a subset of the many potential biodiversity issues in agriculture: we are not, for instance, addressing the important issue of the conservation of genetic resources in agriculture. Ag-Biota is, however, addressing important practical questions regarding the selection of appropriate indicators and the development of knowledge necessary for the development of future monitoring/surveillance systems. At the same time, we are researching a wide selection of highly specific ecological/functional questions relating to the better utilisation of biological diversity within the agro-ecosystem, the development and use of new tools for ecological investigation (such as stable isotope analysis and molecular methods), and asking quite fundamental questions concerning the ecological value of biodiversity using novel approaches to experimental design. As such, the Ag-Biota project represents a considerable investment in Irish biodiversity research at a time of increasing recognition of the important role of agriculture in managing and maintaining biological diversity in the Irish countryside.

Acknowledgements

Ag-Biota is a capability development project funded by the Environmental Protection Agency within the ERTDI Programme (2000-2006) as part of the National Development Plan. We wish to thank the EPA, especially Helen Walsh and Gerard O'Leary, for their support and encouragement during the organisation of the Ag-Biota conference held on 30th March 2005 and throughout the ongoing Ag-Biota project.

References

Allison, G.W. (1999). The implications of experimental design for biodiversity manipulations. *American Naturalist* **153**, 26-45.

Altieri, M.A. (1999). The ecological role of biodiversity in agro-ecosystems. In M.G. Paoleti (Ed.), *Invertebrate Biodiversity as Bioindicators of Sustainable Landscapes*, 19-32. Amsterdam, The Netherlands, Elsevier.

Anon. (2005). *Fact Sheet on Irish Agriculture – April 2005*. Economics and Planning Division, Department of Agriculture and Food, Dublin.
(Available at: http://www.agriculture.gov.ie/index.jsp?file=publicat/publications2005/index.xml).

Augusti, N., Shaylor, S., Harwood, J.D., Vaughan, I.P., Sunderland, K.D. and Symondson, W.O.C. (2003). Collembola as alternative prey sustaining spiders in arable ecosystems: prey detection within predators using molecular markers. *Molecular Ecology* **12**, 3467-3475.

Bailey R.C., Norris R.H. and Reynoldson, T.B. (2004). *Bioassessment of freshwater ecosystems using the reference condition approach*. Boston. USA, Kluwer Academic Publishers.

Bengtsson (1998). Which species? What kind of diversity? Which ecosystem function? Some problems in studies of relations between biodiversity and ecosystem function. *Applied Soil Ecology* **10**, 191-199.

Bolger, T. (2001). The functional value of species biodiversity—a review. *Biology and Environment: proceedings of the Royal Irish Academy* **101b** (3), 199–224.

Boyle K.E., Curry, J.P. and Farrell, E.P. (1997). Influence of earthworms on soil properties and grass production in reclaimed cutover peat. *Biology and Fertility of Soils* **25**, 20-26.

Briones, M.J.I. and Schmidt, O. (2004). Stable isotope techniques in studies of the ecological diversity and functions of earthworm communities in agricultural soils. In S.G. Pandalai (Ed.), *Recent Research Developments in Crop Science*, Volume 1, 11–26. Trivandrum, Research Signpost.

Brown, G.M. and Doube, B.M. (2004). Functional interactions between earthworms, microorganisms, organic matter, and plants. In C.A.Edwards (Ed.), *Earthworm Ecology* (2nd Edition), 213-239. Boca Raton, Florida, CRC Press.

Bullock, J.M., Pywell, R.F., Burke, M.J.W. and Walker, K.J. (2001). Restoration of biodiversity enhances agricultural production. *Ecology Letters* **4**, 185-189.

Commission of the European Communities (CEC, 2000). *Indicators for the Integration of Environmental Concerns into the Common Agricultural Policy*. COM(2000) 20, January, Brussels. (Available at: http://www.europa.eu.int/eur-lex/en/com/cnc/**2000**/com**2000**_00**20**en01.pdf).

Collier, M. and Feehan, J. (2003). Developing a field boundary evaluation and grading system in Ireland. *Tearmann* **3**, 27-46.

Connolly, J., Wayne, P. and Bazzaz, F.A. (2001a). Interspecific competition in plants: how well do current methods answer fundamental questions? *The American Naturalist* **157**,107-125.

Connolly, J., Goma, H.C. and Rahim, K. (2001b). The information content of indicators in intercropping research. *Agriculture, Ecosystems and Environment* **87**, 191-207.

Connolly, J. and Wayne, P. (2005). Assessing determinants of community biomass composition in two-species plant competition studies. *Oecologia* **142**, 450-457.

