CONFERENCE PROCEEDINGS

NITRATE IN WISCONSIN'S GROUNDWATER: STRATEGIES AND CHALLENGES

May 10, 1994
University Center, UW-Stevens Point

Sponsored by:
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NITRATE IN GROUNDWATER: IS IT A PROBLEM AND SHOULD WE CARE?

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Is nitrate in groundwater a problem, and should we care? The short answer to this question is "IT DEPENDS". "On what does it depend?" is the question that requires more thought. Groundwater, like lakes and streams, is water of the state, and so belongs collectively to all Wisconsin citizens. Until about 25 years ago, Wisconsin citizens, through their elected and appointed decision-makers, permitted untreated municipal and industrial wastes to be dumped into lakes and streams until they became unfit for drinking, swimming, and sustaining ecosystems. Wisconsin citizens woke-up in the 1970s and decided that the situation was unacceptable. This change of heart lead to the clean-up of many point pollution sources and a restoration of many surface waters to a healthier condition.

Wisconsin hasn't had a similar awakening with respect to groundwater. Perhaps it's because we're still operating under a myth that our groundwater is "pure". This despite alarmingly high nitrate standard exceedence rates in many agricultural areas, and that 30% of tested wells (J. Postle, WDATCP, oral comm.) contain atrazine. "Is it a problem, and should we care?" is a question that should only be answered by an informed consensus of Wisconsin citizens, after they've been sufficiently informed about the status of <u>their</u> groundwater resource, and the consequences of taking action compared with doing nothing. The remainder of this discussion brings together information that may be helpful when considering this question.

EXTENT OF NITRATE IN WISCONSIN'S GROUNDWATER

Several years ago, the USEPA completed a $12 million study in an effort to quantify the nitrate and pesticide content of the nation's well-water. The study sampled about 1300 wells across the U.S., and found that 1.2% of community wells and 2.4% of domestic wells exceeded the nitrate MCL (Maximum Contaminant Level, or drinking water standard) of 10 mg/L of NO_3-N. Because nitrate problems aren't spread evenly across the U.S., but rather cluster in areas where nitrate sources are present, the EPA study can't be used to draw conclusions about nitrate in Wisconsin groundwater. However, the national statistics provide a good basis for comparing Wisconsin data.

No one has conducted a survey for Wisconsin similar to the USEPA study, but data are available to paint the nitrate picture for Wisconsin on a county-by-county basis. These data sources are:

1. The Central Wisconsin Groundwater Center data base, containing over 22,000 records of wells sampled through Extension agents and Extension programming. Counties are represented by a range of 2 to several thousands of analyses in this database. Only counties with at least 60 samples are considered further in this report.

2. County laboratories for four counties (Dunn, Eau Claire, Marathon, and Portage) that provide nitrate testing services for their citizens. Each of these counties is represented by several hundred samples to several thousand samples.

3. WGNHS nitrate sampling programs. The Wisconsin Geological and Natural History Survey has made data available for 11 counties. Unlike Groundwater Center and county data, the WGNHS selects wells for sampling based on geographical distribution, aquifer type, and well depth. The Survey only samples wells for which a construction report is available. Due to these constraints, WGNHS characterizes its estimates of nitrate standard exceedence rates as conservatively low.

Data for 38 counties are displayed in Fig. 1. Seven of the 38 counties have nitrate standard exceedence rates of $\leq$ 3%, 5 counties of 3.01-5% 14 counties of 5.01-10%, and 12 counties of 10.01-27%. Rock County had the highest exceedence rate with 27%, followed by Portage at 19%, and a number of others in the 14-17% range. The phenomenon of nitrate problems tending to cluster is evident in Fig. 1, and an apparent relationship exists between nitrate exceedence rates and whether the county is a more or less agricultural area (Table 1). Agricultural counties generally have exceedence rates of 10-20%, compared to nonagricultural county exceedence rates of 1-3%, which are about the same as the national average. A notable exception to this observation occurs in agricultural counties with heavy soils. Brown County, for instance, has a nitrate exceedence rate of only 4%. Nitrate clustering is even more pronounced in data at the township level (Table 2), where nitrate exceedence rates may approach 40% in agricultural areas.

Table 1. NO_3-N exceedence rates for domestic wells from selected counties (Source: Central Wisconsin Groundwater Center database.)

