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RIPARIAN BUFFER STRIPS AS A MULTIFUNCTIONAL TOOL IN AGRICULTURAL LANDSCAPES

Investigating Riparian Margins for Vegetation Patterns and Plant–Environment Relationships in Northeast Scotland

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Buffer strips alongside watercourses are a widely accepted method of reducing nutrient and sediment run-off from agricultural land thereby improving water quality. Little attention, however, has been paid to the ecological status of these areas despite the fact that riparian habitats in good condition can provide multiple benefits. We investigated vegetation patterns and plant–environment relationships within three categories of riparian margins in northeast Scotland. The margins were categorized as unbuffered, buffered, or reference (target), the latter representing the best sites available within the catchments. Vascular plant and soil data were collected from 41 sites along the tributaries of two rivers during 2008 and 2009. Ellenberg indicator values revealed trends of decreasing light availability ($P < 0.05$) and decreasing pH ($P < 0.01$) from unbuffered sites to buffered sites to reference sites. Multivariate analysis showed that soil parameters and channel morphology, together with canopy cover and bryophyte abundance, were discriminatory in separating species assemblages. The presence of a tree canopy layer appears to be the key instigator of change in soil conditions and corresponding plant species assemblages. An understanding of the underlying processes is important if vegetation characteristics are to be used effectively as indicators of riparian and water quality and to aid the restoration of riparian habitats.

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Riparian ecotones occupy a key aspect of many landscapes, providing the interface between terrestrial and aquatic ecosystems. Consequently, the management of riparian areas in agricultural regions has been considered a key factor in protecting surface water quality and aquatic ecosystems (Sweeney and Czapka, 2004; Ma et al., 2002; Paine and Ribic, 2002) and as a cost-effective method of flood mitigation (Decamps et al., 2001). Guidelines for the improvement of water bodies have focused on the establishment of riparian buffer strips (Paine and Ribic, 2002). These are defined as uncultivated vegetated areas between agricultural fields and watercourses whose primary function is to reduce the impact of land use activities by intercepting or processing sediment and pollutants before they enter a stream or river, thereby helping to protect aquatic systems (Borin et al., 2010; Stutter et al., 2010; Correll, 2005; Syversen, 2005; Sweeney and Czapka, 2004; Tattari et al., 2003; Ma et al., 2002; Paine and Ribic, 2002).

Less attention has been given to the value and significance of riparian zones in their ability to function with increasing intensification of land use. Vegetation and vegetation type are of key importance in the structure and function of riparian zones through altering the physical and chemical environment and through interactions at every trophic level (Sweeney and Czapka, 2004). Riparian zones in their natural state should be dynamic and species rich, reflecting the physical character of the landscape, local hydrology, and climate (Paine and Ribic, 2002). Such complex and dynamic mosaics of vegetation types and physical features have become increasingly rare and threatened from intensification in their use (Alexander et al., 2010). For example, forest has been replaced with grasses and shrubs, and communities have been further degraded by invasive non-native species such as Himalayan balsam (*Impatiens glandulifera* Royle) (Bowers and Boutin, 2008; Hulme and Bremner, 2006; Tabacchi and Planty-Tabacchi, 2005; Paine and Ribic, 2002). Extensive removal or alteration of riparian vegetation has a negative effect on habitat stability and increases the impact of land use activities, energy flow, and contaminants on the aquatic environment.

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Abbreviations: CCA, canonical correspondence analysis; Ell-F, Ellenberg moisture; Ell-L, Ellenberg light; Ell-N, Ellenberg nitrogen; Ell-R, Ellenberg reaction.

Riparian vegetation plays a number of key roles. It is important in flood management by slowing the water and increasing the water holding capacity throughout the catchment. Trees have higher evaporation rates than other plants, and their roots provide deeper drainage, reducing the water table and mitigating the effects of flood waters. Woody vegetation traps nitrates and provides leaf litter and dissolved organic matter for desirable populations of terrestrial and aquatic invertebrates and microorganisms, which in turn support fish populations (Gregory et al., 1991). Coarse woody debris and tree roots provide valuable habitat for invertebrates and fish (Harmon et al., 1986). Grasses trap suspended particulates and phosphorus (Gregory et al., 1991). Riparian vegetation, especially forest, plays an important role in maintaining lower temperatures in streams (Sinokrot and Stefan, 1993). The cooling effects of shading and evaporation are often essential to the development and survival of species of fish (Correll, 2005). Vertical structure within the riparian zone provides habitat for terrestrial invertebrates, birds, and bats (Dennis et al., 2006).

A key instigator of change in the management of riparian areas in Europe has been the Water Framework Directive (Directive 2000/60/EC [European Commission, 2000]). This requires those responsible to achieve "good ecological status" of all inland waters by 2015 and to prevent further deterioration. In Scotland, where this is infeasible or disproportionately expensive, improvements will be phased to achieve the overall aim of all waters to be in a good condition by 2027 (Scottish Environmental Protection Agency (SEPA), personal communication). Restoration of riparian woodlands is recognized as being important for flood management under the Flood Risk Management (Scotland) Act 2009. The threats to riparian woodland and specialist riparian species have been recognized in additional international legislation, including the Habitats Directive (Directive 92/43/EEC [European Commission, 1992]), and by nonstatutory initiatives at the local level, such as the Local Biodiversity Action Plan in northeast Scotland.

In the present study, we compared species diversity and assemblage patterns of ground-flora communities along streams within agricultural landscapes. Our objectives were (i) to examine the response of various structural components, including diversity parameters to riparian margin type; (ii) to identify major environmental gradients operating in riparian areas; and (iii) to establish their influence.

Materials and Methods

Study Catchments

This study was conducted in July and August 2008 and 2009 in riparian areas along 41 sites within two contrasting river catchments in northeast Scotland. The Ugie catchment (57°53′ N, 2°00′ W) covers 155 km^2 and flows into the sea at the town of Peterhead. According to the Scottish Environmental Protection Agency (SEPA), the water quality in this catchment is classed as poor, failing to reach good status because of nutrient (nitrogen) and sediment characteristics (E. Johnston, personal communication). The catchment is predominantly mixed lowland farmland and contains very little seminatural vegetation. The Tarland Burn catchment (57°13′ N, 2°86′ W) is a subcatchment of the river Dee, which flows into the sea at the city of Aberdeen. Covering an area of 70 km^2, the land use is a mixture of upland mixed farming, conifer plantation, and around 10% seminatural habitats. Part of the catchment falls within the River Dee Special Area of Conservation. Here the water quality reaches moderate status with remaining high levels of nutrients (phosphorus) and hydromorphology characteristics, including extensive channelization (SEPA, 2010).

