QUANTIFYING THE RELATIONSHIP BETWEEN LAND COVER AND BIOLOGICAL CONDITION OF HEADWATER STREAMS

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Abstract

There has been considerable effort recently to gain insight into the relationships between the characteristics of catchments and the ecological status of streams within rural landscapes. Some such studies have been confounded by the natural changes seen in land use longitudinally along a river, or have been limited by poor replication. As part of the long-running Countryside Survey programme, a random-stratified sample of over 400 replicate headwaters across all British landscape types has been sampled every eight years since 1990. The macroinvertebrate and aquatic plant communities were sampled and the hydromorphological features were surveyed at each site. Adjacent riparian vegetation and habitats were mapped in detail as well as catchment land cover for each stream site. Such a comprehensive dataset allows a robust assessment of the association between catchment and riparian land use and stream biological integrity. Very few significant correlations were found between cover of particular land uses and stream condition. In general there was considerable unexplained variation in the biological data. Only the aquatic plant Mean Trophic Rank score (MTR; an index of nutrient enrichment) showed a consistent association with catchment and riparian land cover. MTR scores were positively associated with forest and natural grassland cover and negatively associated with arable and improved pasture cover. Relationships between biotic measures and land cover were generally not improved by restricting the spatial extent of the analysis from the whole of Britain to a more homogenous landscape type, e.g. easterly lowland; nor was it improved by standardising the observed family richness of macroinvertebrates by a site-specific prediction of reference conditions. This analysis reveals the complexity of relating land cover to stream biology. Perhaps either a more elegant measure than %

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cover is needed to describe the influence land use has on stream condition or researchers should focus on quantifying the actual causative agents affecting stream biological communities rather than convenient surrogate variables.

Keywords: land cover, catchment, riparian, macroinvertebrate, macrophyte, headwater stream.

Introduction

It has long been appreciated that streams are intimately connected with their catchment (Hynes, 1975). Geomorphology determines soil, slope and aspect which, together with climate, govern the vegetation cover. Vegetation, in turn, influences the supply of organic matter, sediment and water to the stream, ultimately playing a large part in determining the lotic biological community. There has been no shortage of studies over the past two decades assessing how variation in catchment characteristics affects the stream community (Schlosser, 1991; Strayer et al., 2003; Allan, 2004). In particular, there has been a focus on the consequences of more intensive human exploitation of the land. The manner in which stream biota have been shown to be impacted by such changes to the surrounding landscape varies between studies (Richards et al., 1997; Sponseller et al., 2001). This is most likely due to differences in (1) the choice of land uses that have been encompassed, (2) the range of spatial and temporal scales incorporated and (3) the biological responses measured.

Some studies have been confounded by the natural landscape changes seen longitudinally along a river; where strong biological responses to changes in land cover may actually just reflect normal differences in the community between headwaters and lowland reaches (Probst et al., 2005; Park et al., 2006). Other studies finding a strong link between catchment land cover and stream biology have compared similar-sized headwater subcatchments within a region but have selected sub-catchments with near-homogenous land cover, i.e. they compare biological responses between different categories of catchment land cover, optimising the difference between categories (Moore & Palmer, 2005; Burcher & Benfield, 2006). What I was interested in investigating is whether streams are affected by more subtle variations in land cover, among catchments that are a mosaic of intensively managed, semi-natural and natural patches. Further, I was interested to discover the degree to which results are influenced by spatial scale, underlying natural gradients and biological responses measured.

The Countryside Survey dataset

The dataset used to address these questions was derived from the Countryside Survey (CS) programme (Firbank et al., 2003). Countryside Survey is a uniquely powerful ongoing survey of the rural environment which provides robust estimates of stock and change in the flora and fauna, habitats and landscape features of Britain. Estimates are based on the repeated detailed field recording of many features in a stratified random sample of nearly 600 one-kilometre squares. The stratification is based on a hierarchical classification of the British landscape into land classes nested within environmental zones. The classification is based on geology, climate and topography (Bunce et al., 1996) (Fig. 1). The most recent survey was completed in 2007, with many aspects reported the following year (Carey et al., 2008). For the present study, data from the previous survey in 1998 were used, where in 425 of the 591 squares a single headwater stream was surveyed. The macroinvertebrate community and aquatic plant community were surveyed using standard methods (Murray-Bligh et al., 1997; Holmes et al., 1999; Murphy & Weatherby, 2008). The macroinvertebrate sampling involved a 3-minute active kick sample of the stream bed with a 900 µm-mesh pond net, where all benthic habitats within the site were sampled in proportion to their occurrence. Specimens were identified in the laboratory to species level where possible. The percentage cover of all aquatic plant species found over a 100 m stretch of stream, centred on the macroinvertebrate sampling site, was estimated. A range of environmental parameters were also recorded for each site, including stream width, depth, substratum composition, water chemistry (pH, conductivity, alkalinity and soluble reactive phosphorus), water velocity, slope, altitude and distance to source.