Coombes, D. (2004). Contrasting trends. *Wings* No. 35 (Winter 2004), p7.

Corbet, S.A., Williams, I.H. and Osborne, J.L. (1991). Bees and the pollination of crops and wild flowers in the European Community. *Bee World* 72, 47-59.

Cornell, J. A. (1990). *Experiments with mixtures*. John Wiley and Sons, New York.

Council of Europe (2001). *High-Level European Conference on Agriculture and Biodiversity: towards integrating biological and landscape diversity for sustainable agriculture in Europe*, June 2000, Paris. STRA-

CO/AGRI (2001) 17. Council of Europe, Strasbourg, France. (available at: http://www.coe.int/t/e/Cultural_Co-operation/Environment/Nature_and_ biological_diversity/ biodiversity/agriculture.asp).

Curry, J.P. (1986). Effects of management on soil decomposers and decomposition processes in grassland. In M.J. Mitchell and J.P. Nakas (Eds.), *Microfloral and Faunal Intearctions in Natural and Managed Ecosystems*, 349-398. Dordrecht, the Netherlands, Martinus Nijhoff.

Curry, J.P. (1994). *Grassland Invertebrates. Ecology, Influence on Soil Fertility and Effects on Plant Growth*. Chapman and Hall, London, 437pp.

Donald, P.F., Pisano, G., Rayment, M.D. and Pain, D.J. (2002). The Common Agricultural Policy, EU enlargement and the conservation of Europe's birds. *Agricultue, Ecosystems and Environment* **89**, 167-182.

Donald, P.F., Green. R.E. and Heath, M.F. (2001). Agricultural intensification and the collapse of Europe's farmland bird populations. *Proceedings of the Royal Society of London (B)* **268**, 25-29.

EEA (1999). *Environmental Indicators: Typology and Overview*. Technical report No. 25, European Environment Agency, Copenhagen, Denmark.
(Available at:http://reports.eea.eu.int/TEC25/en/tab_content_ RLR)

EEA (2003). *An Inventory of Biodiversity Indicators in Europe, 2002*. Technical report No. 92, European Environment Agency, Copenhagen, Denmark.
(Available at: http://reports.eea.eu.int/technical_report_ 2004_92/en/tab_content_RLR).

EEA (2004). *High Nature Value Farmland: Characteristics, trends and policy challenges*. Report No. 1/2004, European Environment Agency, Copenhagen, Denmark.
(Available at: http://reports.eea.eu.int/report_2004_1/en/tab_content_RLR)

EPBRS (2004). *Killarney Declaration and Recommendations on Biodiversity Research*. EPBRS Meeting, Killarney, Ireland, May 2004. European Platform for Biodiversity Research Strategy.
(Available at: http://www.epbrs.org/epbrs_library.html).

European Commission (2000). *Indicators for the Integration of Environmental Concerns into the Common Agricultural Policy*. COM(2000)20 final, European Commission, Brussels.
(Available at: http://europa.eu.int/eurlex/en/com/cnc/2000/com2000_0020en01.pdf).

European Commission (2001). *Biodiversity Action Plan for Agriculture*. COM(2001)162 final vol. III, European Commission, Brussels (available at: http://www.epbrs.org/epbrs_library.html).

Fossitt (2000). *A Guide to the Habitats of Ireland*. Kilkenny, The Heritage Council.

Fuller, R.J., Gregory, R.D., Gibbons, D.W., Marchant, J.H., Wilson, J.D., Baillie, S.R. and Carter, N. (1995). Population declines and range contractions among lowland farmland birds in Britain. *Conservation Biology* **9**, 1425–1441.

Furness, R.W. and Greenwood, J.J.D. (Eds.) (1993). *Birds as Monitors of Environmental Change*. London, Chapman and Hall.

Gibson, D.J., Connolly, J., Hartnett, D.C.and Weidenhamer. J.D. (1999). Essay review: designs for greenhouse studies of interactions between plants. *Journal of Ecology* **87**,1-16.

Godray (1994). *Parasitoids: Behavioural and Evolutionary Ecology*. Princeton University Press.

Hector, A., Schmid B., Beierkuhnlein C., Caldeira, M.C., Diemer, M., Dimitrakopoulos, P.G., Finn J.A., Reitas, P., Giller, P.S., Goode, J., Harris, R., Hogerg, P., Huss-Danell, K., Joshi, J., Jumpponen, A., Mulder, C.K., O'Donovan, G., Otway, S.J., Pereira, S.J., Prinz, A., Read, D.J., Scherer-Lorenzen, M., Schulze, E-D., Siamantziouras, A-S.D., Spehn, E.M., Terry, A.C., Troumbis, A.Y., Woodward, F.I., Yachi, S. and Lawton, J.H. (1999). Plant diversity and productivity experiments in European grasslands. *Science* **286**, 1123–1127.

