Catchments, Sub-Catchments and Private Spaces: Scale and Process in Managing Microbial Pollution from Source to Sea

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Abstract

This paper examines the implications of adopting catchment scale approaches for the sustainable management of land and water systems. Drawing on the findings of an interdisciplinary study examining how farm management practices impact on the loss of faecal indicator organisms (FIOs) and potential pathogens from land to water, the paper argues that the overwhelming focus on integration at the catchment level may risk ignoring the sub-catchment as an equally appropriate unit of hydrological analysis. Further the paper suggests that many of the management decisions relevant to water quality are made by land occupiers and, therefore, that the identification of relevant socio-spatial units – the 'private spaces' of land holdings - may be as important or more important to the effective management and planning of water resources as catchment-level planning.

Keywords: Catchment, Scale, Water Pollution.

1. Introduction

The size and appropriateness of defined spatial units for regulatory and management purposes is an issue that has long exercised the attention not only of geographers but also of political scientists, economists, planners and environmental scientists. For geographers and policy scientists there has been a growing interest in the rescaling of statehood and associated regulatory and institutional arrangements as nation states find new ways of partitioning territory (Brenner 2004; Bulkeley 2005). This is picked up in economics and planning in more normative discussions of the appropriate levels of territorial jurisdiction for different public functions, often known as fiscal federalism (Besley and Coates 2003; Oates 2005); and in biogeography and environmental science in notions of 'natural areas' (Gray 2001), 'river basins' (Blackstock, 2009) or 'catchments' (Keirle *et al.*, 2007). Thus the 'spatial turn' has impacted on a number of different disciplines struggling to develop analysis of issues of space and scale. In geography there is a lively debate about scale (Blöschl and Sivapalan, 1995; Brazier *et al*., 2005; Couper 2007; Marston et al., 2005; Parsons et al., 2006; Slaymaker, 2006; Sivapalan, 2003). For example, Moore (2008) has exposed a conceptual confusion surrounding scale as a result of the failure to draw a clear distinction between scale as an empirical or analytical category: "in adopting scale as a category of analysis geographers tend to reify it as a fundamental ontological entity, thereby treating a social category employed in the practice of sociospatial politics as a central theoretical tool" (p203).

In this paper we wish to consider the implications for policy delivery and environmental management on the ground of adopting a specific scalar category, in this case the catchment approach. We do so in part because of the way in which the 'spatial turn' has been taken up so enthusiastically by policy makers, most explicitly perhaps in the notion of water catchments. Thus the 2000 EU Water Framework Directive (WFD) 2000/60/EC (CEC, 2000) requires integration of jurisdiction within and between catchments in the form of integrated river basin management (IRBM). Here integrated management plans are drafted at the river basin scale in contrast to plans previously administered at political units of scale, a shift characterised by Wiering and Immink (2006) as a move from the old water management to the new spatial planning. The WFD represents an overarching framework for an integrated management structure to meet key environmental objectives. These objectives are linked to halting further deterioration of water resources and habitats, promoting the use of water in sustainable ways through the concept of 'ecosystem health' and improving the protection of receiving waters (Collins and McGonigle, 2008). The implementation of the WFD has led to an increase in the regulatory drive to protect and improve the quality of water bodies in the UK. The inclusion of associated directives such as the revised Bathing Waters Directive (rBWD) (CEC, 2006) within the WFD in Europe highlights the recognition of the importance of microbial water quality within this framework.

