2-1-1995

Water Resources Studies Along the Arkansas-Oklahoma Border

Kenneth F. Steele
University of Arkansas, Fayetteville

Follow this and additional works at: https://scholarworks.uark.edu/awrctr

Part of the Environmental Indicators and Impact Assessment Commons, Environmental Monitoring Commons, Environmental Studies Commons, Fresh Water Studies Commons, Natural Resources and Conservation Commons, and the Water Resource Management Commons

Recommended Citation

This Technical Report is brought to you for free and open access by the Arkansas Water Resources Center at ScholarWorks@UARK. It has been accepted for inclusion in Technical Reports by an authorized administrator of ScholarWorks@UARK. For more information, please contact scholar@uark.edu, ccmiddle@uark.edu.
Proceedings of the
Arkansas Water Resources Center
1994 Research Conference

Kenneth F. Steele, editor

Water Resources Studies Along the
Arkansas-Oklahoma Border

Arkansas Water Resources Center Publication No. MSC-168

University of Arkansas
Fulbright College of Arts and Sciences

February 1995
PROCEEDINGS OF THE
ARKANSAS WATER RESOURCES CENTER
1994 RESEARCH CONFERENCE

Kenneth F. Steele, Editor
PREFACE

The watershed approach to water resource issues recently has been re-discovered. Scientists and managers recognize the need to consider the entire watershed when delineating and solving today's water resource problems. A simple example is the effect that streams have on the ultimate "health" of a reservoir. Although technically the term watershed should be used only in reference to surface water, the importance of ground water is included in the watershed approach to problems.

In recognition that we all "live upstream and downstream," the Arkansas Water Resources Center and the Oklahoma Water Resources Institute sponsored a conference titled "Water Resource Studies Along the Arkansas-Oklahoma Border," April 12 and 13, 1994 in Fayetteville, Arkansas. The presentation at the conference ranged from studies in northwestern Arkansas and northeastern Oklahoma to those in the Ouachita Mountains and in the Lake Millwood watershed in southwestern Arkansas and southeastern Arkansas. Although these papers covered a wide spectrum of studies in terms of content and geographical setting they are only representative of current and future studies. The conference underscores the need and value of shared scientific results.

The sponsors are grateful to the speakers for their presentations. Many of the speakers submitted abstracts or manuscripts which are published in these proceedings. A copy of the program is printed on the following page.

Kenneth F. Steele, Director
Arkansas Water Resources Center
113 Ozark Hall
University of Arkansas
Fayetteville, Arkansas 72701
**Arkansas Water Resources Center**, University of Arkansas, Fayetteville, Arkansas, In cooperation with **The Center for Water Research**, Oklahoma State University, Stillwater, Oklahoma

**WATER RESOURCE STUDIES ALONG THE ARKANSAS-OKLAHOMA BORDER**
**APRIL 12 AND 13, 1994**

**Conference Program and Speakers**

**April 12**

12:30 p.m. - Opening remarks and introduction by Kenneth F. Steele, Director, Arkansas Water Resources Center, University of Arkansas, Fayetteville, Arkansas.

1:00 p.m. to 4:30 p.m. - Ground water Workshop by Jerry Thornhill of the Robert S. Kerr Laboratory, Ada, Oklahoma.

**April 13**

8:30 a.m. - Opening remarks by Kenneth F. Steele, Director, Arkansas Water Resources Center, University of Arkansas, Fayetteville, Arkansas.

**Moderator: Don Scott**

8:45 a.m. - *The Moores Creek BMP Effectiveness Monitoring Project* by Dwayne Edwards, Department of Biological and Agricultural Engineering, University of Arkansas, Fayetteville, Arkansas.

9:15 a.m. - *Using SIMPLE to Estimate Non-point Source Loading of Phosphorus and Sediment in the Upper Illinois River Basin to Develop TMDL Strategies* by Daniel E. Storm, Department of Biosystems and Agricultural Engineering, Oklahoma State University, Stillwater, Oklahoma.

9:45 a.m. - *Prioritizing Sub-basins of the Illinois River Basin, Arkansas* by David Parker and Rodney Williams, Department of Civil Engineering, and Don Scott, Department of Agronomy, University of Arkansas, Fayetteville, Arkansas.

10:45 a.m. - *Ecological Structure and Functioning of Ozark Plateau Streams* by Art Brown, Department of Biological Sciences, University of Arkansas, Fayetteville, Arkansas.
11:15 a.m. - Protocols for Assessment of Nutrient Limitation in Streams in Eastern Oklahoma, by Dale Toetz, Department of Zoology, Oklahoma State University, Stillwater, Oklahoma.

11:45 a.m. - An Integrated Ecosystem Approach for Assessment of Water Quality Problems Within Lake Tenkiller and Alternatives for Restoration by S. L. Burks, Water Quality Research Lab, Oklahoma State University, Stillwater, Oklahoma, (No paper published).

Moderator: S.L. Burks

1:30 p.m. - Controlling Influences on Ground-Water Flow and Transport in the Shallow Karst Aquifer of Northeastern Oklahoma and Northwestern Arkansas by J. Van Brahana, Department of Geology, University of Arkansas and U.S. Geological Survey, Fayetteville, Arkansas.

2:00 p.m. - Forest Management Effects on Water Quality in the Ouachita Mountains by D.J. Turton, Department of Forestry, Oklahoma State University, Stillwater, Oklahoma, (No paper published).


3:30 p.m. - Resolving Transboundary Resource Conflicts Along the Oklahoma-Arkansas Border by Paul Matthews, Director, Oklahoma Water Resources Research Institute, Oklahoma State University, Stillwater, Oklahoma.

4:00 p.m. - Survival of Pathogen Indicator Organisms in Soil and Transport into Stream Water by Paul Vendrell, Arkansas Water Resources Center, Water Quality Laboratory, University of Arkansas, Fayetteville, Arkansas.

4:30 p.m. - Closing remarks by Kenneth F. Steele, Director, Arkansas Water Resources Center, University of Arkansas, Fayetteville, Arkansas.
# TABLE OF CONTENTS

<table>
<thead>
<tr>
<th>Title</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>THE MOORES CREEK BMP EFFECTIVENESS MONITORING PROJECT</td>
<td>1</td>
</tr>
<tr>
<td>D. R. Edwards, T. C. Daniel, J. F. Murdoch and Paul Vendrell</td>
<td></td>
</tr>
<tr>
<td>USING SIMPLE TO ESTIMATE NON-POINT SOURCE LOADING</td>
<td>7</td>
</tr>
<tr>
<td>OF PHOSPHORUS AND SEDIMENT OF THE UPPER ILLINOIS RIVER BASIN TO</td>
<td></td>
</tr>
<tr>
<td>DEVELOP TMDL STRATEGIES</td>
<td></td>
</tr>
<tr>
<td>Daniel E. Storm, George S. Sabbagh, C. T. Haan, Michael D. Smolen,</td>
<td></td>
</tr>
<tr>
<td>Mark S. Gregory and Dale Toetz</td>
<td></td>
</tr>
<tr>
<td>PRIORITIZING SUB-BASINS OF THE ILLINOIS RIVER BASIN IN ARKANSAS</td>
<td>9</td>
</tr>
<tr>
<td>Rodney Williams, David G. Parker and Don Scott.</td>
<td></td>
</tr>
<tr>
<td>ECOLOGICAL STRUCTURE AND FUNCTIONING OF OZARK PLATEAU STREAMS</td>
<td>15</td>
</tr>
<tr>
<td>Arthur V. Brown</td>
<td></td>
</tr>
<tr>
<td>PROTOCOLS FOR ASSESSMENT OF NUTRIENT LIMITATION IN STREAMS IN</td>
<td>20</td>
</tr>
<tr>
<td>EASTERN OKLAHOMA</td>
<td></td>
</tr>
<tr>
<td>Dale Toetz</td>
<td></td>
</tr>
<tr>
<td>CONTROLLING INFLUENCES ON GROUND-WATER FLOW AND TRANSPORT IN THE</td>
<td>25</td>
</tr>
<tr>
<td>SHALLOW KARST AQUIFER OF NORTHEASTERN OKLAHOMA AND</td>
<td></td>
</tr>
<tr>
<td>NORTWESTERN ARKANSAS</td>
<td></td>
</tr>
<tr>
<td>J. V. Brahana</td>
<td></td>
</tr>
<tr>
<td>A COMPARISON OF MILLWOOD LAKE WATER QUALITY BETWEEN 1974-75 AND</td>
<td>31</td>
</tr>
<tr>
<td>1993</td>
<td></td>
</tr>
<tr>
<td>Christina R. Laurin, Kent W. Thornton and Joe F. Nix</td>
<td></td>
</tr>
<tr>
<td>RESOLVING TRANSBOUNDARY RESOURCE CONFLICTS ALONG THE OKLAHOMA-ARKANS</td>
<td>37</td>
</tr>
<tr>
<td>BORDER</td>
<td></td>
</tr>
<tr>
<td>Olen Paul Matthews</td>
<td></td>
</tr>
<tr>
<td>SURVIVAL OF PATHOGEN INDICATOR ORGANISMS IN SOIL AND TRANSPORT INTO</td>
<td>45</td>
</tr>
<tr>
<td>STREAM WATER</td>
<td></td>
</tr>
<tr>
<td>Paul F. Vendrell, John F. Murdoch, D. C. Wolf, K. A. Teague, T. C.</td>
<td></td>
</tr>
<tr>
<td>Daniel and D. R. Edwards</td>
<td></td>
</tr>
</tbody>
</table>
THE MOORES CREEK BMP EFFECTIVENESS MONITORING PROJECT

D. R. Edwards, Department of Biological and Agricultural Engineering,
T. C. Daniel, Department of Agronomy, J. F. Murdoch and Paul Vendrell,
Arkansas Water Resources Center, University of Arkansas,
Fayetteville, Arkansas

INTRODUCTION

Land application of manures from confined animal production is a subject of increasing concern in Arkansas. Northwest Arkansas water bodies such as Beaver Lake (the water source for approximately 100,000 persons) and the scenic Illinois River are focal points for such concerns because of the value of the water resources and the dense confined animal production in the respective watersheds.