The goal in site selection was to locate sections representative of different types of riparian margins: unbuffered, buffered and reference. Unbuffered sites were defined as those with no permanent fencing to prevent livestock accessing the watercourse. These sites were subject to disturbance through periodic cutting, human recreational activities, and erosion or by agricultural practices such as grazing, plowing, and cattle waterings. Since 1999, many riparian areas within both catchments have been buffered with the creation of fenced-off strips of at least 2 m width, many of which have also been planted with trees. Buffer strip ages ranged between 1 and 10 yr (average, 7 yr). Reference sites were characterized by mature, woody riparian vegetation where agricultural activity had not taken place in the recent past. These sites were deemed to be the best available within the catchments and to act as restoration target sites. A total of 27 sites were located within the Tarland catchment, and 14 sites were located within the Ugie catchment. To ensure that the geographical settings were comparable, the sites within each category of riparian margin were set within areas of similar land use and widely dispersed throughout the catchments.

Vegetation Sampling

Within each site, a section of 50 m length was selected for intensive study. Within this intensive monitoring location, vegetation data were split into three groups: tree (canopy cover), forbs and grasses (including grasses, sedges, rushes, and horsetails), and bryophytes (mosses and liverworts). The latter group was assessed collectively and not identified to species level. To ensure comparability across sites, only vegetation within 2 m of the water's edge was sampled. The tree layer included all tree species with stems >1 cm diameter at breast height. Trees were counted and identified to species, and the canopy cover was scored categorically from 0 to 4 (0 = no canopy cover, 1 = 0–25% cover, 2 = 25–50% cover, 3 = 50–75% cover, 4 = 75–100% cover). Understorey vegetation was quantified using 1 × 1 m quadrats (n = 10 per site) placed 5 m apart. Quadrats were positioned at random distances between 0 and 2 m from the waters' edge to ensure good coverage of the riparian habitat while retaining comparability across sites. Within each quadrat, all vascular plants were identified to species, and their abundance was calculated as the percent cover. Species frequency at each site was calculated from the number of quadrats in which the species occurred. The percentage cover of litter (dead plant material) was recorded. To assess vegetation structure, vegetation density was calculated as its inverse (i.e., the proportion of bare ground). Sward height was calculated from five measurements of the tallest vegetation in the center and in the four corners of each quadrat.

Site Physical Attributes

An array of physical and environmental parameters were assessed for each quadrat, including slope, aspect, altitude, stream width, adjacent land use, and distance from source of tributary (Table 1). Slope was derived from the difference in bank height at the top of the dyke from that at water level. For buffered and reference sites, the length and width of the fenced buffer, or undisturbed strip, and the length was measured. The age of the buffered margins was determined in consultation with farmers and landowners. Water channels were categorized as meandering, either created or naturally occurring, or channelized. Flooding frequency was estimated from visual assessment and past experience and, at one Tarland site, from an automated monitoring station.

Soil samples were collected for surface soil chemistry and moisture content, and water samples were collected for water chemistry. Soil samples were collected to a depth of approximately 7 cm from the center of each quadrat. Samples were kept field-moist, sieved to 2 mm, and stored at 4°C before analysis. Moisture content was calculated as the difference between the weight of a known quantity of soil before and after oven drying. Potassium chloride (KCl) extracts were performed on field-moist soil and analyzed for nitrogen (as NO_3–N and NH_4–N). Acetic acid extracts were performed on air-dried soil and analyzed for phosphate (as PO_4–P). Water samples were collected throughout the summer and were filtered and analyzed via a Skalar autoanalyzer (Skalar, Breda, The Netherlands) within 3 d of collection. Samples were analyzed for N (total N, organic N, NO_3–N, and NH_4–N), P (as total P, organic P, and PO_4–P), dissolved organic carbon, electrical conductivity, and pH.

Data Analysis

All univariate analyses were performed in Genstat (Genstat 12th edition; Lawes Agricultural Trust, 2009). Preliminary analysis using general linear models (Nelder and Wedderburn, 1972) showed variation within sites not to be a significant factor, and therefore all 10 quadrats for each site were pooled, and further analyses were performed on mean abundance per quadrat. Data were analyzed in terms of three main attri-

Table 1. Environmental variables sampled at each site.

Variable	Abbreviation	Description and units
Bryophyte abundance	–	percent cover bryophytes
Canopy cover	–	percent canopy cover categorical: 0 = 0%, 1 = 1– 25%, 2 = 26–50%, 3 = 51–75%, 4 = 76–100%
Litter	–	percent litter
Vegetation density	–	percent bare ground
Vegetation height	–	sward height, cm
Slope	–	derived slope, degrees
Aspect	–	derived aspect, degrees
Altitude	–	site altitude above sea level, m
Stream width	–	stream width, m
Distance from source	–	distance of site from tributary source, km
Land use	–	adjacent land use, categorical: 1 = pasture, 2 = arable, 3 = woodland
Buffer width	–	width of buffer strip, m
Buffer length	–	length of buffer strip, km
Buffer age	–	age of buffer strip, yr
Flooding	–	estimated frequency of flooding, categorical: 0 = never, 1 = rarely, 2 = occasionally, 3 = frequently
Channel	–	channel path, categorical: 1 = channelized, 2 = artificially created meanders, 3 = natural meanders
Soil moisture	–	percent moisture
Soil ammonium	NH_4–N	mg kg^{-1}
Soil nitrates	NO_3–N	mg kg^{-1}
Soil phosphates	PO_4–P	mg kg^{-1}
Soil pH	–	
Water ammonium	NH_4–N	mg kg^{-1}
Water nitrates	NO_3–N	mg kg^{-1}
Water total nitrogen	Tot-N	mg kg^{-1}
Water organic nitrogen	Org-N	mg kg^{-1}
Water phosphate	PO_4–P	mg kg^{-1}
Water total phosphate	Tot-P	mg kg^{-1}
Water organic phosphate	Org-P	mg kg^{-1}
Water dissolved organic carbon	DOC	mg kg^{-1}
Water conductivity	–	electrical conductivity, $\mu S\ cm^{-1}$
Water pH	–	
Ellenberg nitrogen	Ell-N	derived indicator value for soil nitrogen
Ellenberg moisture	Ell-F	derived indicator value for soil moisture
Ellenberg reaction	Ell-R	derived indicator value for soil pH
Ellenberg light	Ell-L	derived indicator value for light

butes: (i) species diversity, (ii) functional type (e.g., riparian and nonriparian) richness and abundance, and (iii) assemblage structure. The diversity measures used were the Shannon and Simpson indexes.