In this paper, we challenge the new orthodoxy of catchment-level planning in two respects. First, we argue that the overwhelming focus on integration at a catchment level is leading to a neglect of the importance of the sub-catchment as an equally appropriate unit of hydrological analysis (e.g. Buck *et al.*, 2004). Secondly, we suggest that many of the management decisions relevant to water quality are made by land occupiers and, therefore, that the identification of relevant socio-spatial units – the 'private spaces' of land holdings - may be as important or more important to the effective management and planning of water resources as catchment-level planning. Whilst catchments are primarily presented as natural hydrological units, their 'naturalness' also has political implications. It is hardly a novel observation that natural units cut across political and administrative boundaries and the primacy given to the 'natural' boundaries in the catchment approach is consistent with new cross-cutting forms of partnership governance (Edwards *et al.*, 2001). The catchment approach gives rise to both new modes of political engagement within catchment governance (Kallis *et al.*, 2006) and new partnerships which may also serve to undermine pre-existing jurisdictions as has been shown, for example, in the US context by Maddock (2004).

To date, the challenges of hydrological management and farmer behaviour (and of catchment governance) have been approached in relative isolation and according to conventional disciplinary approaches. This paper, drawing on the findings of a research project examining how farm management practices impact on the loss of faecal indicator organisms (FIOs) and potential pathogens from land to water, combines insights from social and natural sciences[1]. Its focus was on the Taw River Catchment in North Devon, UK (see Figure 1 and 2). The research programme included detailed microbial monitoring of water courses in the catchment; an interview survey of 75 farmers across the Catchment eliciting attitudes and practices towards manure, land and livestock management; and a programme of public participation and debate in which emerging scientific evidence was explored in relation to wider understandings and assessments of environmental risk. In all of this, it is important to emphasise the wide range of inter-disciplinarity underpinning the research, involving those with expertise in microbiology, hydrochemistry, soil hydrology, manure management, rural geography and political science (Chadwick *et al*., 2008).

2. Rescaling jurisdictions: the catchment approach

The catchment approach derives from a perceived governance problem that is both territorial and political. The territorial jurisdiction of agencies responsible for water quality has not historically extended to include land and water longitudinally (up-stream) and laterally across the catchment (Moss, 2004). Politically, there is often a lack of co-ordination of the many policy agencies whose actions, or inactions, may influence water quality – planning, agriculture, conservation, etc. (Moss, 2004). Few would deny the veracity of these arguments

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and the challenge facing policy makers is readily comprehensible. The EU Water Framework Directive is a bold attempt to tackle these issues.

The Directive provides a rare example of a policy initiative that was almost universally welcomed, if only as a first step (Macleod et al., 2007), by academics, many of whom had long championed the cause of holistic and integrated solutions to water problems. The catchment scale approach lends itself to fostering public and stakeholder participation and engagement in decision making responsibilities and adoption of mitigation strategies, a key requisite of establishing river basin management plans (RBMPs). The England Catchment Sensitive Farming Delivery Initiative (ECSFDI) is one such mechanism whereby partnerships are developing between farmers and land-owners and agricultural advisors, water companies and competent bodies (Defra, 2008). This initiative has been adopted in England in an attempt to minimise diffuse water pollution from agriculture (DWPA) in 40 priority catchments.

Integrated catchment management (ICM) should aim to deliver scientific clarity with regard to catchment management options and their associated impact at a policy-driven scale. However, such clarity is difficult to achieve. There is limited evidence that effects of a change in land use management can be distinguished in the water quality signal of streams draining agricultural land at large (e.g. > 10km2) catchment scales in the face of, for example, climate change. For example, year on year variation in hydrological conditions (a drought year versus a wet year) can disguise such signals in water quality and this is an issue which has plagued environmental scientists for decades. This is confounded by the constraints of research funding mechanisms which tend to only fund 3-5 year programmes of study. Observing real improvement in water quality over such short time frames in large catchents is therefore unlikely. A recent initiative in England and Wales, funded by the Department for Environment, Food and Rural Affairs (DEFRA) and termed the Demonstration Test Catchment initiative, is one example of a recent longer-term programme seeking to identify how combinations of mitigation measures in small subcatchments (~10km2) of much larger catchments can lead to improvements in water quality. To be able to detect the improvement in water quality signal and maintain some degree of control and understanding of how and why improvements may occur a subcatchment approach such as this is essential.