The University of Arkansas Cooperative Extension Service (CES) and USDA Soil Conservation Service (SCS) began work in 1990 in Moores Creek watershed in northwestern Arkansas to decrease losses of animal manure constituents from application sites and thus improve the quality of downstream waters. The Moores Creek watershed was chosen because of documented water quality problems in Lincoln Lake (water supply for the city of Lincoln, supplied by Moores Creek) and the high concentration of confined animal production in the watershed. The bulk of work performed by CES and SCS in the hydrologic unit was to consist primarily of public education and providing technical assistance for implementing Best Management Practices (BMPs), respectively. Cost sharing for selected BMPs was provided by the USDA Agricultural Stabilization and Conservation Service. The Arkansas Soil and Water Conservation Commission and US Environmental Protection Agency subsequently sponsored a monitoring program with the goal of collecting data that would demonstrate the water quality impacts of SCS and CES programs. One component of the monitoring program was to demonstrate the field-scale effectiveness of nutrient management, which was judged to be a key BMP to be installed in the hydrologic unit. This paper reports on the conduct and results of the field-scale monitoring.

PROCEDURE

Two pairs of fields, ranging from 0.57 to 1.46 ha, were identified and instrumented with runoff monitoring equipment (flumes, depth sensors, automated water samplers, and data loggers) that was operational by September, 1991. Fertilizer application to one of each pair of fields was to be conducted in accordance with nutrient management guidelines prescribed by SCS; the other field in each pair was to receive "unmanaged" fertilizer application. The primary
fertilizer sources were to be poultry litter (a combination of manure and bedding material such as rice hulls and wood shavings) for one pair of fields and poultry manure for the other. The cover for all fields was "tall" fescue (*Festuca arundinacea* Schreb). Soils at all fields were predominately loamy in texture, but there were differences in seric clase and textural classes. There were also some differences in cattle grazing practices both between pairs of fields and between fields within a pair.

Preliminary soil sampling indicated that soil phosphorus (P) levels in the upper 15 cm of soil were sufficiently high that no additions of P were necessary for forage production. Optimum nutrient management would thus consist only of adding required nitrogen (N) to the soils. As a result, the anticipated fertilization scheme was modified so that one field of each pair would receive either poultry litter or poultry manure. The fertilization schedules for the two pairs of fields appear in Table 1.

**Table 1. Fertilization schedule for the monitored fields.**

<table>
<thead>
<tr>
<th>Field/Area</th>
<th>Date</th>
<th>Fertilizer Type</th>
<th>Application Rate kg/ha</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td><strong>N</strong></td>
</tr>
<tr>
<td>RU (Unmanaged)</td>
<td>03/15/92</td>
<td>Poultry Manure</td>
<td>332</td>
</tr>
<tr>
<td>1.23 ha</td>
<td>07/13/93</td>
<td>Poultry Manure</td>
<td>451</td>
</tr>
<tr>
<td>RM (Managed)</td>
<td>03/23/92</td>
<td>NH₄NO₃</td>
<td>67</td>
</tr>
<tr>
<td>0.57 ha</td>
<td>08/14/92</td>
<td>NH₄NO₃</td>
<td>67</td>
</tr>
<tr>
<td></td>
<td>04/22/93</td>
<td>NH₄NO₃</td>
<td>116</td>
</tr>
<tr>
<td></td>
<td>07/14/93</td>
<td>NH₄NO₃</td>
<td>136</td>
</tr>
<tr>
<td>WU (Unmanaged)</td>
<td>03/23/92</td>
<td>Poultry Litter</td>
<td>218</td>
</tr>
<tr>
<td>1.06 ha</td>
<td>08/13/92</td>
<td>Poultry Litter</td>
<td>144</td>
</tr>
<tr>
<td></td>
<td>04/13/93</td>
<td>Poultry Litter</td>
<td>158</td>
</tr>
<tr>
<td></td>
<td>07/20/93</td>
<td>Poultry Litter</td>
<td>194</td>
</tr>
<tr>
<td>WM (Managed)</td>
<td>03/23/92</td>
<td>NH₄NO₃</td>
<td>138</td>
</tr>
<tr>
<td>1.46 ha</td>
<td>04/13/93</td>
<td>NH₄NO₃</td>
<td>102</td>
</tr>
<tr>
<td></td>
<td>07/20/93</td>
<td>NH₄NO₃</td>
<td>102</td>
</tr>
</tbody>
</table>
Runoff samples were collected after each runoff event up to April 30, 1994, and analyzed for nitrate N (NO$_3$-N), ammonia N (NH$_3$-N), total Kjeldahl N (TKN), ortho-P (PO$_4$-P), total P (TP), total suspended solids (TSS), and fecal coliforms (FC) according to standard methods of analysis. Soil samples (0-15 cm depth) were collected quarterly from five locations per field and analyzed for pH, organic matter, inorganic N, P, and selected metals.

**RESULTS**

Flow-weighted mean concentrations of analysis parameters are given in Table 2.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Field</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>RU</td>
</tr>
<tr>
<td>NO$_3$-N</td>
<td>0.14</td>
</tr>
<tr>
<td>NH$_3$-N</td>
<td>0.21</td>
</tr>
<tr>
<td>TKN</td>
<td>2.89</td>
</tr>
<tr>
<td>PO$_4$-P</td>
<td>2.25</td>
</tr>
<tr>
<td>TP</td>
<td>2.38</td>
</tr>
<tr>
<td>COD</td>
<td>50.41</td>
</tr>
<tr>
<td>TSS</td>
<td>40.25</td>
</tr>
</tbody>
</table>

For both managed fields (RM and WM), event runoff concentrations of PO$_4$-P exhibited a significant, linearly-decreasing trend with time (Figures 1 and 2). Event concentrations to TP exhibited a significant, linearly-decreasing trend for field RM, decreasing from approximately 3.2 to 1.6 mg/L over the monitoring period. The decreasing trends in P concentrations are attributed to decreases in soil P concentrations, which were in turn a result of no P additions to the managed fields over the monitoring period. Soil P concentrations in field RM decreased from approximately 300 to 193 mg P/kg soil, and field WM exhibited a soil P decrease from approximately 492 to 260 mg P/kg soil. There were no significant trends in runoff P concentrations for either of the unmanaged fields (RU and WU) over the monitoring period. There were no trends in event runoff concentrations N (NO$_3$-N, NH$_3$-N, TKN) for fields RU, WU, and WM, but event NH$_3$-N and TKN runoff concentrations demonstrated significant, linearly-decreasing trends over the monitoring period. Mean event concentrations of NH$_3$-N decreased from approximately 1.4 to 0.4 mg/L, whereas mean event concentration of TKN decreased from approximately 14 to 4 mg/L for field RM over the monitoring period. The reasons for the decreasing NH$_3$-N and TKN runoff concentrations for
field RM are unclear but might be related to residual N near the soil surface at the beginning of monitoring, since the other managed field (WM) evidenced no significant trends in event N concentrations in runoff. Event runoff concentrations of COD decreased from approximately 115 to 50 mg/L for field RM and from approximately 80 to 40 mg/L for field WM over the monitoring period, most likely due to no further additions of organic fertilizer over the monitoring period. There were no trends in runoff concentrations of TSS. Average FC concentrations ranged from 17,000 to 133,000 colony-forming units/100 mL, almost always exceeding both primary and secondary contact standards.

Figure 1. Event runoff PO₄-P concentrations for field RM.
Runoff losses of all fertilizer constituents analyzed (Table 3) were agronomically low, and N and P losses were small proportions of amounts applied via the fertilizers. There was a strong correlation between runoff amounts and fertilizer constituent losses.