To obtain a more comprehensive and integrated description of the environmental conditions, weighted averages of Ellenberg's indicator values for nutrient availability (Ell-N), moisture (Ell-F), pH (Ell-R), and light (Ell-L) were calculated for each site (Hill et al., 1999). These values are based on an assignment of indicator values for specific environmental conditions to a large number of species (>3000). Weighted averages can therefore be used to interpret environmental gradients from assemblage patterns. Numerous publications have demonstrated that these values are applicable and useful in ecological studies (e.g., Ertsen et al., 1998). Weighted Ellenberg values were calculated as the mean of indicator values weighted with the cover of each species present within the site and were the values used for all further analyses. Comparisons between riparian margin types were made by Kruskal-Wallis one-way ANOVA. Subsequent Mann-Whitney *U* tests were used to identify which riparian margin types differed from one another.

Species with Ell-F indicator values >6 were considered riparian, and species with moisture values <6 were considered nonriparian (Baattrup-Pedersen et al., 2005). Species with a score of exactly 6 were considered indifferent and were not investigated further. Individual species and riparian species abundance comparisons were made by one-way ANOVA. Vegetation characteristics relating to richness and diversity were compared by Kruskal-Wallis one-way ANOVA. Subsequent Mann-Whitney *U* tests were used to identify significantly different riparian margin types. Within the Tarland catchment, where there was a wide range of different-age buffer strips, a general linear model was used to compare the length of time a site had been buffered with the plant species richness of those sites.

The data matrices were subjected to canonical correspondence analysis (CCA) using CANOCO 4.0 (ter Braak and Šmilauer, 2002), which is an eigenvalue ordination method that directly relates multivariate ecological data matrices. Ordinations were performed using plant species abundance and frequency. Very little difference was observed between the analyses of abundance and frequency, so only abundance ordinations are presented here. Stream width, distance from source, and altitude were found to be highly correlated, and therefore only stream width was used for further analysis. Percentage cover of bare ground and bryophytes were log-transformed to normalize the data. Year was entered as a covariable. For all the explanatory variables considered, the best predictors were selected by a forward selection procedure, which is a multivariate extension of the stepwise regression method. The significance of the effects analyzed was tested using a Monte Carlo permutation test with 499 permutations, and additions were halted when no further variables added significantly to the explanatory power of the model ($P < 0.05$). Differences between catchments and margin types were investigated by Kruskal-Wallis one-way ANOVA of the CCA axis scores.

Results

A total of 89 species of understorey plants were recorded. Eight could not be identified to species and were therefore eliminated from further analysis. These species occurred irregularly, covering no more than 4% of any one site, and several were assumed to be garden escapees. Of the species analyzed, 21 were grasses and 60 were vascular plants. Neither grasses nor forbs showed significantly greater abundance at any one type of riparian margin, although the dominant species were generally grasses, such as *Phragmites australis*, *Holcus lanatus*, and *Holcus mollis* (Table 2). Few species were found exclusively within any one type of riparian margin. The assemblage structure was a mixture of weedy agricultural species, riparian, woodland, and upland plants.

Sites varied considerably in understorey species richness ranging from just four species to 35 species, with an average of 16 per 10 m^2. The number of species per quadrat ranged from

Table 2. Mean relative abundance of dominant plant species.

Species	Unbuffered	Buffered	Reference
Juncus effusus L.	0.85 (0.30)†	0.52 (0.18)	3.88 (2.51)
Deschampsia cespitosa (L.) P. Beauv.	2.14 (0.78)	2.34 (0.88)	3.03 (1.21)
Holcus mollis L.	5.58 (1.74)	6.98 (1.58)	5.15 (2.56)
Festuca rubra L.	0.88 (0.41)	2.33 (0.93)	1.27 (0.79)
Agrostis stolonifera L.	2.65 (1.79)	2.36 (1.26)	0.84 (0.60)
Dactylis glomerata L.	2.62 (0.85)	2.72 (0.65)	1.34 (1.09)
Phragmites australis (Cav.) Trin. ex Steud.	10.40 (4.03)	8.78 (2.26)	8.94 (3.61)
Filipendula ulmaria (L.) Maxim.	8.48 (2.84)	10.67 (2.50)	0.30 (0.12)
Arrhenatherum elatius (L.) P. Beauv. ex J. Presl & C. Presl	3.22 (1.12)	2.54 (0.65)	0.28 (0.18)
Holcus lanatus L.	11.46 (2.87)	5.52 (1.90)	9.67 (4.35)
Poa trivialis L.	2.51 (1.36)	0.50 (0.28)	0.31 (0.22)
Rumex obtusifolius L.	1.50 (0.39)	1.67 (0.44)	0.19 (0.07)
Galium aparine L.	1.39 (0.49)	1.60 (0.40)	0.62 (0.24)
Ranunculus repens L.	4.35 (1.45)	2.06 (0.52)	2.84 (1.45)
Urtica dioica L.	3.54 (0.92)	7.47 (1.11)	7.75 (1.70)
Cirsium arvense (L.) Scop.	1.27 (0.43)	1.23 (0.26)	1.04 (0.33)
Anthriscus sylvestris (L.) Hoffm.	0.76 (0.32)	2.10 (0.54)	1.44 (0.63)

† Values are means (±1 SD of the mean).

1 to 13 with an average of six. Buffering had no effect on plant species richness within the Ugie catchment after 10 yr but did have a significant effect on plant species richness within the Tarland catchment (Fig. 1) because richness declined with the age of the buffer strip (F = 15.08; df, 39; P < 0.001).