At the broadest spatial scale each 1 km square with a stream sampling site was assigned to one of six environmental zones (Bunce et al., 1996). Catchment-scale land cover data were acquired from satellite-imagery captured in 1998 (Fuller et al., 2002). Finally, in the field, the riparian land cover along a 10 m wide strip either side of the watercourse was recorded, for 500 m upstream of the macroinvertebrate sampling point. Land cover at both spatial scales was assigned to one of six categories (forest & woodland, arable, improved pasture, natural grassland, heath & bog, and urban). A complete set of biological and land cover data was available for 320 of the 425 stream sites. Catchment land cover differed significantly across the 6 environmental zones (EZs) reflecting the underlying differences in climate, geology and geography. A gradient of land-cover intensification was evident from the true uplands of Scotland (EZ6), through EZ5, EZ3, EZ4 and EZ2 to the easterly lowlands of England (EZ1) (Fig. 2). In England and Wales, arable land cover was dominant in the easterly lowlands, a combination of arable and improved pasture in the...
westerly lowlands, and natural grasslands, heath & bog in the uplands. In Scotland, the lowland catchments featured a mixture of arable, improved pasture and natural grasslands, and the intermediate and true uplands were mainly covered in heath & bog and natural grasslands (Fig. 2). Forest cover accounted, on average, for around 10% of catchments in all environmental zones.

Analysis

Macroinvertebrate data were reported as taxon richness and as the ratio of observed family richness to that predicted by the RIVPACS III model (Wright, 2000). RIVPACS (River Invertebrate Prediction and Classification System) predicts the number of macroinvertebrate families that would be expected to occur at a running water site, in the absence of any anthropogenic stresses, based on measured geomorphological characteristics of the site (Clarke et al., 2003). Comparing the observed richness to that expected (O/E) provides an indication of the biological condition of the site, where a value close to unity indicates that the site is in high or good ecological condition. The aquatic plant data were reported as taxon richness and the derived Mean Trophic Rank (MTR) score. The MTR score is a standard UK methodology for the assessment of the trophic status of rivers using macrophytes and is calculated based on the cover of selected plant species and their sensitivity to nutrient enrichment (Holmes et al., 1999).
Generalised additive models (GAMs) were used to quantify relationships at both catchment- and riparian-scale for each of the four different biological response variables (macroinvertebrate richness, macrophyte richness, O/E taxa, MTR score). While linear regression or rank correlation could also have been used to test the hypotheses, it was felt that constraining the models to linear responses would have meant that some biologically-interesting non-linear patterns may have been overlooked. Generalised additive models do not impose a shape of relationship between the response and predictor variables; non-parametric smoothers are used to describe the relationship in the most parsimonious way possible, as judged by information criteria (Wood, 2006). A measure of the goodness of fit of the model describing the relationship is provided by the deviance explained (expressed as a percentage of total deviance). The estimated degrees of freedom (edf) gives an indication of the non-linearity of the relationship, with a value of 1 indicating a linear relationship and larger values associated with more complex non-linear correlations. A judgement has to be made as to when the GAM has proposed an excessively complex model, i.e. one that is not biologically meaningful. For the present study I chose edf 4 as the threshold above which GAMs were most likely to be over-fitted. The statistical significance of the fitted model is tested using an approximate $F$-test. GAMs were fitted using the ‘mgcv’ package (Wood, 2006) within R 2.8.0 (R Development Core Team, 2008). Because of the increased chance of Type I error, $P < 0.01$ was considered as the significance threshold. Variables were transformed where necessary to meet assumptions of normality.

The following specific $a$ priori hypotheses were tested:

1. Across all environmental zones the ecological status of watercourses:
   - improves with more forest cover and with more natural grassland cover within the catchment and riparian areas
   - deteriorates with more arable land and with more improved pasture within the catchment and riparian areas.

2. In easterly lowland environmental zones (EZs 1 and 4) the ecological status of watercourses:
   - improves with more natural grassland cover within the catchment and riparian areas
   - deteriorates as a greater proportion of the catchment contains forest cover
   - deteriorates as a greater proportion of the catchment contains improved pasture.