County	% exceeding 10 mg L^{-1}
<u>More agricultural</u>	
Brown	4.0
Juneau	13.5
Pierce	17.6
Portage	19.0
Sauk	14.0
Waushara	10.9
Wood	6.4
<u>Less agricultural</u>	
Lincoln	2.2
Oneida	4.7
Sawyer	1.5
Taylor	2.6
Vilas	1.2

U.S. domestic wells	2.4

Additional data to help characterize the Wisconsin nitrate situation is available from eleven Priority Watersheds (Fig. 2). Nitrate standard exceedence rates are less than 10% in 3 priority watershed, 10-20% in five, and greater than 20% in 3.

WHAT ARE THE IMPACTS OF NITRATE POLLUTION?

The major concern with nitrate is, of course, human health. Since health issues are examined in detail later in this proceedings, the following discussion addresses how affected well users are burdened with cost and effort.

Impacts on well users

Domestic well users and owners are frequently impacted by nitrate in groundwater. At least 3500 domestic water wells in Wisconsin are known to exceed the nitrate standard (R. Clark, WDNR, oral comm.), but this number is likely a small fraction of the total, given the small percentage of wells in Wisconsin that have been sampled.

Public wells include those serving everything from taverns to municipalities. According to the Department of Natural Resources (D. Swailes,

Table 2. Nitrate exceedence rates in some agricultural townships.

Township/county	No. records	% over MCL
Byron, Fondulac	50	37
Fremont, Clark	57	21
Green Valle, Shawano	48	33
Lindina, Juneau	40	30
Nepeuskan, Winnebago	33	27
Plover, Portage	1134	25

U.S. domestic wells		2.4
Typical Wisconsin agricultural county		8-19

Table 3. Numbers of public water supplies where nitrate MCLs have been exceeded (minimums, see text).

Municipal wells	16
Other community wells (apartments, mobile home parks, etc.)	51
Schools, businesses, etc.	75
Taverns, churches, restaurants, etc.	693

oral comm.), 835 public water supply wells are known to have exceeded the nitrate MCL (Table 3). This figure is likely low; the Department's list of municipal wells, for example, did not contain at least two wells which newspapers report and WDNR personnel are aware exceed the nitrate standard. The 16 identified municipal systems include Arlington, Delton, Friesland, Janesville, Morrisonville, Warfordville, Sauk City, Waunakee, Fontana, Oconomowoc, Chilton, Crivitz, Mattoon, Augusta, Chippewa Falls, and Whiting. Conspicuously absent, but known to exceed the MCL, are Plover and Fitchburg.

Unlike domestic wells, public wells are regulated by the Safe Drinking Water Act, and well owners are required to take certain actions if the water exceeds MCLs. In the simplest cases (bars and restaurants) where drinking water is between 10 and 20 mg/L NO_3-N, owners may only have to post a notice warning their clientele of the situation. However, the situation can be considerably more complex, and may involve treatment to remove contaminants or replacement of wells.

Avoiding or remedying nitrate contamination of water supplies bears costs. Well replacement costs a minimum of $2000 for residences, and typically $100,000-500,000 for municipalities. Often times, drilling deeper to avoid nitrate results in water containing high iron, manganese, and hardness, requiring considerable expense to treat water for these constituents.

The cost of installing approved home treatment units for nitrate ranges from $800-1600, plus costs of any required pre-treatment units, plus operation and maintenance costs. Capital costs for municipal systems were $750,000 ($1600 per household) for the Village of Whiting (population of 1875), and $1.9 million for the Village of Plover. Other, less tangible costs of nitrate in groundwater might include difficulties selling property and perhaps diminished property value.

Squeezing out the consumer

"Squeezing out the consumer" was coined to describe groundwater users being unable to find water sufficiently free of man-made and naturally-occurring problem constituents. The phrase alludes to consumers drilling deeper and deeper to avoid man-made pollutants, but then running into groundwater at depth which contains natural constituents with nuisance or health threatening properties. Drilling deeper to avoid nitrate and other agrichemicals has forced some water consumers to deal with naturally occurring iron, manganese, and radioactivity. Iron and manganese are nuisance chemicals, and may be costly to remove, however, radioactivity is apparently a cause for health concerns. Some wells in central Wisconsin, for instance, draw water from granite formations when usable water supplies could be obtained from the overlying glacial drift aquifer. These wells not infrequently contain in excess of 30,000 picocuries per liter of radon, when a proposed EPA standard is less than 1000 picocuries per liter.

In some parts of Wisconsin, nitrate pollution has precluded municipalities from siting wells. The City of Waupaca, for instance, is unable to find a new well site, because the entire countryside has high nitrate water. The Village of Plover apparently was resigned to installing a nitrate removal system when its municipal wells were installed because no site with groundwater meeting the MCL could be found.