Table 3. Runoff (mm) and annual runoff losses (kg/ha) of analysis parameters for the monitored fields.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>RU</th>
<th>RM</th>
<th>WU</th>
<th>WM</th>
</tr>
</thead>
<tbody>
<tr>
<td>Runoff</td>
<td>193</td>
<td>43</td>
<td>61</td>
<td>175</td>
</tr>
<tr>
<td>NO₃-N</td>
<td>0.27</td>
<td>0.43</td>
<td>0.28</td>
<td>3.38</td>
</tr>
<tr>
<td>NH₃-N</td>
<td>0.40</td>
<td>0.20</td>
<td>0.99</td>
<td>1.27</td>
</tr>
<tr>
<td>TKN</td>
<td>5.58</td>
<td>1.58</td>
<td>3.92</td>
<td>6.13</td>
</tr>
<tr>
<td>PO₄-P</td>
<td>4.34</td>
<td>0.66</td>
<td>1.57</td>
<td>2.70</td>
</tr>
<tr>
<td>TP</td>
<td>4.59</td>
<td>0.77</td>
<td>1.99</td>
<td>2.67</td>
</tr>
<tr>
<td>COD</td>
<td>97.39</td>
<td>28.81</td>
<td>48.08</td>
<td>80.39</td>
</tr>
<tr>
<td>TSS</td>
<td>77.62</td>
<td>29.52</td>
<td>68.15</td>
<td>117.33</td>
</tr>
</tbody>
</table>
Event runoff losses of fertilizer constituents were strongly related to the duration between application of fertilizer and first runoff-producing storm, with relatively high losses occurring when runoff-producing rainfall occurred shortly after fertilizer application. Except in the case of COD for field WM (decreasing from approximately 10 to 2 kg/ha), there were no significant trends in event losses of analysis parameters. This was to be excepted in the cases where there were no significant trends in event concentrations of analysis parameters. In the cases where significant trends in event runoff concentrations of analysis parameters were detected, the lack of trends in runoff losses can be explained as due to variability in event runoff amounts.

CONCLUSIONS

Nutrient management, which consisted in this study of replacing organic fertilizer with inorganic N for fields with sufficient soil P for forage production, decreased soil P concentrations and mean event runoff concentrations of P over the monitoring period. Using inorganic N instead of organic fertilizer also translated to decreases in runoff COD concentrations. This work thus provides an example of how fertilizer management techniques can be implemented to improve the quality of runoff from pasture fields. However, there are a myriad of other issues that should be addressed before water quality sustainment can be fully integrated with animal manure management. Topics such as how to determine the limiting animal manure constituent (from a water quality perspective), water quality goals, and circumstances in which to initiate various management strategies require additional investigation to best make beneficial use of animal manure constituents while adequately protecting or enhancing water quality.
USING SIMPLE TO ESTIMATE NON-POINT SOURCE LOADING OF PHOSPHORUS AND SEDIMENTS OF THE UPPER ILLINOIS RIVER BASIN TO DEVELOP TMDL STRATEGIES

Daniel E. Storm, George S. Sabbagh, and C.T. Haan, Department of Biosystem and Agricultural Engineering, Michael D. Smolen, Cooperative Extension Service, Mark S. Gregory, Department of Agronomy, and Dale Toetz, Department of Zoology, Oklahoma State University, Stillwater, Oklahoma

ABSTRACT

This is an ongoing project to conduct a comprehensive inventory of pollutant sources in the Upper Illinois River Basin, located in northeastern Oklahoma and northwestern Arkansas. The project is funded, in part, by the Oklahoma Conservation Commission and the US Environmental Protection Agency. The inventory will provide assistance in the implementation of the Illinois River Watershed Implementation Program, which is part of Oklahoma's Section 319 Management Program. This project is one component of a comprehensive program that addresses the wide range of pollution sources within the Upper Illinois River Basin. The overall goal of the comprehensive program is to improve water quality in the Illinois River, which has been designated as a Scenic River, and to protect Lake Tenkiller Reservoir. The inventory will provide phosphorus and sediment loading estimates, prioritize the loadings, and corroborate the inventory with chemical and in situ biological monitoring. Project results can be used to develop Total Maximum Daily Load (TMDL) strategies.

Non-point sources of phosphorus and sediment are estimated with a watershed-scale computer model called Spatially Integrated Model for Phosphorus Loading and Erosion (SIMPLE). SIMPLE is a UNIX-based continuous simulation model, which utilizes digital terrain modeling and geographic information systems to estimate model parameters. SIMPLE has a menu-driven interface developed using C language and X-window tools to operate on a SUN workstation platform.

Data layers are developed using Geographic Resources Analysis Support System (GRASS). SIMPLE estimates phosphorus and sediment loading on a daily mass balance basis, incorporating effects from rainfall, topography, soil properties, animal waste application, and management practices. Model output includes dissolved and sediment-bound phosphorus, runoff volume and sediment yield on a daily, monthly, or annual basis.

SIMPLE is also being used to identify critical source areas of phosphorus and sediment, and prioritized fields based on potential phosphorus loading to streams.
The Oklahoma Conservation Commission, the USDA Soil Conservation Service and the OSU Cooperative Extension Service will use these results to develop comprehensive management plans for education and cost-share assistance that will aid in implementing best management practices. By using this prioritization scheme, efforts will be concentrated in critical areas, which will result in greater improvement in water quality with limited funding.
PRIORITIZING SUB-BASINS OF THE ILLINOIS RIVER BASIN IN ARKANSAS

Rodney Williams and David G. Parker, Department of Civil Engineering, and H. Don Scott, Department of Agronomy, University of Arkansas, Fayetteville, Arkansas

EXTENDED ABSTRACT

The Illinois River basin has experienced water quality impairment from point and non-point sources of pollution for the past several years. In order to help federal, state, and local efforts address the correction of pollution problems, a project to prioritize the sub-basins in the Arkansas portion of the watershed is being conducted. This prioritization will be on the basis of both water quality measurements and on geographic information system (GIS) modeling.

This paper will concentrate on the results of one year of seasonal water quality measurements in each of the thirty-seven sub-basins in Arkansas during base or low-flow conditions. Figure 1 shows the location of each basin and those basins which contain point source discharges from Publicly Owned Treatment Works (POTW) wastewater treatment plants. The sampling locations were at the outlet of each basin.

Figure 2 shows a ranking of the median phosphate concentrations during the study. It appears that the highest ranked basins (0.1 to 1.0 mg/L) are mostly associated with the drainage from two major point sources in the upstream basins. Figure 3 shows the ranking of the median nitrate-N concentrations during the study. The highest ranked (3.0 to 4.4 mg/L) basins are grouped in two areas and do not appear to necessarily be associated with point sources.

Figure 4 shows the median concentrations of phosphate at sites along the main stem of the Illinois River. The sharp increase in concentration between sites 320 and 520 are a result of the phosphate input from Osage Creek. Figure 5 shows the median concentrations of nitrate at sites along the main stem of the Illinois River. The increase in concentration between sites 140 and 120 and also between 320 and 520 are a result on the nitrate input from the Muddy Fork and Osage Creek.

The contributions to nutrient loads and water quality impairment from non-point sources will be determined by sampling storm water runoff in the sub-basins.
Figure 1. Location of sub-basins and sewage treatment plants in the Illinois River Watershed in Arkansas.
Figure 2. Ranking of phosphate concentrations in the sub-basins.
Figure 3. Ranking of nitrate-N concentrations in the sub-basins.
Figure 4. Phosphate concentrations at selected sites on the Illinois River.
Figure 5. Nitrate concentrations at selected sites on the Illinois River.
ECOLOGICAL STRUCTURE AND FUNCTIONING OF OZARK PLATEAU STREAMS

Arthur V. Brown, Department of Biological Sciences, University of Arkansas
Fayetteville, Arkansas

INTRODUCTION

Streams in the Interior Highlands have moderate slope and moderate abundance of gravel. This produces an alluvial gravel channel form with distinct riffle and pool structure. Riffles occur predictably every 5-7 stream widths in undisturbed gravelbed streams (Leopold et al., 1964). Although early theoretical models of stream ecosystems focused on riffle and pool segments (e.g., Illies, 1958), the most popular modern stream ecosystem model is the river continuum concept (RCC) (Vannote et al., 1980). The representative RCC stream begins in a forested watershed in mountainous terrain as a boulder-cobble, debris-regulated channel which gradually gives way to an alluvial gravel streambed in its mid-reaches and then becomes a large river with an alluvial sandbed channel (Brussock et al., 1985). However, Ozark and Ouachita streams often begin in gravelbed channels in the headwaters. Gradual transition of physical characteristics (width, depth, flow rate, light availability, etc.) described by the RCC that is accompanied by the gradual downstream transition of biotic communities is not very evident in gravelbed streams. Instead, the physical attributes change sharply between each riffle and pool and biota are distributed according to this stronger physical template (Brown and Brussock, 1991; Brussock and Brown, 1991).

The numbers and biomass of most invertebrate taxa are higher in riffle areas than pools (Brown and Burssock, 1991). Invertebrates within riffles show a strong preference for the upstream end (Brown and Brown, 1984). The orderly pattern of functional groups of invertebrates by stream orders (sizes), (Strahler, 1957) as described in the RCC is not obvious in riffle-pool streams (Brussock and Brown, 1991). This is also true for fish (Brown and Matthews, in press). Fish taxa are more properly associated with riffles (darters, madtom catfish, and many minnows) or pools (crappie, bass, gar, carp, larger catfish, etc.) than they are with particular sizes of streams.

