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Review

Effects of agricultural land management changes on surface water quality: A review of meso-scale catchment research

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ABSTRACT

Measuring the environmental impacts of agricultural practice is critical for policy formulation and review, including policies implemented to improve water quality. Here, studies that measured such impacts in surface waters of hydrologically diverse meso-scale catchments (1–100 km^2) were reviewed. Positive water quality effects were measured in 17 out of 25 reviewed studies. Successful farm practices included improved landscape engineering, improved crop management and reductions in farming intensity. Positive effects occurred from 1 to 10 years after the measures were implemented, with the response time broadly increasing with catchment size. However, it took from 4 to 20 years to confidently detect the effects. Policy makers and scientists should account for these hydrological and biogeochemical time lags when setting policy and planning monitoring in meso-scale catchments. To successfully measure policy effects, rates of practice change should also be measured with targeted water quality parameters.

1. Introduction

Agricultural management practices that can effectively mitigate against on and off-farm surface water quality degradation have been demonstrated at field (Smith et al., 2001; Melland et al., 2016), hillslope (Freebairn et al., 2009; Sousa et al., 2013) and micro catchment scales (McDowell et al., 2009; Melland et al., 2014; Tomer et al., 2014). In contrast, the effectiveness of farm practice change for water quality improvement at larger scales is less clear (Fenton et al., 2011; Vero et al., 2017). Policy makers need to be informed about the spatial and temporal links between field-scale land management and national-scale water quality in order to develop appropriate policies, to justify expenditure on policy implementation and to promote policy implementation (Roberts & Craig, 2014; Minella et al., 2008; Collins and McGonigle, 2008). Herein, we review the outcomes of studies that have directly measured impacts of agricultural mitigation measures in medium, or meso-scale, catchments (1–100 km^2, incorporating 1st–3rd order streams and representing a scale between farm and river basin scales) over the last 20 years. We use this scale to incorporate the scale of statutory water quality monitoring in rivers while also the link between farm scale and catchment.

Such meso-scale studies are limited in the literature due to the challenging and resource intensive nature of this type of study (Melland et al., 2014). The challenges include the uncertainty in cause-effect relationships due to the complexity of hydrological, climatic, biogeochemical and anthropogenic processes occurring in time and space, and this often results in insufficient collection of water quality and land management information (Cherry et al., 2008). These constraints are compounded by the long periods of time that are normally needed to identify trends and account for time lags in water quality response to, and implementation of, mitigation measures (Meals et al., 2010; Spooner et al., 1987).

When considering hydrological and biogeochemical time lags for nitrogen (N, longer residence times associated with mainly subsurface losses) and phosphorus (P, lower residence times associated with mainly surface losses) within meso-catchments it may not always be possible to document residence times or give detailed data pertaining to e.g. redox conditions. Furthermore, P losses also occur via groundwater and N losses along surface pathways. For the purposes of the present study, permeability, with respect to the soil-subsoil-bedrock continuum, was used as a guide to establish which pathway dominates (Tables S1 and S2). Such a proxy, although not quantitative, can assign dominant pathways of loss, attenuation capacity and highlight if receiving surface waterbodies are dominated by flows derived from surface or

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groundwater (Fealy et al., 2010). For example in meso-scale catchments (Mellander et al., 2014) dominated by imperfect or poorly drained soils the dominant loss pathway will be through surface and shallow subsurface pathways (e.g. lateral movement of infiltrating and shallow groundwater due to low permeability layers such as fragipans or artificial drainage systems) (McDaniel et al., 2008; Shore et al., 2013). In well or excessively-drained equivalents subsurface pathways will dominate but the hydrogeochemistry of the system may vary in terms of dissolved oxygen, electrical conductivity and bacterial energy source availability which in turn may attenuate or enhance nutrient flows via those subsurface pathways. For example, McAleer et al. (2017) examined two well-drained catchments with contrasting subsurface lithologies (slate versus sandstone). Physical factors, including agronomy, watertable elevation and soil-subsoil-bedrock permeability, all influenced the hydrogeochemical signature of the aquifers. Stream nitrate (NO_3^-) load was 32% lower in the sandstone catchment even though agronomic nitrogen (N) inputs were substantially higher than the slate catchment. Therefore, the dominance of surface or groundwater pathways within a catchment and the residence time and geochemistry associated with these pathways must be considered when assessing the efficacy of practice(s) on water quality. In terms of N and biogeochemical lags, soil organic N in the source zone is influenced by the source zone NO_3^- concentration, legacy organic N depletion rate constant, mean annual recharge, soil saturation and soil porosity (Van Meter and Basu, 2015; Ascott et al., 2017 (defined as NO_3^- storage in the Vadose zone)). Outside of the source zone the transformation rate of NO_3^- in the subsurface is important e.g. the denitrification rate in subsoil, subsoil-bedrock interface and in bedrock (Jahangir et al., 2013). In terms of dissolved reactive P it is the chemistry of the soil-subsoil-bedrock continuum and the redox conditions that cause retention or mobilisation of P (Daly et al., 2017). In terms of the subsurface hydrological time lags, which involve mainly dissolved forms of N and P in the unsaturated and saturated zone, parameters such as residence time from the sampling point to the catchment outlet, the physical properties of the underlying aquifer and the overall hydraulic gradient pushing this migration is important (Van Meter and Basu, 2015; Vero et al., 2017). Further complications to conceptual models of nutrient transport can be encountered in groundwater-dominated karst environments where the concentration, load and residence times across different subsurface pathways (conduit versus different fracture sizes) can vary greatly as demonstrated by Fenton et al. (2017) using high resolution loadagraph separation techniques. Acknowledging these conceptual complexities, studies included in the scope of the present review were those that directly measured chemical and/or biological water quality responses in surface water (lakes or rivers) to agricultural practices in meso-scale catchments.