LONG-TERM CONSEQUENCES

What are the consequences of continuing in the current path, with nitrate pollution being virtually unrestricted? In my opinion, the most important three are:

1. Whole groundwater basins will be unable to provide drinking water

meeting standards without some sort of nitrate removal treatment.

This phenomena has already occurred in parts of Wisconsin, and the situation will only become worse as more polluted recharge reaches aquifers and as "clean" water, that which entered aquifers prior to widespread nitrate pollution, discharges into lakes and streams.

2. Groundwater users will be increasingly forced to make trade-offs between agrichemicals or natural problem constituents such as iron, manganese, sulfur, and radioactivity.

   Again, this is already occurring, but will happen more frequently, and over a wider area.

3. Ecosystems will be impacted.

   What is the impact of nitrate virtually flooding groundwater and, through baseflow, surface waters? The answer to this question isn't known. Generally, it is assumed that nitrate doesn't impact surface waters because such systems are phosphorus limited. Then too, at one time it was assumed that chlorofluorocarbons were benign and would not cause ozone depletion. Perhaps more study will reveal that nitrate in natural systems has more than benign properties.

INSTITUTIONAL FACTORS PROMOTING NITRATE POLLUTION OF GROUNDWATER

If Wisconsin collectively decides that "yes" is the answer to "Is nitrate in groundwater a problem, and should we care?", a primary need will be to know what factors promote nitrate pollution of groundwater. I offer the following:

1. Wisconsin has no policy on nitrate pollution from septic systems, though the Public Intervenor's office contends the 1984 groundwater legislation requires state agencies to develop such a policy (later in this proceedings).

2. Farmers are free to apply virtually unlimited amounts of fertilizer and manure to their fields without respect to need, profitability, or environmental degradation.

3. According to UW-Extension, more often than not, farmers apply fertilizer and manure in excess of that required for maximum profitability.

4. Unlike virtually every other pollution source, farmers have no liability for nonpoint agrichemical pollution.

5. Agrichemicals, including nitrate, in water supplies are the consumers problem, not the polluters.

6. The only nitrate control strategy aimed at the agricultural sector is education reaching a minimum of farmers, and voluntary compliance with recommendations.

CONCLUSION

Groundwater nitrate pollution is widespread in Wisconsin and is impacting groundwater users. The consequences of ignoring the problem are potentially severe. Ultimately, a decision to take action or do nothing needs to be made by informed policy-makers reflecting the wishes of an informed citizenry. Citizens must recognize that they have a <u>choice</u>, and if they do not make a choice, it will be made for them by those who pollute or favor the status quo.