For specialist riparian species (those with Ell-F > 6), only abundance changed significantly across the three riparian categories (Table 3). Abundance was found to be higher at unbuffered sites compared with buffered and reference sites. Bryophyte abundance increased significantly from unbuffered to buffered to reference sites. Fifteen tree species were recorded across all sites; alder (*Alnus* sp.) dominated, although birch (*Betula* sp.) and sycamore (*Acer* sp.) were also common. Tree number and species richness were significantly higher at reference sites compared with buffered sites, which were higher than unbuffered sites. Canopy cover increased significantly from unbuffered sites to buffered sites to reference sites.

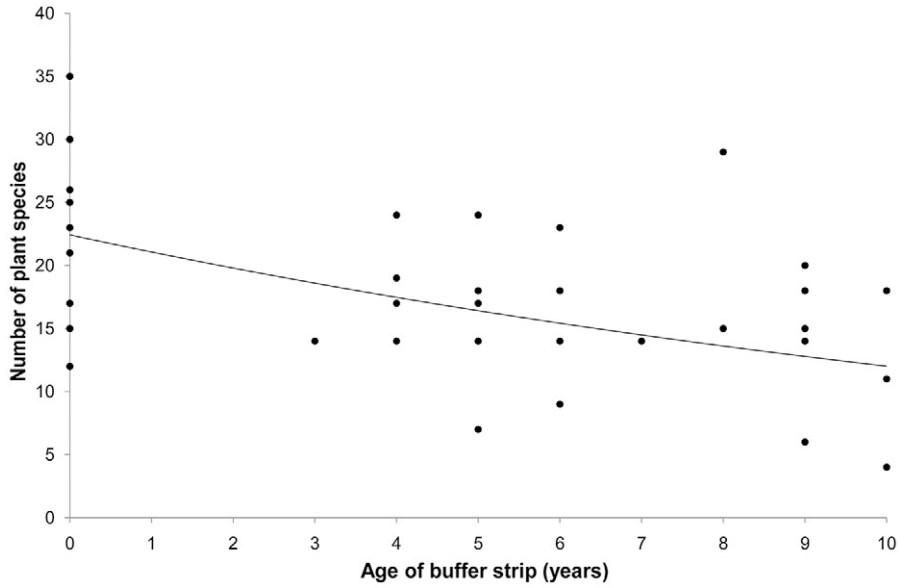

Fig. 1. Comparison of plant species richness and age of riparian buffer strip within the Tarland catchment (y = −1.0398x + 23.245; R^2 = 0.3424). Points represent individual sites.

Ellenberg indicator values were generally found to be highest in unbuffered sites and lowest in reference sites, with buffered sites showing intermediate values (Fig. 2). Ell-L was significantly different (H = 11.69; P = 0.003), and the Mann-Whitney U test showed differences between all three riparian margin types (P < 0.05). Values for Ell-R were also significantly different (H = 10.44; P = 0.005). The Mann-Whitney U test showed a significant difference only between reference sites and unbuffered sites and buffered sites. This agreed with soil pH data because this measure was significantly lower at reference sites than at unbuffered and buffered sites (Table 1). Ell-F was not significantly different across the three types of riparian margin, although the actual soil moisture was significantly higher in buffered compared with reference sites (Table 1). Values for Ell-N were also not significantly different, although

Table 3. Characterization of biotic and abiotic variables for each riparian margin type (n = 41). Canopy is a categorical measure from 0 = no canopy cover to 4 = 75–100% canopy cover.

	Unbuffered	Buffered	Reference	*P* value
Biotic variables				
Bryophyte cover	0.71 (0.32)a†‡	1.73 (0.51)b	3.99 (1.05)c	0.006
Total species richness	17.58 (1.64)	16.29 (0.87)	13.81 (0.90)	ns
Shannon diversity	1.68 (0.15)	1.75 (0.08)	1.60 (0.06)	ns
Simpson diversity	0.68 (0.05)	0.74 (0.02)	0.71 (0.02)	ns
Non-native abundance	0.95 (0.37)	2.16 (0.76)	3.12 (1.66)	ns
Riparian abundance	35.15 (3.90)a	23.28 (2.28)b	17.83 (3.97)b	0.003
Riparian richness	5.71 (0.45)	4.69 (0.24)	4.31 (0.36)	ns
No. of trees	0.58 (0.29)a	2.29 (4.83)b	17.62 (4.83)c	<0.001
Tree species richness	0.50 (0.23)a	1.05 (0.20)b	2.50 (0.34)c	<0.001
Canopy cover	0.25 (0.09)a	0.76 (0.10)b	2.75 (0.11)c	<0.001
Abiotic variables				
Soil moisture	34.67 (1.72)	35.18 (1.18)	30.72 (1.51)	ns
Soil pH	4.97 (0.06)a	4.70 (0.06)b	4.37 (0.19)c	<0.001
Soil PO_4–P	9.08 (1.54)	11.56 (1.48)	11.35 (1.41)	ns
Soil NH_4–N	5.16 (0.78)a	7.04 (0.66)b	9.09 (1.06)c	0.016
Soil NO_3–N	21.56 (2.53)	20.41 (1.95)	25.18 (4.29)	ns
Ell-N§	6.05 (0.09)	5.87 (0.09)	5.55 (0.23)	ns
Ell-F	6.69 (0.20)	6.38 (0.10)	6.21 (0.09)	ns
Ell-R	6.31 (0.07)a	6.14 (0.08)a	5.65 (0.17)b	0.005
Ell-L	6.71 (0.03)a	6.64 (0.03)a	6.40 (0.10)b	0.003

† Values are means (±1 SD).

‡ Significant differences between the three types of riparian margin are indicated by different letters.

§ Ell-F, Ellenberg moisture; Ell-L, Ellenberg light; Ell-N, Ellenberg nitrogen; Ell-R, Ellenberg reaction.