2. Materials and methods

Studies of single, paired and multiple catchments were reviewed, with the latter being included in the review only if the median size of catchment was meso-scale. For each study, a combination of qualitative and quantitative analyses was conducted.

Quantitative analyses included assessments of the response time, the measurement time, the measurement lag (Fig. 1) and the implementation lag. These were defined as:

- Response time was the number of years from when a threshold or maximum rate of implementation of a practice was reported or inferred to have been achieved, to when a (significant) effect on water quality was deduced to have occurred.
- Measurement time was the number of years taken to measure a statistically (or physically) significant water quality response to an agricultural practice and unless otherwise reported, was taken as the total length of the measurement period. This was usually longer than the response time because the initiation of significant water quality effects or trends was only evident or convincing once a longer time series of data was collected. The measurement time was not defined as the sum of the other terms, rather the implementation lag was defined as finishing when the response and measurement times began.
- Measurement lag was the difference between the response time and the measurement time. Measurement lags reflect the extra time needed to measure water quality indicators in order to separate signals/responses from environmental noise and in many cases reflected a period of measurement required before a practice change occurred in order to establish a baseline. In contrast, the response time only started once full/threshold implementation of the practice change was complete.
- Implementation lag was the number of years between the reported or inferred initiation of practice change and when a maximum or threshold rate of implementation was reported, or inferred, to have been achieved.

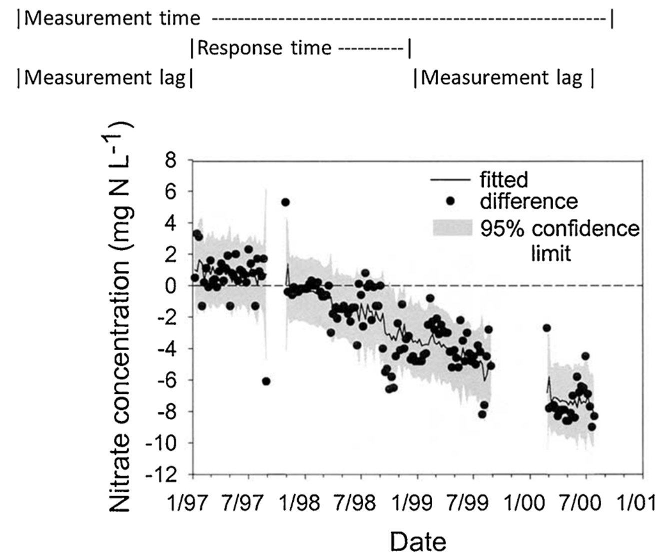

Fig. 1. A 4-year time series of the difference in nitrate concentration between paired treated (4.0 km^2) and untreated (4.9 km^2) subcatchments of the Walnut Creek catchment in Iowa, USA (Jaynes et al., 2004)). A 5-year period of measurements taken to establish similarity between the paired catchments before the practices were implemented is not shown. The practices were assumed to be fully implemented by 1/97 and the practice implementation lag was assumed to be 1 year prior to this. The original figure was modified to highlight when the practices had a significant effect on nitrate concentrations (response time, 2 y post-practice change), the measurement time (5 y pre-BMP plus 4 y post-BMP) and the measurement lag (9 years less 2 years) that were calculated for this review.

Qualitative analyses included summaries of:

- Monitoring approaches used
- Classifications of effects on water quality indicators as positive, neutral or negative
- Classification of positive effects according to the type of hydrological transport pathway most influencing the response of the water quality indicator
- Classification of positive effects according to the type of water quality indicator as chemical (N, P, suspended sediment (SS)) or biological (diatom, macroinvertebrate, macrophyte)
- Classification of drivers of practice change as mostly voluntary, mostly incentivized for research collaboration or mostly mandatory
- Reasons why effects were not measurable
- Reasons why negative effects occurred
- Soil, geology and hydrological flow pathways and residence times.