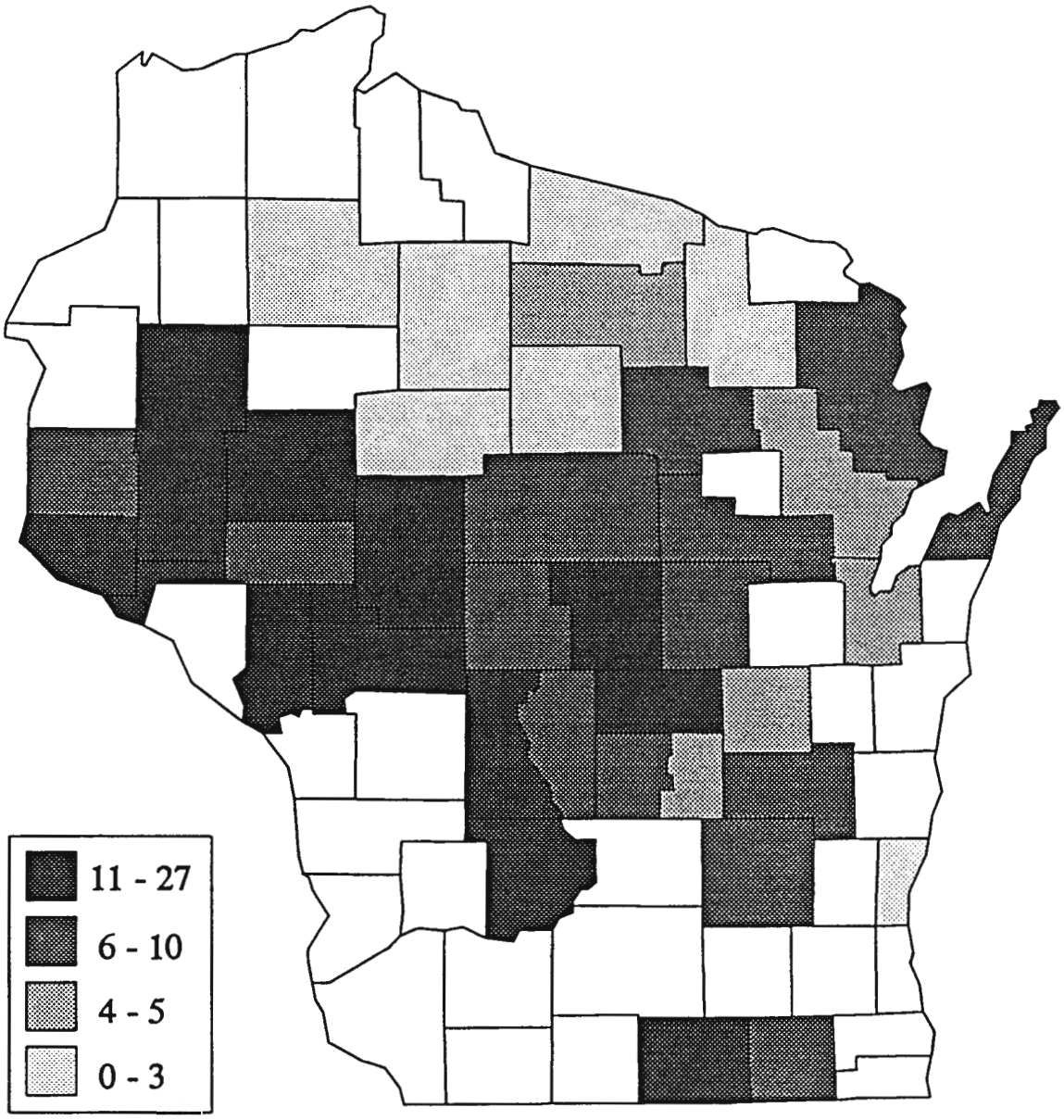

Fig. 1 Percentage of wells exceeding nitrate standard of 10 mg/L nitrate as nitrogen.

NITRATE IN WISCONSIN DOMESTIC WELLS - A SUMMARY BY COUNTY

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This summary describes the nitrate drinking water standard exceedence rates for domestic wells in Wisconsin counties where sufficient data are available. Nitrate and pesticides in water-supply wells have been studied on a national scale by the USEPA, which concluded that 2.4% of domestic wells and 1.2% of community wells in the U.S. exceed the standard of 10 mg/L nitrate-nitrogen (NO_3-N). The USEPA study was too general to make conclusions about exceedence rates at the scale of individual states, however, other data are available for describing the nitrate condition of Wisconsin drinking water wells.

Sources of data

Central Wisconsin Groundwater Center database. The CWGC database (Table 1) contains approximately 22,000 NO_3-N analyses from domestic wells, representing nearly all Wisconsin counties. Twenty-eight counties are represented by greater than 100 samples, 11 by over 500, and 5 by over 1000. The analyses in the data base are from well samples submitted through county Extension offices, either by individuals or by groups participating in Extension "Drinking Water Education Programs". The programs are township- to county-level efforts whereby well sample collection and analysis is followed by an educational session on wells and groundwater.

Some concerns regarding the CWGC data might include (1) the possibility that one well may be represented by more than one sample, thereby skewing the statistics, or (2) the data may not be representative of the wells in a county as a whole. These concerns appear unfounded for counties having substantial numbers of analyses. Questionnaires submitted with water samples indicate that most well-users rarely, if ever, test their well water, so that replicate sampling is a minor concern. The robustness of the data for counties with substantial numbers of analyses is indicated by relatively consistent nitrate standard exceedence rates from year to year. Also, counties where a substantial number of drinking water education programs have been performed have little or no duplication and excellent control on geographical distribution.

I qualitatively assessed the reliability of CWGC data for each county, arranging them in groups 1 (highest reliability data) through 3 (lowest). Counties in groups 1 and 2 are represented by at least 60 samples and have a good geographical sample distribution, while those in category 3 have less than 60 samples and/or a poor geographic sample distribution. The percentage of nitrate exceedences of 10 and 20 mg/L NO_3-N for counties in groups 1 and 2 is presented in Figs. 1 and 2.

County Laboratories. The Central Wisconsin Groundwater Center performed a survey to determine which counties maintained labs that offer nitrate testing services for their residents. Of six counties that maintain labs performing nitrate analyses, four were capable or willing to share data summaries (Table 2). One of these counties is represented by several hundred samples; the others by several thousand.