. Integrated catchment or river basin management must therefore attempt to assimilate information about a particular sub-scale of operation. Thus, ICM represents an abstraction of complex processes integrated over space and time (Bouma *et al*., 2008; Lane *et al*., 2006). For example ICM attempts to account for the complex relationships between flora and fauna, geology and hydrology, soils, the atmosphere, and many other interacting catchment scale factors. While such an integrated approach is to be admired in contrast to previous piecemeal approaches that artificially separate land management from water management, it is fraught with conceptual and methodological difficulties (Blackstock, 2009), as we will outline below. As a result, the envisaged scientific clarity sought from ICM may be difficult to achieve.

Land use questions are most often raised at the farm enterprise level and arguably the farm level represents the most basic management unit. However, we should exercise caution when farm enterprises are aggregated into this higher-level scaling of ICM, given the potential for decision making and land management actions at the local level to be lost in the wider catchment approach. Scaling-up process-based understanding from molecules through to management tools is an ongoing challenge in the physical sciences, so scaling-up combinations of physical and social processes in different sectoral aspects of catchments is even more demanding (Hodgson and Smith, 2007). Additionally, an appreciation of inherent uncertainties encountered when aggregating site-specific process equations that describe local scale environmental functions to make regional-scale predictions is needed (Standing et al., 2007). This is because dominant processes will change depending on the scale of observation.

While Standing et al (2007) make this statement in the context of the scaling-up of microbial dynamics within the environment, we argue that those scaling issues are equally relevant when applied to decision making activities at the field and farm level scaled through to the catchment and river basin scale.

Of course, by promoting a river basin approach, the WFD seeks to promote a rescaling of water governance to better match underlying hydrological realities. Article 3 of the WFD stipulates that member states (MS) are required to identify water bodies to River Basin Districts (RBDs), based on hydrological catchment areas. So the RBD level is the key geographical unit to which the WFD refers, but this is not free from complications when rivers cross national or regional borders. As a result, improved co-operative approaches of working between constituent parts of the UK are needed to manage rivers by hydrological rather than political boundaries.

Hodgson and Smith (2007) suggest that the operation of the WFD at larger scales has brought about increased tensions between catchment stakeholders. For example, some believe that the WFD has missed the opportunity to improve pollution prevention locally. Management at larger spatial scales does allow a more strategic approach but coarse scale spatial units accommodate considerable heterogeneity in, for example, the nature and intensity of agricultural practice. Strategies designed to address regional scale implementation of management approaches arguably translate into potentially ineffective management regimes at the ground level and have the potential to be costly and fail to achieve targets in some areas (Johnes et al., 2007). So at the river basin scale there is a need for *targeted* management. This essentially requires a multi-scaled methodology to facilitate appropriate management at the farm level focussing on the risks of different farming practices for contributing agriculturally derived contaminants to receiving waters. For example, farming at the mouth of the estuary close to any regulated monitoring with, potentially a high impact on bathers, has a greater risk than similar farming closer to the head of the catchment. However, high risk activities may not be particularly welcome at the head of the catchment either, if that is where drinking water sources are found. Herein lies our argument that sub-catchments are a helpful unit of spatial scale. This is in line with the arguments of those who have also advocated the need for a two-tiered approach to catchment management, combining broad regional policies with targeted management in high risk areas at the catchment and farm scale (Heathwaite et al., 2000; Hewett et al., 2009; Johnes et al., 2007).

The argument is strengthened when we consider Article 9 of the WFD requiring states to account for the recovery of costs of water services and to report on the cost-effectiveness of combinations of measures. RBMPs must therefore be grounded by appropriate spatial targeting of cost-effective control options. But how appropriate or feasible is it to gauge cost-effectiveness of water quality protection measures at a catchment scale? Combinations of mitigation will work differently in space and time for different farm enterprises and there will be much predictive uncertainty. Understanding the optimal combination of measures at the catchment scale is a huge challenge and one that scientists and policy practitioners have not yet grasped even at smaller scales with regard to mitigating microbial pollution from individual farms.