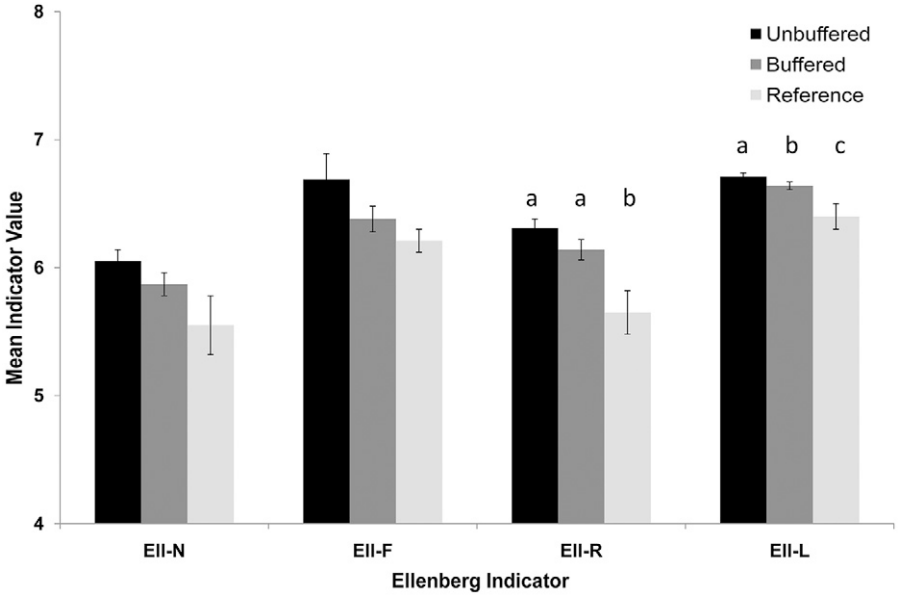

Fig. 2. Weighted average Ellenberg values for unbuffered, buffered, and reference sites. Error bars are ±1 SD of the mean. Letters a and b indicate significant differences between the riparian margin types ($p < 0.05$). Ell-F, Ellenberg moisture; Ell-L = Ellenberg light; Ell-N, Ellenberg nitrogen; Ell-R, Ellenberg soil reaction/pH.

soil NH_4–N significantly increased from unbuffered to buffered to reference sites.

Canonical correspondence analysis of the vegetation data, including environmental parameters and including year as a covariable, found eigenvalues of 0.478 and 0.283 for axes 1 and 2, respectively (Fig. 3 and 4). These two axes explained 7.0 and 4.2%, respectively, of the total variance in the dataset. The first canonical axis was significant (F = 4.250; P = 0.002). Of the 33 variables initially considered in the CCA, only eight were retained in the analysis by forward selection procedure (Table 4). Axis 1 is associated with decreasing human disturbance and is strongly correlated with increasing tree cover and soil phosphorus and decreasing Ell-F, Ell-R, and Ell-N. Axis 2 correlates with channel morphology and bryophyte abundance and is associated with natural disturbance. The plot shows a significant difference in plant species composition between the two catchments for axis 1 values (H = 38.68; P < 0.001) and axis 2 values (H = 5.693; P = 0.017); Ugie sites have low axis 1 but high axis 2 scores, whereas Tarland sites have high axis 1 but low axis 2 scores. Within Tarland, reference sites were significantly different from unbuffered and buffered sites with respect to axis 1 (H = 6.538; P = 0.038) and axis 2 (H = 16.07; P < 0.001).

Discussion

Plant Assemblages and Environmental Variables

In the riparian margins studied, the greatest differences exist between reference and other types of riparian margin. Given that some buffer strips had been in place for up to 10 yr, this finding suggests that human disturbance is not the most important factor in shaping these margins. The single most striking disparity between these sites was the presence of mature trees, which may be a key driver of changes in microclimate, soil chemistry attributes, and hydrology. These changes in site conditions will in turn influence vascular plant species assemblages. The potential role of trees as riparian engineers has been advocated by Gurnell and Petts (2002, 2006), and it is possible that reference conditions are only achieved when the trees reach maturity and the canopy layer is more developed, as has been suggested for in-stream biota (Parkyn et al., 2003). Trees influence the hydrology in many ways, including increasing evaporation and creating, via their roots, deeper drainage, thus reducing the ground water table depth. This is a plausible reason for the decrease in Ell-F and decline in abundance of riparian species toward reference conditions. The observed significant difference in Ell-L and Ell-R values at reference sites compared with other types of riparian margin provides further evidence of the influence of mature trees. The negative correlation between light and tree cover is obvious. With increasing shade, light-sensitive species are shaded out, and species such as *Urtica dioica* become more abundant. Soil pH is one of the most important soil properties because it influences nutrient uptake and plant growth (Finzi et al., 1998). Many trees, including alder (Compton and Cole, 1998), affect soil pH through various mechanisms, including nutrient input, output, and recycling (Binkley and Giardina, 1998).

In other respects, sites with established buffer strips were found to hold intermediate values to those of unbuffered and reference riparian margins. Increasing bryophyte abundance was an expected finding as sites become less disturbed (Frego, 2007). The interactions between bryophytes and vascular plants are not well understood but are likely to include resource competition, effects on regeneration, and alteration of the microclimate. Bryophytes have been found to directly suppress seedling establishment of vascular plants (Soudzilovskaia et al., 2011). Channelization has previously been reported as influencing riparian plant assemblages (Baattrup-Pedersen et al., 2005). Channelization reduces natural flow dynamics and the diversity of riparian habitats available.

Our results showed marked geographic variation between the two river catchments studied. It is no surprise, given the already publicized failings of these particular river catchments (SEPA, 2010), to see strong influences of N and P reflected in the vascular plant species assemblages. The variation between catchments suggests that some variables are operating at the landscape scale. Within the catchments studied, the variation reflects differences in catchment size, nutrient status, habitat diversity, and land use intensity. Plant species composition of nonriparian field margins has been determined by landscape scale variation in environmental attributes (Le Coeur et al., 1997).