STREAM STRUCTURE

Pools are significant barriers to dispersal of riffle-adapted invertebrates in alluvial gravel streambeds. When flow rates exceed 10 cm/sec through entire pools nearly 50% of the invertebrates which drift into pools reach the next riffle (Brussock, 1986). But when flow is less than 10% cm/sec, fewer than 2% reach the next riffle, and more move upstream from the pool back onto the riffle.
According to the RCC, significant numbers of plankton do not occur before eighth or ninth order is obtained, which are large rivers. Slower flow rates and abundant interstitial spaces among gravel particles provide for the production of large quantities of planktonic and benthic meiofauna in gravelbed streams (Brown et al., 1989; Amores, 1991; Richardson, 1989). Although this size class of organisms has long been recognized for its ecological significance in lentic (lake) ecosystems, only within this decade has its importance begun to be realized in lotic (stream) ecosystems (Brown et al., 1989). Meiofauna appear to be very important transformers of very fine particulate organic matter into higher quality food for larval invertebrates and fish (Brown et al., 1989; Amores, 1991). Large woody debris (LWD) is almost entirely absent in the headwaters of Interior Highland streams for some unexplained reason. This is quite atypical for small stream channels (Trotter, 1990) and it may never be known if this is natural because there are no natural, undisturbed watersheds left to examine in this area (Pat Fowler, USFS, personal communication, 1994). Woody debris plays a very significant role in other headwater streams (Molles, 1982; Trotter, 1990). In these other streams, LWD stabilizes stream channels, retains coarse particulate organic matter (CPOM), and provides habitat for all types of biota (Benke and Wallace, 1980). In Interior Highland streams, LWD often has the opposite effect (Brown and Matthews, in press). During floods LWD is floated downstream and may contribute to the scouring action that removes CPOM. Also, when a tree falls into a gravelbed stream, if it falls across a riffle, it narrows the channel, which speeds up the water, and results in removal of the riffle. Subsequently, the stream must redistribute its bedload, particulate organic matter is released to be transported downstream. The leaves that do remain in Ozark streams are utilized very rapidly by microbes and macroinvertebrates (Brown and Ricker, 1982, Petty and Brown, 1982; Brussock et al., 1988).

**IMPACT OF DISTURBANCE**

Physical disturbances such as gravel mining have major impacts on Interior Highland stream ecosystems (Brown and Lyttle, 1992). When a riffle is removed, large volumes of gravel bedload must be moved from upstream to rebuild the riffle. Movement of bedload releases fine sediments, allows entrainment of many invertebrates, fish eggs, and fish larvae, and buries others. As a result streams develop which have wide, shallow channels that favor minnows like grazing Campostoma but provide poor habitat for larger fish species which prefer deep pools.

Our experiments with patch disturbances have revealed that invertebrates recolonize disturbed patches very quickly: within less than one week regardless of patch size (Brown and Lyttle, 1992). The invertebrate communities of Interior
Highland streams appear to be composed of very resilient species, perhaps because they have a history of numerous natural and anthropogenic disturbances (Brown and Brown, unpublished manuscript).

There are few data on stream macrophytes from alluvial streambeds, but their abundance appears to be determined primarily by light availability and disturbance intensity, both within the riffle-pool sequences and along the continuum from the headwaters through middle reaches. Their presence seems to indicate disturbance to riparian vegetation. Macrophytes occur downstream from bridges, but do not grow farther back where the riparian zone is still relatively undisturbed.

Although algal standing crops are generally low, algal production rates are high (Woomer, 1986). Low algal standing crops are the result of intense grazing by fish and invertebrates (Power and Matthews, 1983; Power et al., 1985).

Most plant nutrients (PO₄, NO₃, etc.) enter streams during floods, but during floods the plants are unable to use them. Therefore, inorganic plant nutrients are of little effect and non-point source nutrient loading should be of secondary interest to stream ecologists and environmental biologists in this region. However, it is appropriate for reservoir managers to continue their concern about nutrient loading. There is little, if any, correspondence between levels of plant nutrients and standing crops of planktonic or periphytic algae in the Illinois River (Gakstatter and Katko, 1987; Brown et al., 1991). Apparently algal abundance depends on something other than inorganic plant nutrients, probably grazing by invertebrates and fish.

In this region, we are often quite concerned with the impacts of land application of confined animal wastes. Improper disposal of poultry litter may cause fish kills. Apparently the fish die from low dissolved oxygen that resulted from bacterial respiration driven by excessive dissolved organic carbon (DOC). I have never seen one of these accompanied by excessive growth of algae.

CONCLUSION

In summary, Interior Highland streams have distinct riffle-pool structure which causes them to function differently from others and increases their vulnerability to physical disturbances whether natural, like fallen trees, or anthropogenic, like gravel mining. The streams lack of LWD which results in poor retentiveness of particulate organic matter, yet they are quite productive. Logically, biotic communities of streams in this region must depend on production derived from DOC. The trophic pathway: DOC→Bacteria→Meiofauna→Scrapers, Filter-Feeders, Larval Fish→Larger Fish, must predominate. The best management practices are those which are based on the best knowledge of how systems function. To meaningfully further our understanding of Interior Highland stream ecosystems, we must study the trophic pathway that begins with DOC.
REFERENCES


PROTOCOLS FOR ASSESSMENT OF NUTRIENT LIMITATION IN STREAMS IN EASTERN OKLAHOMA

Dale Toetz, Department of Zoology, Oklahoma State University, Stillwater, Oklahoma

INTRODUCTION

Management practices frequently disturb terrestrial ecosystems with concomitant effects on downstream aquatic ecosystems (Woodmansee, 1984). The impact of these disturbances on lakes and reservoirs, is now reasonably predictable with respect to nutrients (Vollenweider, 1976 and Schindler, 1977). However, it is difficult to gauge the severity of those effects on rivers and streams. In-streams changes in discharge complicate an assessment of the effects of nutrients on the biota (Horner et al., 1983). The purpose of this paper is a general review of methods used to assess nutrient limitation in streams in order to show the usefulness of some methods to characterize the trophic status of streams.

METHODS FOR MEASURING NUTRIENT LIMITATION

There are a number of techniques for measuring nutrient limitation. Generally, methods involving the growth response of algae are viewed as integrating effects of nutrients, light, etc. Until now physiological indicators were more difficult to apply to an assessment of nutrient limitation since algae respond quickly to changing environmental conditions. However, there is growing evidence that physiological indicators and nutrient enrichment bioassays lead to the same conclusion (St. Amand et al., 1989).

Detection methods for P limitation in streams can be grouped into three categories: biomass measurements, nutrient additions (fertilization) and an enzymatic assay. The biomass measures are algal N:P ratios and surplus P. High algal N:P ratios could signal P limitation. Given the fact that N:P occur at a 7:1 ratio in nutrient replete algae, an increase in the N:P ratio in algae might indicate P limitation (Redfield, 1958, Patrick, 1966, and Rhee and Gotham, 1980). The surplus P method takes advantage of the fact that algae store P when not limited by P (Fitzgerald and Nelson, 1968). In this method surplus P is extracted from cells and the quantity of P is measured. Large quantities of P in extracts indicate algae are not P limited.

Nutrient fertilization in stream channels has been used to detect P limitation. Addition of P to a stream channel is done by dripping a nutrient solution from a carboy using replicated artificial or natural stream channels that serve as treatments and controls (Peterson et al., 1985 and Bothwell, 1985). Biomass and/or growth of periphyton is compared between treated and control channels.
The substrate technique involves use of artificial substrates (clay flower pots) which diffuse ions and are point sources of one or more nutrients (Fairchild et al., 1985). After a suitable time biovolume and/or biomass on substrates is determined and treatments are compared to controls. A significant increase of biovolume and/or biomass on treatments over controls indicates limitation by the treatment nutrient.

The alkaline phosphatase activity (APA) technique measures P limitation only (Healey and Hendzel, 1979). It has the basis in that many species of algae have more of the enzyme alkaline phosphatase on cell surfaces when P limited than when not P limited. This enzyme enhances the competitive ability of these species to compete for P by hydrolyzing organophosphates (Berman, 1970; Perry, 1972; Heath and Cook, 1975; Jansson, 1988; Petterson, 1980). It is very rapid and potentially useful to screen many samples (sites).

Based upon research on phytoplankton and algal cultures, Healey and Hendzel (1979) suggest that severe P deficiency in algae occurs when APA is more than 0.005 micromoles per microgram chlorophyll \(a\) per hour. Slight deficiency occurs in the range of 0.003-0.005 (same units) and no deficiency at less than 0.003 (same units). Other similar thresholds are given in terms of other units of biomass: organic or dry weight, ATP, and P content. Gage and Gorham (1985) propose a similar scheme (severe P starvation, warning level, and surplus P accumulation) for phytoplankton.

Since the APA assay is an enzymatic reaction, environmental conditions and reaction time are critical. A buffer is used to maintain an alkaline pH. The pH optimum is different from species to species, hence initial range finding experiments are necessary. Many workers attempt to duplicate field temperatures if the assay is done on natural samples. However, this requirement is not absolute since any reasonable standard temperature can be employed, although it is necessary to keep reaction and measurement temperatures the same (McCombs et al., 1979).

Ideally, the substrate concentration used in the reaction should give maximum enzymatic activity. Often this is possible due to background fluorescence. It is suggested that to avoid this problem the initial concentrations of substrate should be high enough so that no more than 10% is used in a reaction (McCombs et al., 1979). Healey and Hendzel (1979) found that 10 \(\mu\)M MFP supported saturated rates of APA.