One clear hindrance to catchment-scale working is the reliance on assumptions and scaling rules. For optimised mitigation at the catchment scale one assumption is that of a wide uptake of management strategies. Yet if we were to investigate uptake of measures on a farm-by-farm basis then such assumptions are likely to be inconsistent with the ground-truthing of data simply because a diverse subset of socio-economic factors are likely to dictate farmer decision making and uptake at the local level. In turn, it becomes difficult to guarantee that within a catchment, neighbouring farms will not jeopardize microbial water quality at the expense of the effort of others (Oliver et al., 2007). We therefore need to know the spatial pattern of uptake but the catchment scale may be too coarse to represent these patterns in a meaningful way. This argument has been substantiated by LaWare and Rifai (2006) who highlight how grazing is generally modelled uniformly throughout catchments, and yet farmers adopt a range of contrasting land management practices. Any assumption of uniform grazing intensity or application of nutrients to land is manifestly misplaced. If efficacy of mitigation is controlled by geographical location and associated attributes of the landscape (such as soil type and topography) then there are considerable uncertainties linked to catchment scale outputs as a consequence of interpolation and the need to scale up.

3. Understanding local water in tackling pollution: the importance of sub-catchments

Certain sub-catchment areas are likely to be more sensitive to land management change relative to others and site specific issues are likely to be incorrectly represented in larger aggregate scales (Hatton-Ellis, 2008). A concern relating to catchment scale research is that the significance (and cause) of critical tributaries of poor water quality can be masked by the catchment approach if spatial units are aggregated to a whole catchment without interrogation of the contributory sub-catchment components. Work by Stapleton et al (2008) on the river Douglas sub-catchment of the Ribble catchment in the UK has illustrated this point. They were able to demonstrate that in the 1583km^2 Ribble catchment, over half the load of FIOs discharged were attributed to the relatively small sub-catchment of the River Douglas. By targeting such critical sub-catchments and their associated physical functioning and the inherent processes driving sub-catchment response we can perhaps argue that gains can be made at the catchment scale. Integrated catchment management by its very name assumes the concept of a whole systems approach but risks ignoring smaller sub-compartments of areal units, potentially overlooking the importance of smaller scale hydrological functioning.

Sub-catchment management has the potential to offer many benefits. For example, managing the peak in concentration of a contaminant reaching the main tributary of a catchment after a storm event could be useful to prevent all peaks (of flow and associated contaminants) from a series of lower order streams converging at the same time and causing a critical load that would be of high risk and unacceptable with regard to microbial monitoring standards. So theoretically the successful management of sub-catchments can result in positive impacts at the catchment level. However, sub-catchment manipulation is logistically difficult because a suite of factors operate in a spatial and temporal sense (e.g. different rainfall volumes and intensity ranges, different soil conditions etc.). Small scale catchment management can make an impact, but when scaled-up to the catchment the bottom-line evaluation is whether or not we can detect evidence from land management changes made at the local level (e.g. Wood et al., 2005; Haygarth et al., 2005).

Small catchment areas or sub-catchments allow for the physical mapping of land use and other landscape features required for predictive modelling using field by field surveying approaches. At larger catchment scales relevant to the WFD such mapping is impossible due to time constraints and instead nationally available datasets are required which contain inherent uncertainties and extrapolations. Thus increasing scale results in lower resolution data and increased uncertainty in data accuracy. Kay et al (2005) address such issues in their Ribble (the UKs sentinel WFD catchment) study. Detailed field mapping provided a ground-truthed dataset with which to compare with coarser land cover classes available at 25m resolution and yielded clear discrepancies between data sources, particularly with respect to built up areas, woodland and improved pasture. The authors also observed the misclassification of the land use dataset (in terms of woodland and built up areas) with OS 1:50000 map data. The OS and survey data corresponded well providing a mechanism to modify the errors in the coarse scale land use data. This clearly highlights difficulties in obtaining coarse scale data at large catchment scales and that there exists higher reliability when dealing with smaller compartments of catchments. However, we need to gauge accuracy level required in accordance with the intended use of the model (Jakeman et al., 2006) and we need to convey the degree of uncertainty linked to model output so that end-users (e.g. policy practitioners) can appreciate limitations and not partake in inappropriate model use and abuse.