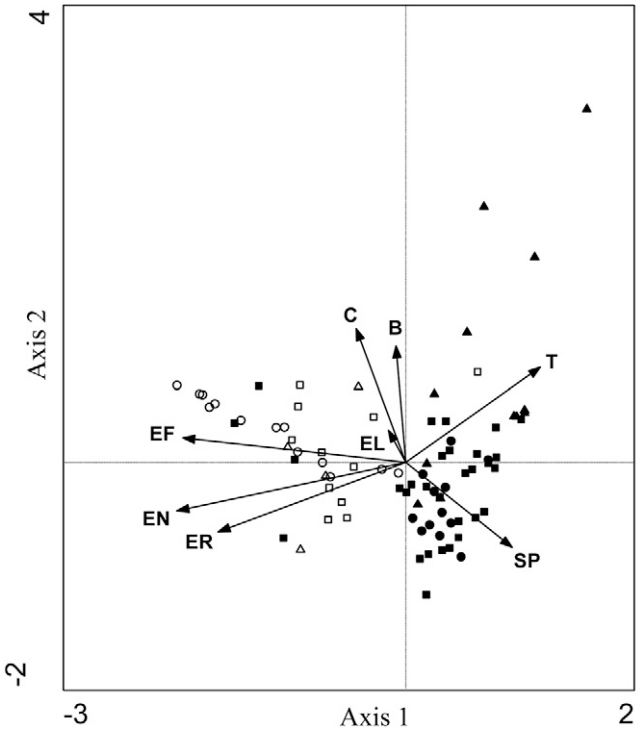

Fig. 3. Canonical correspondence analysis ordination for the three types of riparian margins and significant environmental variables. Unbuffered sites are shown as circles, buffered sites as squares, and reference sites as triangles. Closed symbols denote Tarland catchment sites; open symbols denote Ugie catchment sites. B, bryophyte abundance; C, channel morphology; EF, Ellenberg moisture; EL, Ellenberg light; EN, Ellenberg nitrogen; ER, Ellenberg reaction; SP, soil phosphorus; T, canopy cover.

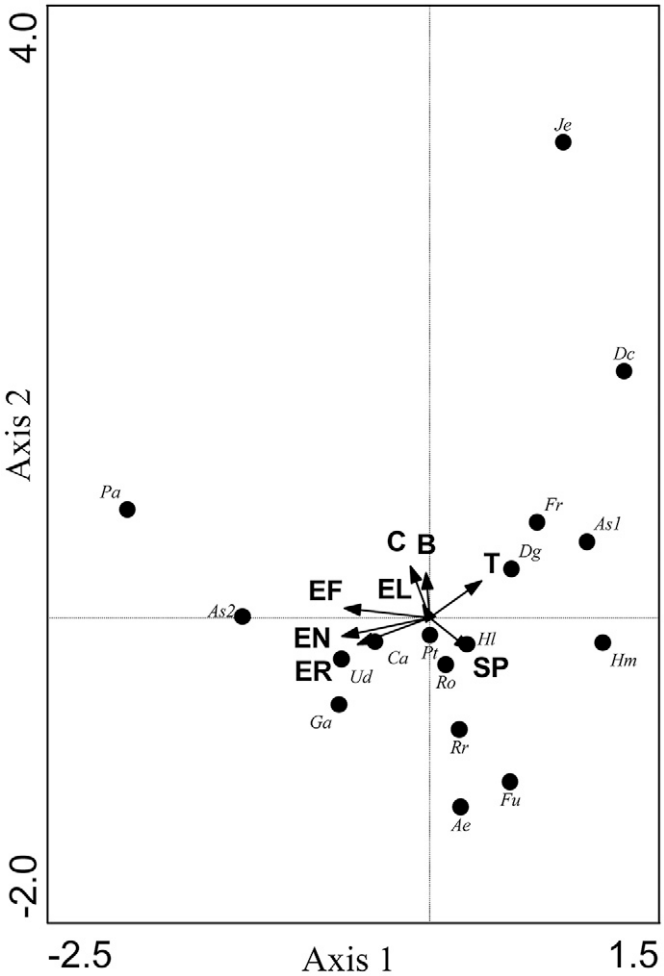

Fig. 4. Canonical correspondence analysis ordination for vascular plant species and significant environmental variables. Only those species with weighting >10% are shown. Variable abbreviations: B, bryophyte abundance; C, channel morphology; EF, Ellenberg moisture; EL, Ellenberg light; EN, Ellenberg nitrogen; ER, Ellenberg reaction; SP, soil phosphorus; T, canopy cover. Species abbreviations: Ae, *Arrhenatherum elatius*; As1, *Agrostis stolonifera*; As2, *Anthriscus sylvestris*; Ca, *Cirsium arvense*; Dc, *Deschampsia cespitosa*; Dg, *Dactylis glomerata*; Fr, *Festuca rubra*; Fu, *Filipendula ulmaria*; Ga, *Galium aparine*; Je, *Juncus effuses*; Hl, *Holcus lanatus*; Hm, *Holcus mollis*; Pa, *Phragmites australis*; Pt, *Poa trivialis*; Ro, *Rumex obtusifolios*; Rr, *Ranunculus repens*; Ud, *Urtica dioca*.

Limitations of Buffer Strips

Responses of riparian plant communities may also be difficult to predict because of the complex nature of the ecosystem (Baattrup-Pedersen et al., 2005). In particular, total species richness has been considered by others (e.g., Bower and Boutin, 2008) to be an unreliable predictor along stream bank sites because of considerable variation. In the current study, we observed a temporal decline in the species richness of plants alongside riparian margins in relation to buffer strip age. This finding is in agreement with that of a previous study (Lyon and Gross, 2005) and with general ecological theories that state that there is a fall-off in species richness as biomass increases (Grime, 1974) or as disturbance declines (Huston and Smith, 1987). Lyon and Gross (2005) warned that it cannot be assumed that protecting more pristine systems will be positively correlated with protecting higher levels of plant diversity.

More is known about the general water quality functions of riparian buffers than we know about how to restore buffers or how quickly and effectively they regain their functions after restoration (Correll, 2005). Anthropogenic modification and disturbance are pervasive factors in an agricultural landscape and are likely to influence the plant species composition of all natural habitats therein. Corbacho et al. (2003) found that disturbance extended to all study sites along riparian habitats in an agricultural landscape and that natural assemblages are rarely encountered (Bowers and Boutin, 2008). Rather than focusing on species per se, further work should be directed at ecosystem functioning and the reconnection of terrestrial–aquatic linkages.

With some of our sites being buffered for up to 10 yr, this suggests the process of restoring these areas is a long-term one. If this is the case, perhaps a further goal is to identify ways this process can be accelerated. For example, plants are considerably less mobile than vertebrates or invertebrates and perhaps take longer to respond to changing conditions (Fritch et al., 2011). It is clear that longer-term monitoring of riparian buffer strips is needed in addition to investigating management options that may help speed up the restoration process. It is too early in the restoration of the sites studied to comment on the role of hysteresis and thus whether the plant community will regain the same composition as the reference sites or establish a different structure reflecting its past degradation.