**DISCUSSION**

All methods of measuring P limitation have strengths and weaknesses. The biomass methods reflect past concentrations of N and P in the stream and integrate the effects of past current velocity, light, grazing, etc. Thus, it is difficult to determine the importance of interactions. The nutrient addition (fertilization) methods have the advantage of direct manipulation of one nutrient while all other
factors affecting growth are assumed to be constant. In addition, the history of the community being sampled is known in substrate tests, even though it is not identical to that in the stream, since measurements are made of periphyton that have colonized artificial substrata during the period of observation. However, nutrient addition techniques are cumbersome and slow. Further, enrichment with substrata selectively favors only a few species of algae (Fairchild et al., 1985 and others). In the Fairchild et al., (1985) study enrichment favored only 8 of the 46 taxa represented in samples.

Alkaline phosphatase activity (APA) can show P limitation in unialgal culture, but in natural assemblages bacteria can also contribute to APA (Jones, 1976 and Hulett-Cowling et al., 1971), and not all APA is bound to cells (Stewart and Wetzel, 1982 and Wetzel, 1981). APA is also sensitive to extant nucleotides in lake water (Francko, 1984). However, in a recent study in the Glover River, Oklahoma, I was able to show agreement between substrata tests for P limitation, surplus P and APA (Toetz, 1994).

My current research involves use of biological indicators to test for nutrient limitation (Toetz, 1994). The objective is to help water planners prioritize subbasins in the Illinois watershed for abatement of nutrient pollution using biological indicators. APA and surplus P were inversely related in a study of P limitation in eight streams selected to represent a range of nutrient loadings in the Illinois River watershed in Oklahoma. Data analysis is ongoing and will presently produce empirical models to predict which subbasins require management for nutrients.

ACKNOWLEDGMENTS

My research on nutrient limitation has been supported by grants from the Center for Water Research, Oklahoma State University (OWRRI Projects A-120, A-113) and the National Science Foundation, DEB 8012095 and BSR 8514329 and the Department of Zoology, Oklahoma State University.

REFERENCES


Bothwell, M., Phosphorus limitation of lotic periphyton growth rates: An intersite comparison using continuous-flow-troughs, (Thompson River system, British Columbia), Limnol. Oceanogr.,


CONTROLLING INFLUENCES ON GROUND-WATER FLOW AND TRANSPORT IN THE SHALLOW KARST AQUIFER OF NORTHEASTERN OKLAHOMA AND NORTHWESTERN ARKANSAS

J.V. Brahana, Department of Geology, University of Arkansas
Fayetteville, Arkansas

INTRODUCTION

This paper is a brief summary of recent work that is helping refine the understanding of the dominant influences on ground-water flow and transport in the Boone-St. Joe aquifer. This aquifer is a silicious carbonate sequence that occurs near land surface throughout much of the Springfield Plateau in a multistate area of the southern and western Ozarks (Figure 1). Major objectives of this paper are:

• to briefly describe the controlling hydrogeologic influences and the role each plays in ground-water flow and transport in the Boone-St. Joe aquifer; and

• to discuss an updated conceptual model of flow and transport.

Figure 1. Location of shallow karst aquifers of the Springfield Plateau.
DOMINANT FACTORS AFFECTING FLOW AND TRANSPORT

Brahana and others (1988) identified almost 70 factors that affect the hydrogeologic response of carbonate rocks globally. In the geographic area of interest, four general factors appear to be most dominant: 1) lithology and stratigraphy; 2) structural geology and tectonic setting; 3) hydrologic boundaries; and 4) weathering and geomorphology.

Lithology and Stratigraphy

Lithology and stratigraphy are important factors because they define the framework and hydraulic properties, such as primary permeability and porosity, of the sequence of aquifers and confining units. For example, the occurrence and thickness of shales of the Chattanooga confining unit exert a strong control on the vertical interchange of ground water between the Boone-St. Joe aquifer and the underlying Everton and Cotter aquifers. Where the Chattanooga confining unit is absent, ground-water flow systems appear to be well-developed and vertically integrated; where the confining layer is present, the flow systems are effectively isolated. Lithologic control of hydrogeology is also apparent in the relation between percentage of insoluble residues contained within the limestone (such as chert and clay), and the presence of surface and near-surface karst features. Where the Boone-St. Joe aquifer is relatively pure (<10 percent insoluble residues), sinkholes and cavern passages longer than several hundred meters are more likely to occur. Development of surface karst features in northeastern Oklahoma and northwestern Arkansas generally occurs in areas underlain by relatively pure limestone, including: 1) eastern Carroll (Brahana, et al., 1993) and southern Boone Counties, Arkansas (Brahana, et al., 1991); 2) areas of the Springfield Plateau near the Eureka Springs escarpment where the Boone Formation is less than 7 meters thick or where the St. Joe Member of the Boone Formation crops out (Fanning, 1994; Stanton, 1993); and 3) areas where the Batesville Sandstone which overlies the Boone Formation and is less than 4 meters thick (Stanton, 1993; J.D. McFarland, Arkansas Geologic Commission, written communication, 1993). At some sites, sinkholes exist where the Boone Formation is greater than 7 meters thick, but these sites are restricted areally, and commonly associated with major fracturing. In areas where the insoluble residues of the Boone-St. Joe aquifer range from greater than 20 to about 70 percent, the aquifer surface is covered by regolith of variable thickness, and sinkholes, dissolutionally enlarged joints, cavern passages, and other karst features are masked and obstructed by chert and clay. At local to intermediate scales (< 0.5 kilometer to >1.0 kilometer), the continuous chert layers function as local confining units, effectively perching local water levels above the regional water level of the Boone-St. Joe aquifer.
Structural Geology and Tectonic Setting

Structural geology and tectonic setting define features that enhance concentration of flow within the integrated ground-water system. These features include ubiquitous orthogonal joint systems that allow local recharge to reach the deeper, more permeable parts of the flow systems, and faults, which facilitate vertical flow from overlying and underlying aquifers to springs that serve as regional drains. Permeability likely is enhanced where regional faulting is present, and large springs and dissolution landforms commonly are concentrated along major faults (Fanning, 1994).

In addition to the brittle fractures that facilitate vertical movement of ground water in this sequence of aquifers and confining units, the orientation of the distinct lithologic units in near-horizontal layers effectively concentrates most of the lateral flow in the Boone-St. Joe aquifer along bedding planes (Stanton, 1993). Dips are typically less than one degree, and the preferred flow paths along bedding planes are oriented favorably in the phreatic zone to form continuous cells from points of recharge to points of discharge.

Hydrologic Boundaries

Hydrologic boundaries define gradients and control flow and transport in the Boone-St. Joe aquifer. Springs are natural point-discharge sites from ground-water flow systems. Springs integrate flow areally and temporally throughout the region; in some cases, where the springs are localized astride faults, the springs integrate flow stratigraphically. Rivers generally serve as flow boundaries, and typically act as drains from the ground-water flow systems.

Comparison of stream stage, precipitation, and continuous ground-water level data provides valuable insight into integrated aquifer response. The normal range of seasonal water-level fluctuations (non-pumping) exceeds 15 meters at some locations; velocities range from centimeters per day to meters per second (Stanton, 1993). Velocities are lowest in the regolith and the clay-choked bedding planes of the Boone Formation (Stanton, 1993), and fastest in the open conduits of the St. Joe Member of the Boone Formation (Fanning, 1994). Particularly during the non-growing season, vertical recharge commonly exceeds lateral ground-water flow following intense storms, resulting in water-level rises of several meters. During the growing season, evaporation and transpiration from the unsaturated zone capture almost all of the recharge, and water-level rises are rare (Stanton, 1993).

For the most part, ground-water divides are coincident with surface-water divides, and interbasin diversion of ground water by karst piracy is uncommon. Regional and intermediate-scale potentiometric maps of the Boone-St. Joe aquifer provide a sound approximation for estimating flow directions; on a site-specific scale, the non-homogeniety and anisotropy of the aquifer require refined definition to identify and monitor specific flow paths (Stanton, 1993).
Weathering and Geomorphology

Weathering and geomorphology define a set of near-surface physical and chemical processes. These factors control breakdown of rock to soil, which affects the resulting regolith thickness, infilling of the evolving aquifers by insoluble sediments, vertical unloading due to erosion, and horizontal unloading due to escarpment retreat. Examples of these factors are present at research sites near the Eureka Springs escarpment and along the Buffalo National River, where ground-water conditions and karst features become strongly anisotropic and nonhomogeneous. Examples of these factors away from escarpments under more isotropic and homogeneous conditions include outcrop areas of the Boone aquifer where regolith thickness exceeds 17 meters. Although underlain by a variably-developed, generally poorly-defined, incipient karst aquifer, the regolith overlying the Boone Formation acts as a porous water-table aquifer that slowly releases water to recharge the underlying aquifer. The hydrogeology of areas where regolith is thick may not be as sensitive to specific land-use practices as those areas where regolith is thin or absent.