While the sub-catchment is argued to be equally as important as the catchment scale, there are issues at this scale too. The importance of field FIO sources depends on the timing and extent of faecal deposition and die-off rates (Vinten et al., 2008), and land application of manures in spatially and temporally heterogeneous patterns (Scholefield et al., 2007). Consequently FIO source burdens are spatially and temporally complex and so understanding spatial and temporal distributions of faeces is difficult to gauge at larger scales. The most appropriate scale for understanding and utilising spatial and temporal faecal input data is arguably the farm scale. This is because specific livestock numbers are known through direct collaboration with farm enterprises and farmer co-operation providing unique local detailed knowledge. At the sub-catchment through to catchment scale the distribution of livestock is more difficult to ascertain, complexity is increased through aggregation of a number of farming enterprises. Similarly difficulties exist in understanding all manure applications in space and time. Individual farm surveys can be time consuming when conducted at catchment and sub-catchment scales and confidentiality issues restrict farm by farm details being released across catchment areas. Thus widespread accessible farm information is generally only obtained via the annual Agricultural Survey (formerly Census) which itself is complicated by the fact that data is collected at the level of the farm holding yet the results are used at various aggregate levels. This translates to a series of potential sources of error in the Agricultural Survey, whose magnitude it is necessary to appreciate so as to use the data within acceptable bounds. The key sources of error relate to: (i) missing data in the form of non-respondents or farms that were missed from sampling strategy; (ii) erroneous records in the register of farms with respect to farm numbers and coverage due to shifts in farm enterprise sales and (iii) mismatches in the geo-referencing procedure whereby farm enterprises are allocated to a specific geographical location. Furthermore, the sampling unit for the Agricultural Survey is the farm holding, which in itself is not a geographical unit, but is instead a grouping of fields or units that are farmed together. They are not necessarily geographically one entity and can be dispersed in the landscape across relatively wide areas (or catchment / sub-catchment boundaries) as evidenced in the Taw catchment farmer survey. Survey responses can inform on activities that take place on-farm but they do not shed light with regard to specific locations of activity and this is a limitation which needs to be completely transparent. Geographical questions cannot be answered in a definitive manner because of this caveat.

Geographical estimates can be made as an alternative, for example within 1 km^2 grid areas. The relevance of such data scaled up to 1 km^2 grids is questionable. Figures 3i and 3ii shows the Taw catchment overlaid with a $1km^2$ grid typical of spatial aggregated modelling approaches. If each cell is assigned a land-use class, predominant soil type class and associated Agricultural Survey data it remains highly questionable as to how accurate this aggregation really is. A dominant soil type in a $1km^2$ grid may even only account for 40% of the land area in a spatial grid if for example another three soil types were present in smaller proportions of ~ 20%, and so it is not always clear how useful it is to assign a predominant soil type to a grid cell. In Figure 4 it is evident that farm boundaries do not align conveniently with grid cell delineations and this raises issues relating to the distribution of livestock in catchment scale models as governed by agricultural census data aggregated to the $1km^2$ grid.