3. In the westerly environmental zone (EZ 2) the ecological status of watercourses:
   - improves with more forest cover within the catchment and riparian areas and with more natural grassland cover within the catchment area
   - deteriorates as a greater proportion of the catchment area is used for arable agriculture.

4. In upland environmental zones (EZs 3, 5 and 6) the ecological status of watercourses:
   - improves as a greater proportion of the catchment and riparian area contains forest cover
   - deteriorates as a greater proportion of the catchment contains improved pasture.

Of the 76 relationships between land cover and biological condition quantified, 13 (19%) were found to be statistically significant ($P < 0.01$) while not being excessively complex (i.e. edf < 4) (Table 1). These associations could account for between 5 % and 38 % of the variation in the biological data, with only four of these 13 relationships accounting for over 20 % of the biological variation (Table 1, Fig. 3).

**Land cover–biota relationships across all of Britain**

**Forests**

Across Britain, macroinvertebrate taxon richness and MTR scores in headwater streams had weak relationships with catchment forest cover (Table 1). Plant richness was not related to catchment forest cover. At a riparian scale, only MTR score was related to variation in forest cover.

**Arable**

MTR score was also the only measure of biological quality that varied significantly with catchment or riparian arable land cover (Table 1). MTR scores declined with increasing arable land cover, with the relationship being particularly pronounced at a catchment scale, between 0 % and 30 % cover; there tended to be no further reduction in MTR score beyond that threshold (Fig. 3a).

**Improved pasture and natural grassland**

Macroinvertebrate richness was weakly related with catchment cover of improved pasture (Table 1). MTR scores declined with increasing catchment (Fig. 3b) and riparian (Fig. 3c) cover of improved pasture (Table 1). Variation in plant richness was not associated with catchment or riparian cover of improved pasture. Most measures of biological quality were unrelated to variation in the catchment or riparian cover of natural
Table 1. Relationships between biological condition of headwater streams (as measured by macroinvertebrate richness and Mean Trophic Rank (MTR) score) and land cover (measured at catchment or riparian scale), assessed at two spatial extents (all Great Britain or restricted to individual environmental zones). Relationships were quantified using Generalised Additive Models (GAMs) and the goodness of fit of the model is provided by the deviance explained (expressed as a percentage of total deviance (% de)). The estimated degrees of freedom (edf) gives an indication of the non-linearity of the relationship, with a value of 1 indicating a linear relationship and larger values associated with more complex non-linear correlations. The statistical significance of the fitted model is tested using an approximate F-test.

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land cover–biota relationship was assessed to investigate whether it strengthened relative to that found across the whole of Britain.

**Easterly lowlands of Britain**

Within easterly lowland areas of Britain, the biological condition of headwater streams, as measured by macroinvertebrate or macrophyte richness, was unrelated to the cover of arable land, at catchment or riparian scale. Therefore there was no discernible strengthening in the relationship when the analysis was restricted from the whole of Britain to just the easterly lowlands. MTR scores were related to catchment-scale cover of arable land, as was found at the all-Britain scale but, contrary to expectation, residual ‘noise’ about the relationship was not reduced by restricting the analysis to a single environmental zone (Table 1).

The relationship between biological condition of headwater streams and the cover of natural grassland did not improve when restricted to the easterly lowland areas of Britain, at catchment or riparian scale.

**Westerly lowlands of England and Wales**

In westerly lowland areas of England and Wales, MTR scores increased in association with increasing riparian forest cover (Table 1, Fig. 3d). A similar, though noisier, relationship was found across all Britain (Table 1). Stream biological condition was unrelated to catchment-scale cover of arable land.

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Within the uplands of Britain there was a weak relationship between macroinvertebrate richness and catchment-scale cover of forest (Table 1). It was only marginally more reliable in terms of the % deviance explained, than the equivalent association at an all-Britain spatial extent.

Only MTR scores decreased significantly in association with increasing improved pasture cover in upland catchments (Table 1). This latter

**Stratifying by environmental zone**

One of the study objectives was to assess to what extent relationships were influenced by underlying natural gradients. A means of factoring out some of the confounding effects of natural gradients, would be to focus analyses on individual environmental zones rather than the whole of Britain, thereby reducing the degree of natural variability between catchments. The 320 stream sites were therefore assigned to one of three broad landscape types: easterly lowlands of Britain, westerly lowlands of England and Wales, and uplands of Britain. Within each landscape the land cover–biota relationship was assessed to investigate whether it strengthened relative to that found across the whole of Britain.

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relationship, being linear, was simpler than that found across all sites, but could only account for 12.8% of the variation explained as opposed to 26.8% for the equivalent all-Britain model.