WGNHS county surveys. The Wisconsin Geological and Natural History Survey has conducted surveys of nitrate in wells for several counties (Table 3). Data is available for 11 counties is presently available. Unlike Groundwater Center and county data, the WGNHS selects wells for sampling based on geographical

distribution, aquifer type, and well depth, rather than making analytical services available to all well-users. Their goal is to describe the spatial distribution of nitrate across a county, rather than characterize the quality of drinking water wells. The WGNHS program will only include wells which are documented by a construction report. Hence, areas where large numbers of-well-users derive water from driven wells (which typically have no construction report) will not be well represented. The WGNHS describes nitrate exceedence rates calculated from their data as "conservatively low" (F. Madison, oral comm.).

Comparison of counties with multiple data sources

Several counties have multiple data sources (Table 4). Fair agreement exists between the two counties represented by county-lab and CWGC data (Portage and Marathon counties). The WGNHS nitrate exceedence rates are substantially smaller than those of other data sources where a significant percentage of nitrate exceedences exist. Several hypotheses for this include the following:

(1) The WGNHS data are biased toward wells in deep formations that are less prone to nitrate contamination, but are also less frequently used for water supplies.

(2) The WGNHS data are biased against driven wells because they rarely have construction reports. Yet, this type of well provides a significant portion of the drinking water in some counties and tends to lie in areas more susceptible to nitrate contamination (F. Madison, WGNHS, oral comm.).

(3) County-lab and CWGC data are from samples provided by well-users. Well-users who live in areas with more nitrate contamination may be more likely to have their water tested than others.

Discussion

Nitrate standard (10 mg/L NO_3-N) exceedence rates are available for 38 counties. The mean county exceedance rate is 8.6% with and a range of 1 to 27%. The following distribution applies (the average was used for counties with more than one data source):

% Samples Exceeding Standard	Number of Counties
1-4.99	11
5-9.99	15
10-14.99	7
15-19.99	4
>20	1

Estimates of the percentage of wells exceeding 20 mg/L NO_3-N are available for 33 counties. The mean rate is 1.5% and range is 0 to 5%. The following distribution applies (the average was used for counties with more than one data source):

% Samples Exceeding 20 mg/L	Number of counties
0	7
0.01-0.99	2
1-1.99	10
2-2.99	8
3-3.99	4
>4	1

Most Wisconsin counties have higher nitrate standard exceedence rates than the national rate, and an apparent disparity exists between what might be termed more agricultural counties and less agricultural counties. Less agricultural counties (e.g., Forest, Sawyer, Vilas) have NO_3-N exceedence rates of about 1-3% (close to the national average), compared to 10, 15, or even >20% in more agricultural counties. The 1-3% exceedence rate in less agricultural counties might represent the typical expected contribution of nonagricultural sources (e.g., septic systems and lawn fertilization). Also, the exceedence rate in some agricultural counties with thick, clayey soils, such as Brown and Winnebago, is low.

The available data seem sufficient to characterize the nitrate status of much of Wisconsin, with the exception of some of the southern third of the state. This gap should be filled, as agriculture is a major land-use in these areas.

Table 1. Percentage of samples exceeding 10 mg/L and 20 mg/L nitrate nitrogen, by county. Data source: Central Wisconsin Groundwater Center database.

COUNTY	NO. SAMPLES	% ≥ 10 mg/L	% ≥ 20 mg/L
	Counties with	≥60 samples	
	Reliability	Group 1	
ADAMS	567	10	3
BROWN	276	4	1
CLARK	1613	11	1
DOOR	141	9	1
FOND DU LAC	771	9	1
GREEN LAKE	78	5	0
JUNEAU	312	14	2
LANGLADE	533	8	1
LINCOLN	505	2	1
MARINETTE	179	6	1
OCONTO	297	4	0
ONEIDA	409	5	1
PORTAGE	5185	19	4
SAUK	316	14	3
SHAWANO	449	8	2
TAYLOR	589	3	1

TREMPEALEAU	192	18	3
VILAS	487	1	0
WALWORTH	321	7	2
WAUPACA	1930	7	2
WAUSHARA	769	11	5
WINNEBAGO	1771	4	1
WOOD	1285	7	2
	Counties with Reliability	≥60 samples Group 2	
DODGE	78	9	0
FOREST	112	0	0
JACKSON	64	16	2
MARATHON	88	8	0
MARQUETTE	231	6	2
OZAUKEE	89	2	0
PIERCE	236	18	4
PRICE	109	3	0
SAWYER	288	1	0
	Counties with Reliability	<60 samples Group 3	
ASHLAND	4	0	0
BARRON	4	0	0
BAYFIELD	9	0	0
BUFFALO	9	22	11
BURNETT	4	0	0
CALUMET	7	0	0
CHIPPEWA	8	13	0
COLUMBIA	23	9	0
DANE	51	43	10

Table 1, cont'd.