Alternatively, Figure 5 shows the result of a mapping exercise conducted with a participating farmer in the Taw survey. This interactive approach allowed for the representation of farm data in a graphical format to better understand the geographical distribution of livestock across a farm boundary and the location of manure spreading activity relative to watercourses and drains. This map format approach was used as part of the RELU project to supplement a more structured farm survey questionnaire as an attempt to resolve some of the issues discussed in terms of identifying specific locations of farm activity. Clearly in the example of the Agricultural Survey, the level of aggregation is of considerable importance in determining the quality of data associated with geographical estimates. In circumstances whereby a few farm holdings are analysed, say at a local level, geographical estimates can be very poor. In contrast, as aggregation increases the proportion of the land that is misspecified reduces.

4. Private jurisdictions: the farmer still matters

As discussed in the second section of this paper, much of the impetus for the catchment approach derives from the perceived problem of overlapping and competing institutional jurisdictions. However, in this section we suggest that 'solving' that particular problem merely through the realignment of spatial policy arrangements is to ignore the importance of spatial units of land management which certainly are no respecters either of policy or natural/physical boundaries. Indeed, blanket approaches to management will not work.

Few would argue with the claim that land occupancy arrangements are important to environmental management. Although the amount of research on precise relationships between occupancy and environmental outcomes has been limited (Winter, 2007), there is a long history of observations of relationships between tenancy and agricultural economic performance (Higgs, 1972; Hill and Gasson 1985; Schickele, 1941). Occupancy arrangements may influence the level of investment in pollution mitigation technologies with owners or secure tenants more likely to be able to invest (c.f. Fish et al., 2009). Tenants on short-term or insecure leases, especially if at a high rent, are less likely to adopt long term benign stewardship as one of their management goals (Winter et al., 1990). Occupancy change itself – a new tenant or a transfer from one generation to the next – has been identified as a particular trigger for environmental change (Munton and Marsden, 1991; Potter and Lobley, 1996). Scales of governance (both spatial and temporal) are not always in-tune with those of individual farm occupiers or the maintenance of the ecosystem itself. Here we seek to build on these insights by looking at the spatiality of occupancy. If units of occupancy (farm holdings) are critical to variation in environmental management and management outcomes, then it follows that the spatial mosaic of holdings and the connectivity between them (or not) is a critical factor that confronts the institutional tidiness of the catchment approach. Farm boundaries may not, indeed almost certainly will not, match catchment or sub-catchment boundaries. Figures 2 and 3 exemplify this for the Taw catchment in North Devon.

In addition, holdings may be spatially fragmented and the land use and management prescriptions on particular fields will be influenced by a whole set of socio-spatial considerations that cannot easily be read off from the underlying physical characteristics. For example a catchment planning approach may identify certain categories of land (for example free draining and distant from water courses) that are best suited to spreading livestock manure. Such plans may even identify that enough land is theoretically present within a specific catchment for specified numbers of livestock. But the feasibility of this 'on the ground' will depend on farmers' ability to comply with the catchment wide plan and this will be influenced by issues such as ease of access (road and farm track lay-out, presence or otherwise of barriers such as woods) and the distribution of different categories of land within a particular farm. Controlling downstream pollution of watercourses clearly requires an understanding of land uses within catchments but also the decision making processes amongst land managers inhabiting the spatial units. Differences in adoption decisions of farmers may reflect very detailed differences in physical farm lay-out and land use, as shown in work in New Zealand on why dairy farmers may or may not chose to fence off streams (Beswell et al., 2007). But differences are also likely to reflect a wider set of decisions, beliefs and attitudes of farmers operating at the farm enterprise level as well as policy and economic constraints. Thus decision making is complex and multi-faceted and attitudes may be highly diverse within a catchment and certainly not always compatible with the dominant norms of spatial planning. Clear examples can be seen in the quotations drawn from our catchment-wide farmer survey shown in Figure 6.

Understanding watersheds requires more than integrating the inherent compartments into one geographical unit. There needs to be an appreciation that understanding catchments in turn requires an understanding of the values and needs of the people associated with the catchment (Allan et al., 2008). It is therefore apparent that the success (or failure) of large scale planning approaches is linked to the extent of active support of those who manage the land on a day to day basis.