**Using RIVPACS to factor out influence of underlying natural gradients**

An alternative approach to lessening the confounding effect of underlying natural gradients and biogeography on relationships between land cover and biological condition, was to use the ratio of observed macroinvertebrate richness to that expected (O/E\textsubscript{taxa}) for each stream site (given its location, physical characteristics and the absence of human impacts) as the dependent biological variable, in place of macroinvertebrate richness. This measure effectively provides an indication of the extent to which the macroinvertebrate community is impacted by anthropogenic impacts.

However, O/E\textsubscript{taxa} was unrelated to variation in any of the land covers analysed, either at catchment or riparian scale. For those few instances where there were significant relationships between macroinvertebrate richness and land cover, the corresponding relationships with the macroinvertebrate data presented as O/E\textsubscript{taxa} were actually weaker. This biological measure is the standard tool used by UK government environment agencies, so it is somewhat surprising to find that it does not distinguish between headwater streams at either end of the land-use intensity gradient. It suggests that the biological quality of headwater streams, as quantified by their macroinvertebrate communities, is not overly sensitive to catchment or riparian land cover. Alternatively it may be that the prediction of the expected richness, derived from RIVPACS, is itself not independent of any potential impacts from more intensive land covers. Specifically, it should be noted that nearly 180 of the RIVPACS reference sites are situated on watercourses with greater than 50% arable cover in their catchments (derived from the CORINE 2000 land cover map and assuming that these data are broadly representative of the situation when reference sites were actually sampled for RIVPACS) (Davy-Bowker et al., 2007). This could negate the potential for O/E\textsubscript{taxa} to respond to the land-cover intensification gradient, as the predicted number of macroinvertebrate families may be suppressed by diffuse impacts from such agriculture. The selection and screening of RIVPACS reference sites did not explicitly include as a criterion a threshold of ‘natural’ catchment land cover. Rather, sites were selected and screened primarily on the basis of the perceived quality of the biological community, geographical and stream typological coverage (Wright, 2000). Moors River at Pinnocks Moor in Dorset, for example, is a small stream site with 80% arable land-cover, but was considered to be among the best biological quality examples of its type available and is a RIVPACS reference site. It would be challenging to build a RIVPACS bioassessment model for Britain using only sites with ‘natural catchments’ as such catchments do not exist for many stream types.

**Discussion**

As described above, very few significant correlations (19% of those quantified) were found between cover of particular land uses and stream condition. In general there was considerable unexplained variation in the biological data. There was no evidence that stream biota responded more closely to riparian land cover than to catchment land cover. Furthermore, the land cover–biota relationships were not consistently better defined when investigated within a particular landscape type, relative to that found across the whole of Britain. Even after restricting the spatial extent of the study sites, there was still considerable residual variability in the biological response to changes in land cover.

Other studies relating changes in stream biological communities to land cover have often reported stronger associations (Harding & Winterbourn, 1995; Roth et al., 1996; Allan, 2004; Moore & Palmer, 2005). Such studies have usually assessed streams along a gradient of modification from near total forest cover to intensively farmed or urban catchments, thus maximising the potential for strong biological responses (Allan et al., 1997). There are also studies that report a lack of land cover–biota associations, as found in the current study. Strayer et al. (2003) found that macroinvertebrate richness was unrelated to catchment cover of cultivated land or pasture across 269 sites in the mid-Atlantic region of USA. Sponseller et al. (2001) found that macroinvertebrate density and richness was unrelated to catchment or riparian corridor cover of non-forested land in nine Appalachian streams. It is difficult to draw together a consensus from all these studies as their results are often a function of the study design and the nature of the selected sites (Allan et al., 1997). Hence, caution should be exercised in applying the findings from a study in one catchment or region to the management of catchments elsewhere.

In this study, there was rarely a significant association between riparian-scale land cover (riparian corridor 500 m long × 10 m wide) and the biological end-points. The exceptions were relationships between MTR score and riparian forest, arable and improved pasture cover across Britain and riparian forest in the westerly lowlands. Most significant associations tended to be at catchment scale. Roth et al. (1996) also found that correlations between biological condition and land use were stronger at catchment rather than local riparian-scale. In contrast, Sponseller et al. (2001) reported that most biological end-points investigated were more closely correlated with changes in riparian non-forest cover than catchment...
cover. It is likely that conclusions drawn on the relative importance of catchment vs. riparian factors will depend to some extent on the range of variables analysed. Furthermore, there is no standard width for the riparian corridor; different studies have based their analysis on variously-sized riparian zones, which must have some influence on the outcomes and conclusions drawn. In reality, the effective width of the riparian zone will vary between and along watercourses with changes in physical attributes (e.g. slope) of the local landscape, so there is no universal recommended threshold width. The issue is not an unimportant one, because there is a hope in catchment management circles that stream biotic integrity could be assured by the right riparian land cover, regardless of the degree of modification in the wider catchment. However, this study does not provide any strong evidence to support this theory.