Hochberg, M.E. and Ives, A.R. (Eds.) (2000). *Parasitoid Population Biology*. Princeton University Press, Princeton, New Jersey, USA.

Holland, J.M. and Luff M.L. (2000). The effects of agricultural practices on Carabidae in temperate agroecosystems. *Integrated Pest Management Reviews* **5**, 109-129.

Hoogerkamp, M., Rogaar, H. and Ejsackers, H.J.P. (1983). Effect of earthworms on grassland on recently reclaimed polder soils in the Netherlands. In J.E. Satchell (Ed.), *Earthworm Ecology – From Darwin to Vermiculture*, 85-104. London, Chapman and Hall.

Jones, T.H. and Bradford, M.A. (2001). Assessing the functional implications of soil biodiversity in ecosystems. *Ecological Research* **16**, 845–858.

Joshi, J., Schmid, B., Caldeira, M.C., Dimitrakopoulos, P.G., Good, J., Harris, R., Hector, A., Huss-Danell, K., Jumpponen, A., Minns, A., Mulder, C.P.H., Pereira, J.S., Prinz, A., Scherer-Lorenzen, M., Siamantziouras, A.S.D., Terry, A.C., Troumbis, A.Y. and Lawton, J.H. (2001). Local adaptation enhances performance of common plant species. *Ecology Letters* **4** (6), 536-544.

Karr J.R. (1991). Biological integrity: a long neglected aspect of water resources management. *Ecological Applications* **1**, 66-84.

Karr J.R. and Chu E.W. (2000) Sustaining living rivers. *Hydrobiologia* **422/423**, 1-14.

Kirwan, L., Bélanger, G., Finn, J., Fothergill, M., Frankow-Lindberg, B., Garcia-Sarrion, R., Ghesquiere, A., Golinski, P., Helgadóttir, A., Jørgensen, M., Kadiuliene, Z., Nyfeler, D., Nykänen-Kurki, P., Parente, G., Vasileva, V., Collins, R., Connolly, J., Lüscher, A., Porqueddu, C. and Sebastià, M.T. (2005). Higher yield and fewer weeds in four-species grass/legume mixtures than in monocultures: results from the first year at 20 sites of COST Action 852. *Proceedings of XX International Grassland Congress*, p.425. Dublin, June 2005.

Kleijn, D., Berendse, F., Smit, R. and Gilissen, N. (2001). Agri-environment schemes do not effectively protect biodiversity in Dutch agricultural landscapes. *Nature* **413**, 723-725.

Knight, M.E., Martin, A.P., Bishop, S., Osborne, J.L., Hale, R.J., Sanderson, A. and Goulson, D. (2005). An interspecific comparison of foraging range and nest density of four bumblebee (*Bombus*) species. *Molecular Ecology* **14**, 1811-1820.

Kromp, B. (1999). Carabid beetles in sustainable agriculture: a review on pest control efficacy, cultivation impacts and enhancement. *Agriculture, Ecosystems and Environment* **74**, 187-228.

Lovei, G.L. and Sunderland, K.D. (1996). Ecology and behaviour of ground beetles (Coleoptera: Carabidae). *Annual Review of Entomology* **41**, 231-256.

Marchant, J.H., Hudson, R., Carter, S.P., Whittington, P.A. (1990). *Population Trends in British Breeding Birds*. British Trust for Ornithology, Thetford, UK.

McGarrigle, M.L., Bowman, J.J., Clabby, K.J., Lucey, J., Cunningham, P., MacCárthaigh, M., Keegan, M., Cantrell, B., Lehane, M., Clenaghan, C. and Toner, P.F. (2002). *Water Quality in Ireland 1998 – 2000.* Dublin, Environmental Protection Agency.

McNabb, D.M., Halaj, J. and Wise, D.H. (2001). Inferring trophic positions of generalist predators and their linkage to the detrital food web in agroecosystems: a stable isotope analysis. *Pedobiologia* **45**, 289-297.

Metcalf, R.L. and Luckmann, W.H. (1994). *Introduction to Insect Pest Management.* Wiley-Interscience, New York, USA.

Minns, A., Finn, J.A., Hector, A., Caldeira, M., Joshi, J., Palmborg, C., Schmid, B., Scherer-Lorenzen, M., Spehn, E., Troumbis, A. and the BIODEPTH project (2001). The functioning of European grassland ecosystems: potential benefits of biodiversity to agriculture. *Outlook on Agriculture* **30**, 179-185.