CONCEPTUAL MODEL

A refined conceptual model of ground-water flow and transport integrates the four general controlling influences into eight questions (Figure 2). The hydrogeologic controls included in this model are intended to simplify the complex physical system into factors that can be considered for resource management. Specific environmental and engineering problems require site-specific studies, but in general, this model is intended to provide a cost-effective preliminary assessment to determine relative environmental risk at a site. The questions are:

- Presence of Chattanooga Shale?
- Purity of carbonate unit (percentage)?
- Karst features at land surface?
- Thin cover (<5 meter) lithology overlying pure carbonate?
- Proximity to major fault, joint, or lineament?
- Proximity to major spring?
- Regolith thickness?
- Proximity to Eureka Springs escarpment?
In general, the absence of the Chattanooga Shale, the more pure a carbonate unit, the presence of karst features at land surface, the thinner the cover overlying a pure carbonate, the shallower the depth to the St. Joe Member of the Boone Formation, the closer the distance to a major fault, joint, or lineament, the closer the distance to a major spring, the thinner the cover of regolith, and the closer the distance to the Eureka Springs escarpment, the more environmentally sensitive the area of the Springfield Plateau.

CONCLUSION

This and other ongoing studies focusing on dominant hydrogeologic factors are providing improved conceptual models of flow and transport that are the basis for hydrogeologic quantification and numerical modeling. These studies provide an empirical data base, that when coupled with a systematic water-quality sampling program, may serve as a valuable tool to assess the impact of land-use and waste-management practices on ground-water quality in northeastern Oklahoma and northwestern Arkansas.
ACKNOWLEDGEMENT

The author gratefully acknowledges the contribution of University of Arkansas graduate students Greg Stanton, Bobby Fanning, and Dorothy Walters, and Professor Ken Steele, of the Arkansas Water Resources Center; this paper draws heavily on their data and suggestions. Interested readers are referred to Stanton (1993), Fanning (1994), Walters (1994), Brahana, et al. (1991), and Brahana, et al. (1993), for more detailed presentation of data and interpretations for related studies in the area.

REFERENCES


Walters, D.S., Methods and procedures for recovering host-specific viruses from karst groundwaters to indicate fecal contamination by specific species: Geological Society of America Abstracts with Programs, v. 26 no. 1, p. 29-30, 1994.
A COMPARISON OF MILLWOOD LAKE WATER QUALITY BETWEEN 1974-75 AND 1993

Christina R. Laurin, Kent W. Thornton, FTN Associates, LTD, Little Rock, Arkansas and Joe F. Nix, Ross Foundation, Arkadelphia, Arkansas

This paper summarizes a water quality study of Millwood Lake in southwest Arkansas conducted by FTN Associates, Ltd., with Ouachita Baptist University (OBU) for the Arkansas Department of Pollution Control and Ecology (ADPCE). It answers the following questions: 1) what is the current trophic state of Millwood Lake; 2) has the trophic state changed since the National Eutrophication Study in 1974; 3) what are the sources contributing to this trophic state; and 4) have these sources changed?

Millwood Lake is a Corps of Engineers impoundment located in southwest Arkansas (Figure 1). The lake covers portions of Hempstead, Howard, Sevier, and Little River Counties. Millwood Lake receives drainage from approximately 11,000 km². Approximately half of the watershed is located in Oklahoma. The lake itself is small in relation to the size of the watershed (118 km² surface area) and shallow (2.1 m mean depth, 10 m maximum depth).

Figure 1. Location map showing Millwood Lake in Southwest Arkansas.

31
METHODS

During the 1993 water year, OBU conducted routine water quality sampling at four stations in Millwood Lake as well as four stations on tributaries to the lake. The lake stations were located in the Little River arm of the lake near Yarborough Landing, in the Saline River arm of the lake near Cottonshed Landing, in the Mine Creek arm of the lake near Okay Dike and near Millwood dam. The routine tributary stations were on the Rolling Fork, Cossatot, and Saline Rivers and Mine Creek. ADPCE has a routine water quality monitoring station on the Little River near Horatio, so these data were used from that tributary. There were 9 routine sampling events under a variety of limnological conditions.

Data collected by the Environmental Protection Agency (EPA) during the 1974 National Eutrophication Survey (NES) study of Millwood Lake were used as a baseline for comparison with the 1993 data. EPA conducted monthly monitoring at 16 tributary stations and collected spring (March), summer (June), and fall (October) samples at three of the four Millwood Lake stations monitored in 1993 - Little River and Mine Creek arm stations and the dam station (EPA, 1977).

In-lake water quality data during March, June and October from the 1993 study were compared for the same stations sampled in the NES. The NES nutrient budget which was based on normalized tributary flows, was compared to loadings calculated from mean nutrient concentrations and total tributary flow measured during the 1993 water year. For the Rolling Fork, Little and Saline Rivers, flows measured at the US Geologic Survey gaging stations were very similar to the normalized flows from the NES. Flows for the Cossatot River and the laterals (minor tributaries including Mine Creek, major tributaries downstream from the gaging stations, and immediate drainage to the lake) were estimated using runoff coefficients based on measured flows, and were also similar to the normalized flows from NES.

COMPARISONS

Trophic State

Table 1 lists mean values measured in the lake for trophic state indicator parameters. Using these values, the Carlson's Trophic State Index (Carlson, 1977, as modified by Walker, 1985) for the lake during each study has been calculated. The criteria shown are based on Carlson's Trophic State Index for a eutrophic lake classification.
All parameters for both studies are similar, and in the eutrophic range. The mean chlorophyll \(a\) during the 1993 study is less than in 1974. The lower chlorophyll \(a\) (indicating lower algal productivity) does not appear to be caused by decreases in nutrient concentration. Mean nutrient concentrations were similar in 1993 and 1974, but chlorophyll \(a\) concentrations were lower in 1993 at all three of the lake stations, and during all three sampling events. Mean Secchi transparencies were less at all three stations during all three sampling events. Millwood Lake water is highly stained from lignins/tannins leaching from standing timber, and has low light penetration. Sediment loads in 1993 were significant (approximately 56,000,000 kg/yr) and further reduced transparency. Lower algal productivity in 1993 appears to be related to light limitation, rather than nutrient limitation or changes in N:P ratios.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Eutrophic Criteria</th>
<th>1974</th>
<th>1993</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chlorophyll (a)</td>
<td>10 - 30 (\mu g/L)</td>
<td>15 (\mu g/L)</td>
<td>8 (\mu g/L)</td>
</tr>
<tr>
<td>Total Phosphorus</td>
<td>30 - 65 (\mu g/L)</td>
<td>47 (\mu g/L)</td>
<td>48 (\mu g/L)</td>
</tr>
<tr>
<td>Secchi Transparency</td>
<td>1.5 - 0.7 m</td>
<td>0.8 m</td>
<td>0.6 m</td>
</tr>
<tr>
<td>Carlson's TSI</td>
<td>53.0 - 65.0</td>
<td>60.0</td>
<td>59.4</td>
</tr>
</tbody>
</table>

### Table 2. Nutrient Loads (kg/yr) to Millwood Lake

<table>
<thead>
<tr>
<th>Sources</th>
<th>Total Phosphorus</th>
<th>Total Nitrogen</th>
</tr>
</thead>
<tbody>
<tr>
<td>Point Sources</td>
<td></td>
<td></td>
</tr>
<tr>
<td>((% \text{ of total}))</td>
<td>17,590 (7%)</td>
<td>65,541 (15%)</td>
</tr>
<tr>
<td>Nonpoint Sources</td>
<td></td>
<td></td>
</tr>
<tr>
<td>((% \text{ of total}))</td>
<td>242,715 (93%)</td>
<td>360,356 (85%)</td>
</tr>
<tr>
<td>Total</td>
<td>260,305</td>
<td>425,897</td>
</tr>
</tbody>
</table>

* Calculated using normalized flows
Figures 2 and 3 illustrate the changes in the loads of phosphorus and nitrogen to the lake and indicate the nutrient loads by source. Total phosphorus and nitrogen loads to Millwood Lake are approximately 40% and 20% greater, respectively, in 1993 than the loads reported in the NES. The relative contribution of non-point source nutrients increases with increases in watershed size; nutrient loads from the Little River > laterals > Cossatot River > Saline River. The relative contributions from each source were fairly similar in 1974 and 1993.

Figure 2. Total phosphorus loading to Millwood Lake in 1974 and 1993.

Figure 3. Total nitrogen loadings to Millwood Lake in 1974 and 1993.
The percent change in areal nutrient loadings for these four basins, 1974 compared to 1993, are shown in Figure 4. Areal nitrogen and phosphorus loads have increased in the Little River and Cossatot River basins. Areal nitrogen loads were higher in 1993 in the Saline River basin. Areal phosphorus loads were greater in 1993 in the Mine Creek and laterals basin.

![Figure 4. Percent change in annual non-point source areal nutrient loadings by basin.](image)

The source of the dramatic change in the areal nutrient loading from the Cossatot River basin is uncertain. In the mid 1980s extensive harvesting of timber took place in this basin. This might account for part of the change. There has also been a dramatic increase in swine production in Sevier County; from 3,000 swine in 1987 to 46,000 swine in 1991. The Cossatot River basin is almost entirely in Sevier County, so this might account for part of the change.