Table 4. Explanatory variables of plant assemblages identified by a forward selection procedure within a canonical correspondence analysis.

Variable	Forward selection		Cumulative % explained
	F ratio	*P* value	
Ell-F†	4.779	0.002	11.39
Canopy cover	2.677	0.002	17.63
Ell-L	2.413	0.002	23.17
Soil PO_4–P	2.345	0.002	28.41
Bryophyte cover	2.235	0.002	33.42
Ell-N	2.154	0.002	38.11
Ell-R	2.252	0.002	42.94
Channel	1.771	0.002	46.72

† Ell-F, Ellenberg moisture; Ell-L, Ellenberg light; Ell-N, Ellenberg nitrogen; Ell-R, Ellenberg reaction.

Implications for Management in Northeast Scotland

Management options that involve minimal change as currently adopted in many agri-environment systems (e.g., fencing) do not produce swards of conservation value (Fritch et al., 2011). At the site scale, planting and subsequent management of trees within riparian margins will help to speed up the process of change rather than relying on natural regeneration alone. Additionally, restoration measures that have included improvements to the hydromorphology through recreating meanders and floodplains have often met with more success (Jähnig et al., 2009). However, as our results have demonstrated, site management alone is not sufficient to induce desirable change, and a more holistic, catchment-based approach is recommended. In particular, nutrient budgeting is likely to be more successful if implemented at the landscape scale.

There is considerable doubt that the aim of attaining "good ecological status" will be achievable by 2015 if actions do not increase significantly in size, frequency, and comprehensiveness (Bjerring et al., 2008). Riparian organisms, such as floodplain vegetation, are not considered by the Water Framework Directive, yet limited studies have shown these organisms to respond more rapidly to restoration than benthic invertebrates (Jähnig et al., 2009). Awareness of the range of ecosystem services that restoration can provide to society and the positive effect of river restoration in the context of legal frameworks is crucial to increase public support for the restoration of riparian areas, which can lead to longer-lasting, larger-scale measures.

Summary

The results of this study showed marked geographic variation between the vegetation assemblages in the two catchments. Within each catchment, there was still more of a tendency for buffered sites to be different from reference sites. Significant variables influencing plant assemblages included soil chemical parameters, canopy and ground layer cover, and channel morphology. Despite the geographic variation, our results suggest that buffered sites across the two catchments were similar and that buffering serves to homogenize the vegetation assemblage and structure.

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References

Alexander, K., G. Foster, and N. Sanderson. 2010. Good ecological status of inland water bodies: fencing of riverbanks is not good for biodiversity. Br. Wildlife 21:326–332.

Baattrup-Pedersen, A., N. Friberg, S.E. Larsen, and T. Riis. 2005. The influence of channelisation on riparian plant assemblages. Freshwater Biol. 50:1248–1261. doi:10.1111/j.1365-2427.2005.01383.x

Binkley, D., and C. Giardina. 1998. Why do tree species affect soils? The warp and woof of tree-soil interactions. Biogeochemistry 42:89–106. doi:10.1023/A:1005948126251

Bjerring, R., E.G. Bradshaw, S.L. Amsinck, L.S. Johansson, B.V. Odgaard, A.B. Nielsen, and E. Jeppesen. 2008. Inferring recent changes in the ecological state of 21 Danish candidate reference lakes (EU Water Framework Directive) using palaeolimnology. J. Appl. Ecol. 45:1566–1575. doi:10.1111/j.1365-2664.2008.01557.x

Borin, M., M. Passoni, M. Thiene, and T. Tempesta. 2010. Multiple functions of buffer strips in farming areas. Eur. J. Agron. 32:103–111. doi:10.1016/j.eja.2009.05.003

Bowers, K., and C. Boutin. 2008. Evaluating the relationships between floristic quality and measures of plant biodiversity along stream bank habitats. Ecol. Indic. 8:466–475. doi:10.1016/j.ecolind.2007.05.001

Compton, J.E., and D.W. Cole. 2001. Fate and effects of phosphorus additions in soils under N_2-fixing red alder. Biogeochemistry 53:225–257. doi:10.1023/A:1010646709944

Corbacho, C., J.M. Sanchez, and E. Costillo. 2003. Patterns of structural complexity and human disturbance of riparian vegetation in agricultural landscapes of a Mediterranean area. Agric. Ecosyst. Environ. 95:495–507. doi:10.1016/S0167-8809(02)00218-9

Correll, D.L. 2005. Principles of planning and establishment of buffer zones. Ecol. Eng. 24:433–439. doi:10.1016/j.ecoleng.2005.01.007

Decamps, H., M.E. McClain, and R.J. Naiman. 2001. Riparia: Ecology, conservation, and management of streamside communities: Aquatic ecology. Academic Press, Salt Lake City, UT.

Dennis, P., K. Beaton, S.J. Langan, and J.A. Stockan. 2006. Accumulated landscape ecological effects of riparian management in river catchments used for agriculture. *In* Proceedings of the IALE (UK) Annual conference Water and the Landscape: The Landscape Ecology of Freshwater Ecosystems, 5–8 Sept. 2006, Oxford Brookes University, Oxford, UK.

Ertsen, A.C.D., J.R.M. Alkemade, and M.J. Wassen. 1998. Calibrating Ellenberg indicator values for moisture, acidity, nutrient availability and salinity in the Netherlands. Plant Ecol. 135:113–124. doi:10.1023/A:1009765529310

European Commission. 1992. Directive 92/43/EEC (Habitats Directive). Official Journal of the European Communities, 21 May 1992.

European Commission. 2000. Directive 2000/60/EC (Water Framework Directive). Official Journal of the European Communities, 22 Dec. 2000.

Finzi, A.C., C.D. Canham, and N. Van Breemen. 1998. Canopy tree-soil interactions within temperate forests: Species effects on pH and cations. Ecol. Appl. 8:447–454.