There were consistent differences between the responsiveness of the four biological measurements to changes in land cover. Variation in both aquatic plant richness and $O/E_{\text{aqua}}$ between streams was not related to changes in catchment or riparian land cover. In contrast, macroinvertebrate richness and MTR appeared to be more affected. Other studies have also found that different biological end-points can respond differently to the same environmental gradients (Johnson & Hering, 2009). Johnson & Hering (2009) found that the composition of benthic diatom and aquatic plant assemblages showed a non-linear response to a nutrient gradient, but only at low-medium stress levels and beyond a certain threshold there was no response. Benthic macroinvertebrate assemblage composition, on the other hand, responded to the same gradient but only at much higher concentrations. The current study found that MTR score was the measure that most often showed a response to the land cover gradient. This is likely to be due to the fact that variation in land cover will have consequences for the level of nutrient inputs to watercourses; the stress that MTR is specifically designed to indicate. The other three measures assessed are indices of general environmental degradation and may inherently not be as sensitive to land cover changes.

It is particularly interesting to find that the RIVPACS-derived $O/E_{\text{aqua}}$ index showed no association with variation in land cover. This raises the possibility that the widely-used measure of general degradation (family richness) is in fact not capable of detecting the subtle impacts of land-use intensification on streams and rivers. An alternative index may be needed that better incorporates aspects of the macroinvertebrate community that respond to the changes associated with land-use intensification, e.g. an index that includes relative abundance or community composition information. It may also be that use of the RIVPACS tool is not appropriate for assessing the biological response to variation in land cover. RIVPACS predictions are based on measurements of the geomorphological characteristics of the site, e.g. water width and depth, and substratum composition. In theory, these predictor variables should be independent of the stresses potentially acting on the biological community. Unfortunately, this may not always be the case when it comes to impacts of intensive land use on streams and rivers.

It must also be remembered that land cover is really just a surrogate for the actual proximal factors potentially affecting the stream community, e.g. excessive sediment, nutrients or pesticides entering the stream or hydromorphological modifications to the channel. Perhaps it is too gross a simplification to assume that a given percentage cover of land use has an equivalent effect on all streams. While it is convenient, with the aid of GIS applications, to acquire and use catchment and riparian land cover information, it may be more ecologically meaningful to quantify the causative agents at a scale that is appropriate to relate to the biological data, e.g. reach-scale sediment delivery.

Finally, what is evident from the present study is that the differences between streams in their biological condition appear to be, by and large, independent of variation in the cover of different land uses in their catchments. While it is unlikely that streams are not in fact ruled by their catchments, what is clear is that the relationship between the two is not an easily modelled/predicted one. It is possible that catchment land cover has not been measured at the right spatial or even temporal scale (Maloney et al., 2008), or perhaps that the land cover data are being presented in the wrong format. It may be that % cover is too crude a measure for the purpose of these analyses, and that the location of patches of different land cover in relation to the stream and to each other should somehow be incorporated. This inevitably makes the modelling more complex but it has been attempted by some workers, e.g. Johnson et al. (2007) and with some partial success. Furthermore, the land cover–biota relationship is likely to also be disrupted by unpredictable local-scale impacts such as cattle crossing and pollution point-sources, which were not included in the current analysis.

Either way it is clear that there is plenty of scope for using extensive datasets, such as Countryside Survey data, along with data from other sources, to better understand how catchments control streams (sensu Hynes, 1975). Alternatively, rather than relying on information from general extensive surveys, it may be more productive to undertake an intensive, appropriately-designed field experiment to quantify the biological effects of a given land-cover gradient, or even more specifically to assess the impact of a particular stressor, e.g. sediment. Indeed the changes taking place in the UK countryside over the past decade as a result of agri-environment schemes (where land managers agree to lessen their ‘footprint’ on catchments and to actively seek to create and protect...
freshwater habitats), means that there is an opportunity to monitor the biological response to and effectiveness of these restoration initiatives.

It is likely that a combination of both extensive surveys and targeted field experimental approaches will provide the best opportunity for gaining a better understanding of the relationship between catchments and water quality, to allow us to manage our use of the land to better protect freshwater resources.

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