Management can be successful at the local level but it requires clear strategies for engagement and knowledge exchange often based on decision-making support and guidance for farmers. At present there are a number of top-down government-led strategies of this kind aimed at limiting contamination of surface waters at the farm scale (e.g. Defra manure management plan; Defra soil management plan; SEPA risk assessment for manures and slurries). These strategies are linked to codes of good agricultural practice (Defra, 2009) and environmental stewardship schemes, and seek to raise awareness of manure spreading strategies and other farm activities that could potentially cause pollution of watercourses. They also help identify land considered most vulnerable for contributing towards sediment and nutrient related watercourse pollution. There are two problems with the current approach. Firstly, there is relatively little by way of sustained 'on the ground' engagement of advisors/officials – or street-level bureucrats (Lipsky 1980) - with farmers in the manner that characterised the war-time and post-war food production campaign (Self and Storing, 1962). Secondly, little attention has focused on microbial risk assessment strategies at the farm level with regard to farm-to-field-to-water pathways. Attention has been focussed primarily on farm-to-fork risk (Havelaar et al., 2007) because of the direct potential impact on human health. Making informed decisions at the field scale is crucial because agricultural land is heterogeneous, and inherent spatial variability in soils and hydrological flow pathways influences the loss of pollutants from land to water, often at sub-field scales (Hewett et al., 2004). So while IRBM involves assessing the bigger picture, it risks ignoring the focus of site specific management issues which can be critical to catchment functioning.

Some examples of critical local differences can be highlighted from the small subset of farms shown in Figure 7, which are located in, and drain, a sub-catchment of the Taw catchment. Detailed surveying of these farms revealed information on land, manure and animal management and this highlighted some critical ‘local’ differences between farms within a relatively small spatial area, and in some cases draining into the *same section of stream reach*. For example, of the seven farms identified here, two of the farms allowed livestock to both drink from the stream and also ford the stream, allowing for potential direct defecation of faeces into watercourses. This can negate any water quality benefits attributed to those farms which restrict livestock access to the stream. Other disparities also exist. For example, three of the seven farms admitted to spreading their human domestic waste to land following the emptying of their septic tank systems. This action is not permitted in the UK, and human waste can contain pathogenic viruses which can be transferred into the receiving watercourses following rainfall. So the same drainage area is potentially impacted by the action of these three specific farm operations and can impact on the water quality draining through farmed land of others. Furthermore, different enterprises (dairy, beef and sheep, beef and dairy) are generating different forms of livestock manure (slurry versus farmyard manure) which accommodate differential properties governing the die-off of microbial contaminants contained within the manures. While one farmer applies slurry to land via injection techniques to limit emissions of odour and ammonia, a neighbouring dairy farmer applies slurry via broadcast methods admitting that ‘I bought up land so that on paper it looked like I had enough land to increase my herd size while still being able to spread the muck at acceptable rates, but in reality I spread to the most convenient fields’ and quipping that ‘I get a thrill when I hear muck splash in the water’. The management approaches adopted on different farms will in turn impact on the risks associated with each farm and their vulnerability for contributing microbial pollutants to water. It is detailed and variegated forms observation and surveillance that produce insights into such local, and instrumental, differences in attitudes, capacities and approach, but these are rarely accessible to policy makers.

Considering the survey in the context of the whole catchment we expect differences in management to be even more diverse. Here we found that 27 of the 75 farms allowed for livestock to ford watercourses with ‘dairy & beef’ farms typically allowing for this at a higher frequency. Sheep farms did not permit any livestock access, not because sheep farmers are better disposed to controlling pollution but because of risk to sheep of being swept away and drowning during storms. A higher proportion of farmers allowed stock to drink from watercourses, rather than permitting full access (77%). In total, 69 % of farmers considered stocking density when grazing fields, with dairy farmers most likely to take this into account when devising management plans (83%). Arable farmers gave the least consideration to stocking density of fields used for grazing.