DOUGLAS	2	0	0
DUNN	4	50	0
EAU CLAIRE	2	0	0
FLORENCE	10	10	10
GRANT	8	13	0
GREEN	3	33	0
IOWA	13	15	8
JEFFERSON	30	17	10
KENOSHA	8	0	0
KEWANEE	10	10	0
LA CROSSE	3	0	0
LAFAYETTE	26	23	0
MANITOWOC	19	32	0
MENOMINEE	1	0	0
MILWAUKEE	9	0	0
MONROE	3	33	0
OUTAGAMIE	26	4	0
POLK	7	0	0
RACINE	29	0	0
RICHLAND	6	0	0
ROCK	15	33	0
RUSK	18	0	0
SHEBOYGAN	43	0	0
ST. CROIX	41	24	7
VERNON	10	0	0
WASHBURN	22	5	0
WASHINGTON	19	11	0
WAUKESHA	15	7	0

Table 2. Percentage of samples exceeding 10 mg/L and 20 mg/L nitrate nitrogen, by county. Data source: County laboratories.

COUNTY	NO. SAMPLES	% ≥ 10 mg/L	% ≥ 20 mg/L
DUNN	440	23	5
EAU CLAIRE	6928	9	
MARATHON	5446	5	0.5
PORTAGE	4120	15	3

Table 3. Percentage of samples exceeding 10 mg/L and 20 mg/L nitrate nitrogen, by county. Data source: WGNHS county surveys.

COUNTY	NO. SAMPLES	% ≥ 10 mg/L	% ≥ 20 mg/L
BARRON	722	10	
CHIPPEWA	728	12	
CLARK	1431	11	
DUNN	600	9	
EAU CLAIRE	338	6	0.5
PEPIN	236	16	
PIERCE	537	9	
PRICE	653	2	
ROCK	406	27	
ST. CROIX	666	7	
TREMPELEAU	535	8	

Table 4. Comparison of nitrate standard exceedence rates in counties with multiple data sources. Only CWGC data in reliability groups 1 and 2 are included. Clark County is excluded because of significant data overlap.

COUNTY	CWGC	COUNTY LABS	WGNHS
DUNN		23	9
EAU CLAIRE		9	6
MARATHON	8	5	
PORTAGE	18	15	
PIERCE	18		9
PRICE	3		2
TREMPELEAU	18		8