Frego, K.A. 2007. Bryophytes as potential indicators of forest integrity. For. Ecol. Manage. 242:65–75. doi:10.1016/j.foreco.2007.01.030

Fritch, R.A., H. Sheridan, J.A. Finn, L. Kirwan, and D.Ó. Huallacháin. 2011. Methods of enhancing botanical diversity within field margins of intensively managed grassland: A 7-year field experiment. J. Appl. Ecol. 48:551–560 doi:10.1111/j.1365-2664.2010.01951.x

Gregory, S.V., F.J. Swanson, W.A. McKee, and K.W. Cummins. 1991. An ecosystem perspective of riparian zones. Bioscience 41:540–551. doi:10.2307/1311607

Grime, J.P. 1974. Vegetation classification by reference to strategies. Nature 242:344–347. doi:10.1038/242344a0

Gurnell, A.M., and G.E. Petts. 2006. Trees as riparian engineers: The Tagliamento River, Italy. Earth Surf. Process. Landf. 31:1558–1574. doi:10.1002/esp.1342

Gurnell, A.M., and G.E. Petts. 2002. Island-dominated landscapes of large floodplain rivers, a European perspective. Freshwater Biol. 47:581–600. doi:10.1046/j.1365-2427.2002.00923.x

Harmon, M.E., J.F. Franklin, F.J. Swanson, P. Sollins, S.V. Gregory, J.D. Lattin, N.H. Anderson, C.P. Cline, N.G. Aumen, J.R. Sedell, G.W. Lienkaempeer, K. Cromack, and K.W. Cummins. 1986. Ecology of coarse woody debris in temperate ecosystems. Adv. Ecol. Res. 15:133–302. doi:10.1016/S0065-2504(08)60121-X

Hill, M.O., J.O. Mountford, D.B. Roy, and R.G.H. Bunce. 1999. Ellenberg's indicator values for British Plants. Institute of Terrestrial Ecology, Huntingdon, UK.

Hulme, P.E., and E.T. Bremner. 2006. Assessing the impact of *Impatiens glandulifera* on riparian habitats: Partitioning diversity components following species removal. J. Appl. Ecol. 43:43–50. doi:10.1111/j.1365-2664.2005.01102.x

Huston, M., and T. Smith. 1987. Plant succession: Life-history and competition. Am. Nat. 130:168–198. doi:10.1086/284704

Jähnig, S.C., S. Brunzel, S. Gacek, A.W. Lorenz, and D. Hering. 2009. Effects of re-braiding measures on hydromorphology, floodplain vegetation, ground beetles and benthic invertebrates in mountain rivers. J. Appl. Ecol. 46:406–416. doi:10.1111/j.1365-2664.2009.01611.x

Lawes Agricultural Trust. 2009. Genstat release version 12. Rothamstead Experimental Station, Harpenden, UK.

Le Coeur, D., J. Baudrey, and F. Burel. 1997. Field margin plant assemblages: Variation partitioning between local and landscape factors. Landsc. Urban Plan. 37:57–71. doi:10.1016/S0169-2046(96)00370-2

Lyon, J., and N.M. Gross. 2005. Patterns of plant diversity and plant-environmental relationships across three riparian corridors. For. Ecol. Manage. 204:267–278. doi:10.1016/j.foreco.2004.09.019

Ma, M., S. Tarmi, and J. Helenius. 2002. Revisiting the species-area relationship in a semi-natural habitat: Floral richness in agricultural buffer zones in Finland. Agric. Ecosyst. Environ. 89:137–148. doi:10.1016/S0167-8809(01)00325-5

Nelder, J.A., and R.W.M. Wedderburn. 1972. Generalized linear models. J. R. Stat. Soc. Ser. A Stat. Soc. 135:370–384.

Paine, L.K., and C.A. Ribic. 2002. Comparison of riparian plant communities under four land management systems in southwestern Wisconsin. Agric. Ecosyst. Environ. 92:93–105. doi:10.1016/S0167-8809(01)00269-9

Parkyn, S.M., R.J. Davis-Colley, N.J. Halliday, K.J. Costley, and G.F. Croker. 2003. Planted riparian buffer zones in New Zealand: Do they live up to expectations? Restor. Ecol. 11:436–447. doi:10.1046/j.1526-100X.2003.rec0260.x

SEPA. 2010. River basin management plan for Tarland Burn. Available at http://apps.sepa.org.uk/rbmp/pdf/23338.pdf (verified 3 Aug. 2011).

Sinokrot, B.A., and H.G. Stefan. 1993. Stream temperature dynamics: Measurements and modelling. Water Resour. Res. 29:2299–2312. doi:10.1029/93WR00540

Soudzilovskaia, N.A., B.J. Graae, J.C. Douma, O. Grau, A. Milbau, A. Shevtsova, L. Wolters, and J.H. Cornelissen. 2011. How do bryophytes govern generative recruitment of vascular plants? New Phytol. 190(Feb.):1019–1031. doi:10.1111/j.1469–8137.2011.03644.

Stutter, M.I., A.J.A. Vinten, S.J. Langan, and J.A. Stockan. 2010. Managing stream riparian areas for multiple environmental benefits: The role of buffer strips. p. 374–379. *In* Proceedings of the SAC and SEPA Biennial Conference, Agriculture and the Environment VIII Climate, Water and Soil: Science, Policy and Practice. 31 Mar.–1 Apr. 2010, Edinburgh, UK.

Sweeney, B.W., and S.J. Czapka. 2004. Riparian forest restoration: Why each site needs an ecological prescription. For. Ecol. Manage. 192:361–373. doi:10.1016/j.foreco.2004.02.005

Syversen, N. 2005. Effect and design of buffer zones in the Nordic climate: The influence of width, amount of surface runoff, seasonal variation and vegetation type on retention efficiency for nutrient and particle runoff. Ecol. Eng. 24:483–490. doi:10.1016/j.ecoleng.2005.01.016

Tabacchi, E., and A.M. Planty-Tabacchi. 2005. Exotic and native plant community distributions within complex riparian landscapes: A positive correlation. Ecoscience 12:412–423. doi:10.2980/i1195-6860-12-3-412.1

Tattari, S., T. Schultz, and M. Kuussaari. 2003. Use of belief network modelling to asses the impact of buffer zones on water protection and biodiversity. Agric. Ecosyst. Environ. 96:119–132.

ter Braak, C.J.F., and P. Šmilauer. 2002. CANOCO reference manual and CanoDraw for Windows. User's guide: Software for Canonical community coordination. Version 4.5. Microcomputer Power, Ithaca, NY.