Of those farms surveyed 47 (63%) had made a farm environmental record as part of a stewardship scheme. Lowest uptake of environmental records was for poultry enterprises (33%) whereas 100% of arable farms had completed such a record. Additionally, 67% (50 of the 75) had a farm waste management plan for cross compliance, and on a farm type breakdown, this was found to be significantly higher for dairy farms (89%). Only 31% of farmers interviewed had come across the term ‘diffuse pollution’ with highest recognition of the term found among dairy farmers. We found that 83% of the farmers surveyed were aware of pollution prevention advice quoted in codes of good agricultural practice (CoGAP). Among dairy farmers this was well above average at 94%.

In total, 56 of the farms (75%) believed that their manure storage capacity was ‘about right’, though among dairy farmers this was lower (61%). Of the 75 farms, 48% stated that they had a strategy in place to prevent overflowing of manure storage and 13% stated they planned to increase storage capacity. Only 54% of those asked said that they inspected their waste storage facilities. Some 24% of farmers admitted that there were times of the year when they needed to spread on frozen ground, and this complements the proportions who thought their storage capacity was too little. Furthermore, 15% spread to poorly drained land and 12% said they need to spread on steep sloping land. Only 11% of dairy farms had over 6 month’s storage available (the minimum requirement if a dairy farmer was in an NVZ). The most frequently cited storage capacity was three months. Worryingly, 16% of dairy farmers admitted to spreading their slurry only when it reached capacity in their stores. Of all surveyed farms, 77% stated that they use buffer strips next to streams. The use of such measures was highest amongst dairy farmers (94%) and lowest among arable farmers (33%). Dairy farmers also accounted for the highest percentage of those who spread manure on fields adjacent to watercourses (83%). Typical spreading rates for different farm types varied considerably (e.g. <3 m^3 ha^{-1} to >61 m^3 ha^{-1}). This synopsis of farm survey data highlights clearly the extent to which differing forms of management – animal, land and manure related – are taking place within the same catchment area.

5. Conclusions

In this paper we have presented evidence designed to disturb the new orthodoxy surrounding catchment planning. Our intention is not to suggest that catchment studies and a policy emphasis on the catchment are inappropriate *per se*. For many purposes this is the right level for analysis and action, but not for all. A similar point has been made by Haygarth *et al* (2005) in an overview of phosphorus transfer and the scientific expertise relevant to understanding the issue in an integrated way. They make a plea for multi-scaled science from the molecule to the catchment. We do not propose a jurisdiction of the molecule. But we do suggest that management at the field and farm scale remains crucial to water quality outcomes. That being the case, the emphasis on catchment planning should not be taken as a reason to avoid a continuing policy emphasis on engagement with land occupiers through regulation, knowledge transfer and participative methods that include farmers.

Of course, delivering on the WFD requires coordination that transcends a continuum of scales. Clearly, multi-scale appreciation is needed; all scales are relevant and research and policy must adopt scale appropriate approaches to addressing the questions raised. The catchment approach has emerged as a powerful new orthodoxy within environmental governance. However, despite its pretence to undermine inappropriate political territorial jurisdictions, it continues to privilege a hierarchial notion of scale. Placing so much confidence in 'natural' delineations of scale, does not eliminate the puzzling issue of reconciling a range of political territorial jurisdictions with competing notions of 'natural' places and spaces. Jurisdictions persist. Nor does the delineation of a catchment eliminate the need for knowledge of sub-catchments and of, what we have termed, private spaces. Larger scale approaches that align with RBMP favour more risk-based strategies to protect and restore whole ecosystems as dictated by the WFD but they are constrained by the complexity of dealing with dynamic socio-biophysical systems, what Watson (2004) see as 'wicked' and 'messy' management problems.

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