Percent of Private Well Samples with Nitrate-N ≥ 10mg/l

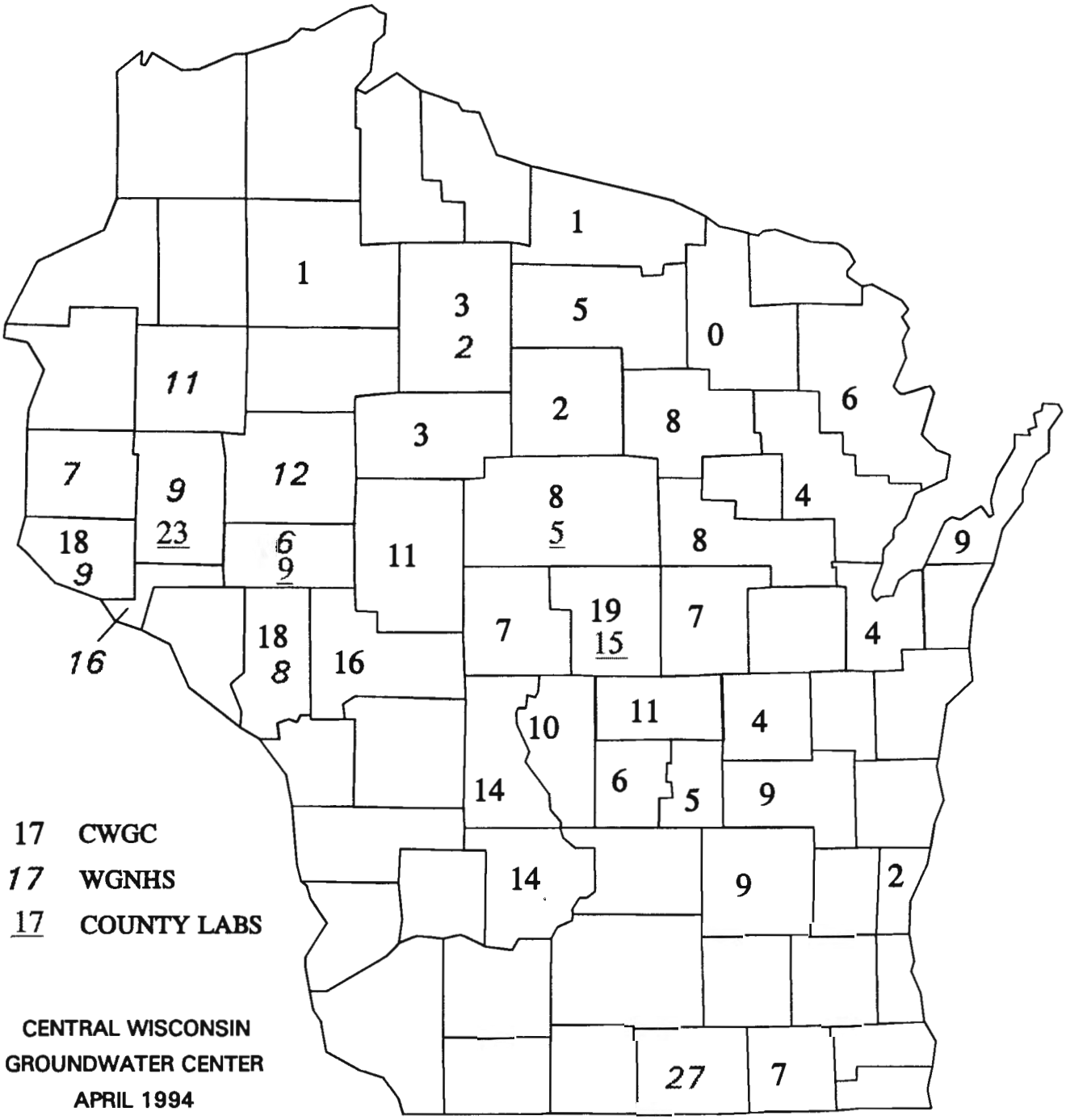


Figure 1.