Muldowney, J., Curry, J.P., O'Keeffe, J. and Schmidt, O. (2003). Relationships between earthworm populations, grassland management and badger densities in County Kilkenny, Ireland. *Pedobiologia* **47**, 913-919.

Newton, S., Donaghy, A., Allen, D. and Gibbons, D. (1999). Birds of conservation concern in Ireland. *Irish Birds* **6**. 333-342.

Nyffeler, M. and Sunderland, K.D. (2003). Composition, abundance and pest control potential of spider communities in agro-ecosystems: a comparison of European and US studies. *Agriculture, Ecosystems and Environment* **95**, 579-612.

OECD (2001). *OECD Experts Meeting on Agri-Biodiversity Indicators (Zurich, November, 2001): Summary and Recommendations*. Organisation for Economic Cooperation and Development. (Available at: http://www1.oecd.org/agr/biodiversity/index. htm).

Ponsard, S., Bethenod, M.T., Bontemps, A., Pelozuelo, L., Souqual, M.C. and Bourguet, D. (2004). Carbon stable isotopes: a tool for studying the mating, oviposition, and spatial distribution of races of European corn borer, *Ostrinia nubilalis*, among host plants in the field. *Canadian Journal of Zoology* **82**, 1177–1185.

Parkinson, D., McLean, M.A. and Scheu, S. (2004). Impacts of earthworms on other soil biota in forest soils, with some emphasis on cool montane forests. In C.A.Edwards (Ed.), *Earthworm Ecology* (2nd Edition), 241-259. Boca Raton, Florida, CRC Press.

Price, P.W. and Waldbauer, G.P. (1994). Ecological aspects of pest management. In R.L. Metcalf and W.H. Luckmann (Eds.), *Introduction to Insect Pest Management*, 35-72. New York, Wiley-Interscience.

Pringle, C. (2003) What is hydrological connectivity and why is it ecologically important. *Hydrological Processes* **17**, 2685-2689.

Proctor, M. and Yeo, P. (1973). *The pollination of flowers.* London, Collins.

Purvis, G. and Curry, J.P. (1981). The influence of sward management on foliage arthropod communities in a ley pasture. *Journal of Applied Ecology* **18**, 711-726.

Purvis, G., Anderson, A., Helden, A. and Kirwan, L. (2005). The Ag-Biota Project: A preliminary assessment of potential indicators of biodiversity in agricultural grasslands. *Proceedings of the XXth International Grassland Congress*, Dublin, June 2005 (in press).

Quicke, D.L.J. (1997). *Parasitic Wasps*. London, Chapman and Hall.

Ramseier, D., Connolly, J. and Bazzaz, F.A. (2005). Carbon dioxide regime, species identity and influence of species initial abundance as determinants of change in stand biomass composition in five-species communities: an investigation using a simplex design and RGRD analysis. *Journal of Ecology* **93**, 502-511.

Reynoldson, T.B. and Metcalfe-Smith, J.L. (1992). An overview of the assessment of aquatic ecosystem health using benthic invertebrates. *Journal of Aquatic Ecosystem Health* **1**, 295-308.

Roots, O. and Talkop, R. (Eds.) (1997). *Estonian environmental monitoring*. Estonian Ministry of Environment, Environment Information Centre, Talinn, Estonia.

Rosenberg, D.M. and Resh, V.H. (Eds.) (1993). *Freshwater biomonitoring and benthic macroinvertebrates*. Chapman and Hall, London and New York.

Schmidt, O. and Scrimgeour, C.M. (2001). A simple urea leaf-feeding method for the production of ^{13}C and ^{15}N labelled plant material. *Plant and Soil* **229**, 197–202.

Schmidt, O., Curry, J.P., Dyckmans, J., Rota, E. and Scrimgeour, C.M. (2004). Dual stable isotope analysis (delta^{13}C and delta^{15}N) of soil invertebrates and their food sources. *Pedobiologia* **48**, 171–180.

Schneider, K., Migge, S., Norton, R.A., Scheu, S., Langel, R., Reineking, A. and Maraun, M. (2004). Trophic niche differentiation in soil microarthropods (Oribatida, Acari): evidence from stable isotope ratios (^{15}N/^{14}N). *Soil Biology and Biochemistry* **36**, 1769–1774.

Schulte R.P.O., Lantinga, E.A. and Struik, P.C. (2003). Analysis of the production stability of mixed grasslands I: a conceptual framework for the qualification of production stability in grassland ecosystems. *Ecological Modelling* **159**, 43-69.