The changes in areal loadings from the Little River basin are not as dramatic as those for the Cossatot River basin, however, the Little River contributes the largest portion of inflow and nutrient loads to Millwood Lake. A small change in this large load could have a greater potential for effecting Millwood Lake water quality than a greater change in the relatively small load from the Cossatot River.
CONCLUSIONS

Answers to the questions asked at the beginning of this discussion are:
Millwood Lake is eutrophic.
The 1993 trophic state of Millwood Lake is similar to its 1974 state.
Non-point loadings are the primary sources contributing to
Millwood Lake's trophic state.
Non-point source nutrient loadings to Millwood Lake were greater
in 1993 than they were in 1974.

The Little River watershed contributes the greatest proportion of the nutrient
and sediment load to Millwood Lake. Any changes that occur in its basin
and/or water quality have a greater potential for affecting the quality of
Millwood Lake than the other catchments. Because the Little River subbasin
straddles the Oklahoma-Arkansas border, joint management of the Millwood
Lake watershed is necessary to maintain Millwood Lake water quality.

REFERENCES

U. S. Environmental Protection Agency, Report on Millwood Reservoir, Hempstead, Howard,
Environmental Protection Agency, Corvallis, OR, 1977.

Carlson, R. E., A Trophic State Index for Lakes, Limnology and Oceanography, 22(3), 361,
1977.

Walker, W. W. Jr., Empirical Methods for Predicting Eutrophication in Impoundments, Report 3,
Phase II: Model Refinements, Technical Report E-81-9, U.S. Army Engineer Waterways
Experiment Station, Vicksburg, MS, 1985.
RESOLVING TRANSBOUNDARY WATER CONFLICTS ALONG THE OKLAHOMA-ARKANSAS BORDER

Olen Paul Matthews,
University Center for Water Research, Oklahoma State University,
Stillwater, Oklahoma

INTRODUCTION

Because political boundaries and watershed boundaries are not the same, conflicting management goals can create tensions between states that share a watershed. Additional conflicts occur because states and the federal government have concurrent jurisdiction over water management. Problems are inevitable, and the legal system has ways of resolving them.

Three basic legal solutions exist: legislation, litigation, and negotiated agreements. At times, administrative solutions are included as a separate category (Matthews, 1988). Individual states may legislate, but the impact on activities outside their boundary is limited. Federal legislation preempts contradictory state laws and can impose uniform requirements across state boundaries. Legislation, therefore, is a federal solution where transboundary problems are concerned (Grant, 1983). Litigation between states occurs, and the Supreme Court has often decided such cases (Arkansas v. Oklahoma, 1992). States and individuals also become involved in litigation with federal agencies (Champion International Corp. v. EPA, 1988). Often the preferred solution is negotiation leading to a contract or an interstate compact between states. Even if an agreement is made, enforcement of the agreement or unclear terms may lead to litigation (McCormick, 1994).

In this paper I examine how such conflicts are resolved by discussing ownership and jurisdiction over water, and by explaining the legal implications of water quality conflicts between upstream and downstream states. The recent conflict between Arkansas and Oklahoma over Illinois River water quality illustrates the problem.

OWNERSHIP AND JURISDICTION OVER WATER

Private Water

Private water exists because the federal or state governments have chosen not to exercise jurisdiction over water in some parts of the hydrologic cycle or because the state has given water "ownership" to an individual. Water that is owned is still subject to state jurisdiction, and rights may be lost if water is used in ways that adversely impact others.
In most western states diffused surface water is considered private and is unregulated. In some states small lakes or springs may also be private. Diffused surface water is surface runoff before it reaches a watercourse. A watercourse has a bed, banks, mouth, and a reasonably regular flow (State v. Hiber, 1935). Similarly, soil moisture is ignored. Private water like this is generally usable only by the property owner and is considered to be part of the incidents of land ownership. Although diffused surface water can be captured, which increases the "yield" of a particular owner and results in conflicts with those downslope, this water cannot reasonably be captured by non-owners without trespassing. Historically in the eastern states riparian rights were attached to riparian land in a similar way, but these rights are increasingly being regulated by the states (Dellapenna, 1994).

Oklahoma, by statute, recognizes private ownership of ground water. The way ground water is used, however, is controlled by state law in the same way as local governments control land use. Not all states recognize private ownership of ground water. Atmospheric water ownership is also problematic, but some ownership recognition has been given to those who increase yield and capture it (Davis, 1978).

**State Jurisdiction**

Western states often claim ownership of water as a state or that water is held in trust for the people of the state. Although state ownership of water has been termed a legal fiction, as a minimum "ownership" means the states have concurrent jurisdiction along with the federal government to regulate water use (Sporhase v. Nebraska, 1982). Western states regulate water use in two ways. First, individuals are given rights to use water under the appropriation doctrine. Secondly, the state or public can control access for recreation on navigable streams, retains ownership of the beds of navigable streams, controls water quality, sets minimum stream flows, and has other rights associated with the "public trust." Eastern states have similar jurisdiction, but exercise it without first claiming ownership.

The broadest of these is the public trust doctrine, because citizens can use the concept to force the state into actions they would not do otherwise. For example, under state law, California had granted Los Angeles water rights that led to a decrease in the level of Mono Lake. The environmental harm that resulted was acceptable under state statutory law, but by using the public trust doctrine the Court recognized a public right better than the private right of Los Angeles. This right would protect the environment. The public trust doctrine sets limits on what the state may do, by reserving to the people of the state, power which cannot be superceded by the state (National Audubon Society v. Superior Court of Alpine County, 1983).
Federal Jurisdiction

States' claims to ownership of water are a way of trying to make water theirs, exclusively. But, the federal government has had a role in water management with regard to navigation since the early 1800s. Even on allocating consumptive uses, a federal shadow has been present through federally funded projects under the Reclamation Act and other federal water projects. Congress has usually recognized state primacy in water allocation, and 37 federal statutes specifically say federal laws are not meant to interfere with state laws (Sporhase v. Nebraska, 1982).

Because water use in one part of the hydrologic cycle can have impacts on other parts, the federal deference to state law was politically sound but impractical in application. Federal control over navigation has been asserted from the beginning of our constitutional history, and interfering with navigation is unacceptable. When large scale hydropower dams became feasible, the federal government asserted "exclusive" power over their licensing. States could require additional licenses but could not stop a federally approved project (Federal Power Commission v. Oregon, 1955). Failure of states to establish adequate water quality laws led to the Federal Water Pollution Control Act and the Clean Water Act. Federal water quality laws allow states to administer their own programs, but federal approval is required and federal minimum standards are set. In the early 1980s the pretense of limited federal power over water was put to rest when the Supreme Court determined water was an article of commerce (Sporhase v. Nebraska, 1982). If Congress chooses to regulate water in any way, conflicting state laws are preempted.

Even with water quantity, federal power has been present. When Congress created Indian reservations enough water was set aside to accomplish the purposes of the reservation. This has been interpreted to mean enough water to irrigate all the "practically" irrigable acres on the reservation (Arizona v. California, 1963). In addition to these reserved rights, Congress set aside additional water on federal land such as the national forests (United States v. New Mexico, 1978). These federal rights were created when the reservations were established and have priority over subsequent allocations created under state law.

Jurisdictional Confusion

As can be seen above, water can be "owned" as an incident of land ownership, or a private right to use water can be established under state law. These private property rights are protected from unconstitutional takings but are subject to state and federal laws. State laws traditionally controlled water allocation, but a federal presence has always precluded exclusive state jurisdiction.
Recent Supreme Court decisions recognize federal power, which although politically infeasible, could from a legal perspective be exclusive. Because Congress has chosen not to regulate all aspects of water, states retain jurisdiction as well and can exercise it as long as they are not in conflict with federal law. Also, the power of states is limited by the public trust doctrine.

The conflicts resulting from simultaneous ownership or jurisdiction can be resolved by applying rules of law. The rules are often easy to determine but difficult to apply. Application becomes even more complex when a watershed is divided by a state boundary. The states, as equal sovereigns, have interests in managing the same water in different parts of the hydrologic cycle. Jurisdiction over water is exercised sequentially however, and the upstream state's actions have impacts downstream. With water quality this situation is further complicated by the overlay of federal law. The material below discusses how such conflicts are resolved and focuses on recent court decisions.

WATER QUALITY CONFLICTS BETWEEN STATES

The Courts and State Conflicts

Before federal legislation provided a statutory framework for resolving conflicts between states, courts used the common law doctrine of nuisance. In 1906 the Supreme Court settled its first interstate water pollution case (Missouri v. Illinois, 1906). In order to improve water quality in Lake Michigan, which was the source of Chicago's water supply, waste water discharges were diverted to the Illinois River. The Illinois River joins the Mississippi just upstream from St. Louis. Missouri claimed the change in Chicago's discharge site was contributing to pollution in the Mississippi and causing health problems for its citizens. In the resulting law suit, Missouri was unable to convince the Court that Illinois should be held responsible. Missouri's contributions to the Mississippi's pollution, made it difficult to assign blame to Illinois alone. Although a legal mechanism for resolving such disputes was in place, proving harm and balancing benefits against burdens, made alleviating pollution difficult.

Until the 1970s litigation remained the principal means of resolving interstate conflicts. With the passage of the Clean Water Act Amendments of 1972 the ability to litigate under federal common law was questioned (Milwaukee v. Illinois, 1981). A suit was brought by Illinois against Milwaukee to prevent discharges into Lake Michigan where currents pushed the polluted waters south into Illinois. In the original case the Supreme Court recognized the use of federal common law, but a few months later the 1972 Amendments were passed (Illinois v. Milwaukee, 1972). Milwaukee then claimed the 1972 Amendments were a total restructuring of water
pollution law at the federal level and federal common law had been preempted. The Supreme Court agreed, ruling the statute provided states an adequate opportunity to seek redress against actions by neighboring states. When Congress acts, federal common law can be superceded.

The question of whether state nuisance laws were preempted was left open, but not for long (International Paper Co. v. Ouellette, 1987). A dispute arose because a New York paper mill was discharging effluent into Lake Champlain through a discharge pipe that ended just short of the New York-Vermont border. Property owners in Vermont sued using Vermont's common law of nuisance. The Court decided the Clean Water Act preempted the common law of the affected states but not that of the originating state. If New York's nuisance law had been used in the suit, the Court would have accepted it. Since Vermont and New York laws are similar, the results would most likely be the same no matter which state's laws were used. The next challenge was related to the authority of a downstream state to enforce its water quality standards on an upstream state.

Arkansas Vs Oklahoma

In 1985 Fayetteville applied for a permit on its new sewage treatment plant (Arkansas v. Oklahoma, 1992). The EPA granted the permit allowing a discharge of 6.1 million gallons with half released into the White River watershed and half into the Illinois River watershed. Thirty-nine miles downstream the discharge into the Illinois watershed reaches the Oklahoma border. Although overflow protection exists for the system, a 1988 release resulted in a fine to Fayetteville by the EPA for water quality violations.

Across the border in Oklahoma, the Illinois River had been proposed for inclusion under the federal Wild and Scenic Rivers Act when it was passed in 1970. Local interests did not want federal protection and designated the river instead as a state scenic river. The river is heavily used for recreation with over 3,000 commercial use permits in existence. Within the Illinois watershed on both sides of the border, intensive agricultural uses from poultry operations and plant nurseries contribute to non-point source problems. Several small cities also discharge into the river. Because Oklahoma has designated the river as scenic, the water quality standards include a non-degradation provision.

When Fayetteville applied for a NPDES permit, Arkansas did not have an approved water quality plan, making the EPA responsible for issuing the permits. The EPA issued the permit with specific limitations on the quantity, content, and character of the discharge. Oklahoma challenged the permit's issuance claiming the "no degradation" standard would be violated by the addition of Fayetteville's discharge. The issue was brought before an
Administration Law Judge (ALJ), was appealed to EPA's Chief Judicial Officer, and returned to the ALJ. The result of these administrative proceedings was a determination of "law" and a finding of facts. Oklahoma's water quality standards were applicable to the Fayetteville permit because they had been approved by the EPA. The no-degradation standard was interpreted to mean no "detectable" violation. When applying this standard, the ALJ concluded no detectable violation would occur, and the permit should be issued.

The appeal from the EPA went to the 10th Circuit Court of Appeals where Arkansas argued, upstream states need not comply with a downstream state's water quality standards, and Oklahoma argued against the EPA's determination of no detectable violations (Oklahoma v. EPA, 1990). The Court upheld EPA's requirement for considering a downstream state's water quality standards and went on to conclude that the EPA was wrong in determining no violation of Oklahoma's standards. Because Fayetteville would discharge material which would actually reach Oklahoma, the no degradation standard must be violated. Courts give substantial deference to agency decisions and overturn them only when the agency is arbitrary and capricious. The Circuit Court held the agency was arbitrary and capricious because Oklahoma standards had been misapplied and expert testimony ignored. The convincing argument was -- anything added to an already degraded river must increase pollution and would violate the "no degradation" standard.

Although this argument is convincing from a philosophical view, evidence had been introduced to show no harm would result from the additional discharge, and in fact the added water might dilute the existing contaminants. On appeal, the Supreme Court agreed the statute required compliance with a downstream state's water quality laws, but reversed the earlier decisions requirement for no additional discharge on three grounds (Arkansas v. Oklahoma, 1992). First, agencies are given substantial deference in interpreting standards. In this case the EPA interpreted "no degradation" to mean nothing "detectable or measurable" which the Court termed reasonable in light of the purposes of the Clean Water Act. Second, the Court stated that the Appeals Court had substituted its own factual findings for those of the EPA. Agencies at the original hearing make initial findings of facts which will be upheld if substantial evidence supports them. Just because alternative facts are plausible does not mean agency findings should be reversed. Third, the Appeals Court based its determination of arbitrary and capricious on the two points above. But, a court cannot substitute its own interpretation of law and facts when the agency's interpretation of law is reasonable and the findings of fact are based on substantial evidence.
DISCUSSION

The Arkansas v. Oklahoma decision makes several points very clear. NPDES permits must comply with the approved water quality standards of a downstream state because those standards become federal law when approved. Although the downstream state does not have a veto on an upstream state's permits, the EPA can condition permits to comply with the standards. Administrative discretion in decisions will be overturned if arbitrary and capricious, administrative findings of fact bind later decisions if there is substantial evidence, and courts will defer to agency interpretations of their own regulations. In this case both sides partly won and lost. This seems to be a correct interpretation of interstate water quality law. Oklahoma can have enforceable water quality standards, but the Fayetteville permit did not violate these standards.

Recently, interstate water quality law has taken another twist. In 1987 the Clean Water Act was revised so that Indian tribes may be treated as states and establish water quality standards (Albuquerque v. Browner, No. 93-82-M Civil, 1993). The EPA approved water quality standards for Isleta Pueblo in New Mexico. Albuquerque, which is just upstream, challenged the standards because the city's NPDES permits must comply. The Federal District Court in New Mexico upheld the EPA's authority which was clearly granted by statute. This case is now on appeal. At present, several tribes in Oklahoma that do not have reservations, have asked the EPA to approve water quality standards for them. If these are approved an additional "player" may enter the game along the Arkansas - Oklahoma border (Chandler, 1994).

REFERENCES


Oklahoma v. EPA, 908 F. 2d 595 (10th Cir.), 1990.

Sporhase v. Nebraska, 102 S. Ct. 3456, 1982


SURVIVAL OF PATHOGEN INDICATOR ORGANISMS IN SOIL AND TRANSPORT INTO STREAM WATER

Paul F. Vendrell and John F. Murdoch, Arkansas Water Resources Center, D. C. Wolf, K. A. Teague, and T. C. Daniel, Department of Agronomy, and D. R. Edwards, Department of Biological and Agricultural Engineering, University of Arkansas, Fayetteville, Arkansas

ABSTRACT

Pathogen indicator bacteria (fecal coliform) are above levels acceptable for swimming in many of the Northwest Arkansas streams. The potential for fecal bacteria to contaminate water is determined by survival in soil and transport into water bodies. A laboratory study was conducted to determine the affects of temperature on the die-off rates of fecal coliform and coliphage in soils amended with poultry litter, Escherichia coli, and sterile water. Soil (Captcha silt loam) was amended with E. coli culture at $1.32 \times 10^3$ CFU/g dry soil. Poultry litter was applied at a rate of 0.1 g/g dry soil. All treatments were incubated at 5 and 35°C following amendment and initial moisture adjustment to -0.03 MPa. Fecal coliform and coliphage numbers were determined at time intervals and first order kinetics applied. Rates indicate that increased temperature enhances die-off of both bacteria and virus in the study soil. When incubated at 35°C fecal coliform populations were reduced by 99.9% within 9 days after soil amendment with E. coli culture. However, at 5°C it took 25 days for the same reduction. Litter amendments at 5°C took 86 days for a 99.9% reduction in fecal coliform and 691 days for the same reduction in coliphage. Coliphage were at considerably higher levels in poultry litter amended soils compared to all other treatments. As an indicator organism fecal coliform is less persistent in soil than coliphage. Utilizing results from this laboratory experiment a survival study will be conducted under field conditions.

Fecal coliform transport was observed in Moores Creek watershed, a tributary of the Illinois River in Arkansas, using biweekly grab samples and flow triggered automatic samplers. All storm events form the spring of 1991 until the spring of 1994 were observed using automatic samplers. Grab samples were taken every two weeks over that same period. Grab samples from up stream and down stream locations were compared to reveal trivial spacial differences. Grab samples underestimated maximum possible fecal coliform counts when compared to samples taken from storm runoff using automatic samplers. Transport of fecal coliform occurs temporally during storm events that produce increased stream flow. Fecal coliform levels increased above primary contact level (200 cfu/100mL) during all storm flow events. Increases in fecal coliform coincided with hydrography rises in all cases. The fecal coliform peaks with or slightly after the peak of the hydrography. However, bacteria levels fall much faster than does the tailing hydrography. Relating the survival study to the transport study, there was
no observable reductions in transport between summer and winter storms. However, most manure applications in this watershed occur in the spring and summer. Increased soil amendments could have possibly masked the differences in survival rate due to temperature. Future studies are needed to incorporate survival and transport into the same experiment.