Scrimgeour, C.M. and Robinson, D. (2004). Stable isotope analysis and applications. In K.A. Smith and M.S. Cresser (Eds.), *Soil and Environmental Analysis: Modern Instrumental Technique* (*3rd Edition*), 381–431). New York, Marcel Dekker.

Sepp, K., Mikk, M., Mand, M. and Truu, J. (2004). Bumblebee communities as an indicator for landscape monitoring in the agri-environmental programme. *Landscape and Urban Planning* **67**, 173-183.

Sheridan, H. (2005). *The Experimental Restoration and Enhancement of Grassland Field Margins within the Irish Rural Environmental Protection Scheme (REPS).* Unpublished Ph.D. thesis, University College Dublin.

Siriwardena, G.M., Baillie, S.R., Buckland, S.T., Fewster, R.M., Marchant, J.H. and Wilson, J.D. (1998). Trends in the abundance of farmland birds: a quantitative comparison of smoothed common birds census indices. *Journal Applied Ecology* **35**, 24–43.

Sponseller, R.A., Benfield, E.F. and Valett, H.M. (2001). Relationship between land-use, spatial scale and stream macroinvertebrate communities. *Freshwater Biology* **465**, 1409-1424.

Stockdill, S.M.J. (1982). Effects of introduced earthworms on the productivity of New Zealand pastures. *Pedobiologia* **24**, 29-35.

Sunderland, K.D. (1987). A review of methods of quantifying invertebrate predation occurring in the field. *Acta Phytopathologica et Entomologica Hungarica* **22**, 13-34.

Symondson, (2002). Molecular identification of prey in predator diets. *Molecular Ecology* **11**, 627-641.

Symondson, W.O.C., Sunderland, K.D. and Greenstone, M.H. (2002). Can generalist predators be effective biocontrol agents? *Annual Review of Entomology* **47**, 561-594.

Tilman, D., Wedin, D. and Knops, J. (1996).

Productivity and sustainability influenced by biodiversity in grassland ecosystems. *Nature* **379**, 718-720.

UNEP (1992). *Convention on Biological Diversity*. United Nations Environment Programme. (Available at: http://www.biodiv.org/doc/legal/cbd-en.pdf).

UNEP (1997). *Recommendations for a core of indicators of biological diversity*. UNEP/CBD/SBSTTA/5/12, Subsidiary body on scientific, technical and technological advice, United Nations Environment Programme.
(Available at: http://www.biodiv.org/doc/meetings/sbstta/sbstta-03/official/sbstta-03-09-en.pdf).

UNEP (1999). *Development of indicators of biological diversity*. UNEP/CBD/SBSTTA/5/12. United Nations Environment Programme, subsidiary body on scientific, technical and technological advice.
(Available at: http://www.biodiv.org/doc/meetings/sbstta/sbstta-05/official/sbstta-05-12-en.pdf).

Van Dreische, R.G. and Bellows, T.S. (1996). *Biological Control*, Chapter 7 (103-127). New York, Chapman and Hall.

Vimmerstedt, J.P. (1983). Earthworm ecology in reclaimed opencast mining sites in Ohio. In: J.E. Satchell (Ed.) *Earthworm Ecology – From Darwin to Vermiculture*, 229-240. London, Chapman and Hall.

Waage, J.K. (1990). Ecological theory and the selection of biological control agents. In M. Mackauer and L.E. Ehler. (Eds.), *Critical Issues in Biological Control*. Andover, UK, Intercept.

Waage, J.K. and Greathead, D. (Eds.) (1986). *Insect Parasitoids*. London, Academic Press.

Waldhardt. R., Simmering, D. and Albrecht, H. (2003). Floristic diversity at the habitat scale in agricultural landscapes of Central Europe – summary, conclusions and perspectives. In W. Buchs (Ed.), *Biotic Indicators for Biodiversity and Sustainable Agriculture*; *Agriculture, Ecosystems and Environment* **98**, 79-85. Amsterdam, Elsevier.

Wharton (1993). Bionomics of the Braconidae. *Annual Review of Entomology* **38**, 121-143.

Wolters, V. (2001). Biodiversity of soil animals and its function. *European Journal of Soil Biology* **37**, 221-227.

Wright, J.F., Sutcliffe, D.W. and Furse, M.T. (Eds.) (2000). *Assessing the biological quality of fresh waters: RIVPACS and other techniques*. Freshwater Biological Association, Ambleside, UK.

Zhu, Y., Chen, H., Fan, J., Wang, Y., Li, Y., Chen, J., Fan, J., Yang, S., Hu, L., Leung, H., Mew, T., Teng, P.S., Wang Z. and Mundt, C.C. (2000). Genetic diversity and disease control in rice. *Nature* **406**, 718-722.