Percent of Private Well Samples with Nitrate-N ≥ 20mg/l

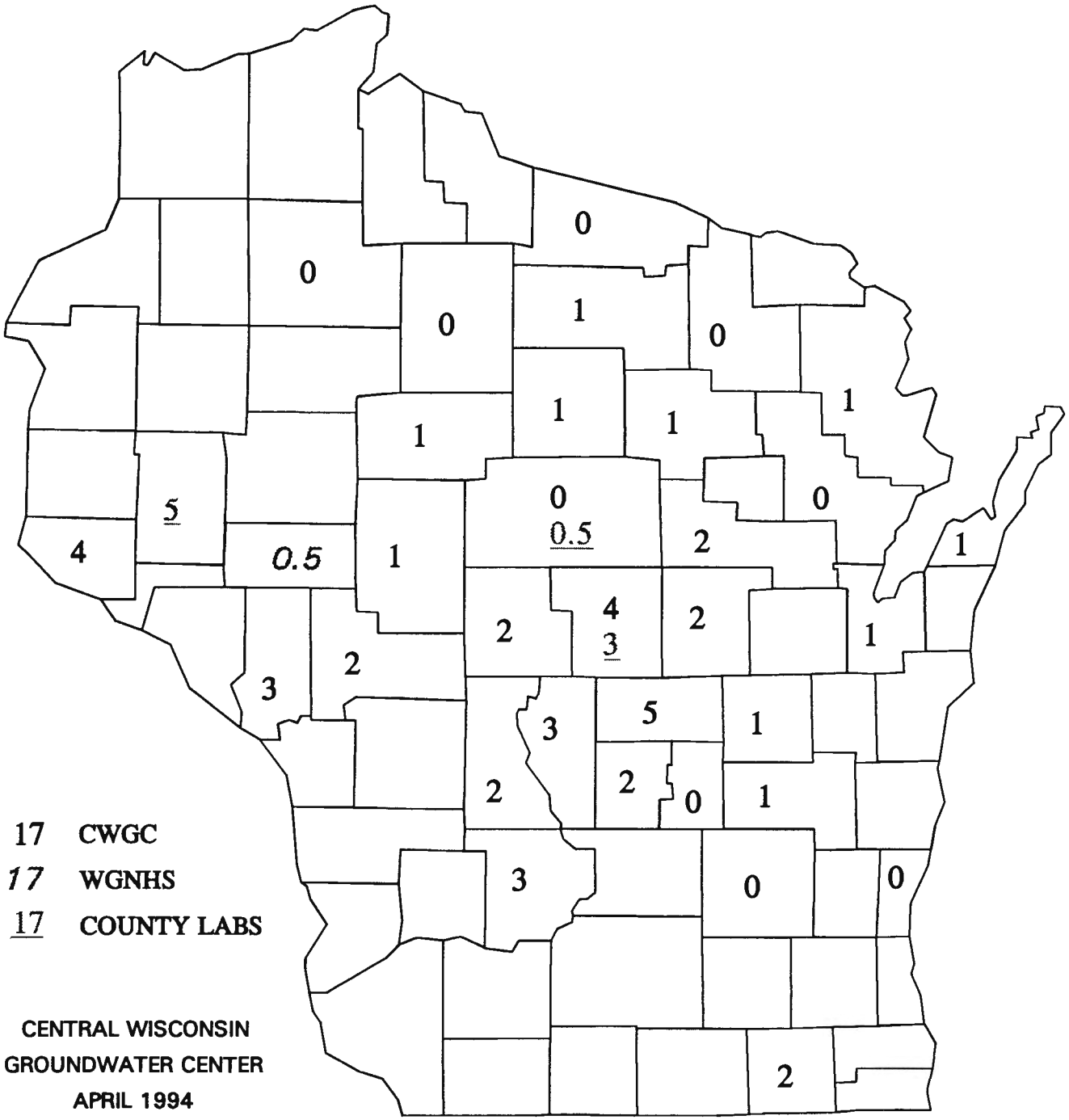


Figure 2.

NITROGEN IN THE NATURAL ENVIRONMENT

R.F. Hensler, Associate Professor of Soils

The Nitrogen Cycle or Where Does Nitrate Come From and Go To.

Nitrogen is matter. Matter cycles; thus, nitrogen cycles. If too much or too little occurs at a specific spot in the cycle at a given time, something is wrong. There is too much nitrate in some groundwaters; we are not managing nitrogen right in all situations.

Let's look at some background. Figure 1 shows the average moisture and temperature distribution in the upper midwest. Plants begin growing after thaw in spring and are stopped by frost in fall. Nitrogen is taken up by plants as they grow and water is evaporated from the soil and transpired by the growing plant. Plant growth rates, microbial activity and the rate of nitrogen cycling increase as temperature increases as long as moisture is adequate.

We can determine nitrogen needs by multiplying dry matter yield by nitrogen concentration. Non-irrigated corn on sandy soils needs 120 pounds N per acre, while corn from Stevens Point and north on medium and fine textured soils needs 150 pounds N per acre; corn in the southern part of the state needs more because of the higher yields with the longer growing season.

The air we breath contains 78% nitrogen gas. But grass plants can't use this. However, lightning, and air pollution from fossil fuel combustion and ammonia from anaerobic decomposition are sources of ammonium and nitrate that fall from the atmosphere. Legumes fix atmospheric nitrogen via symbiotic bacteria and chemical fixation of atmospheric nitrogen is the basis of nitrogen fertilizer manufacturing. But, I won't focus on that today.

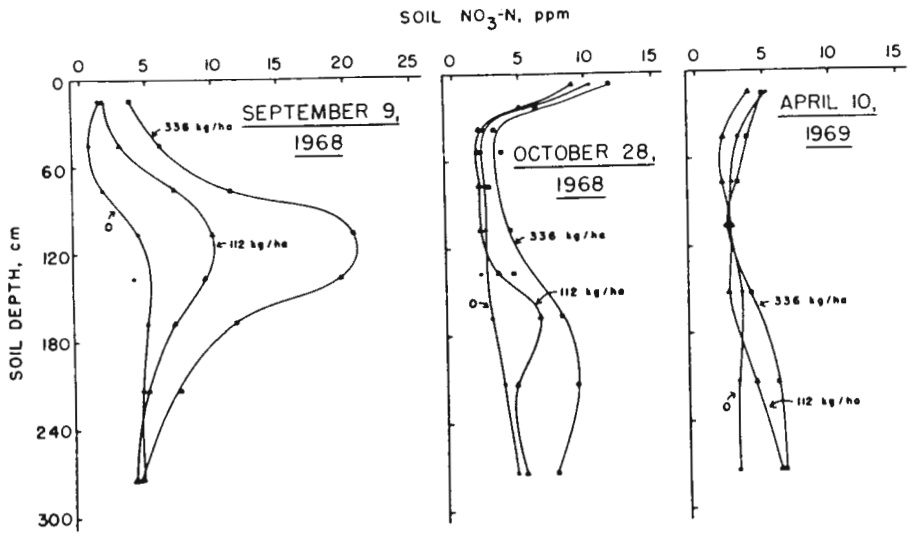
Ammonification is the mineralization of organic nitrogen sources to ammonium by microbial activitiy. This reaction occurs to grass clippings, corn stalk residues, leaves, manures, composts, soil organic matter, etc.; any organic material will be broken down by microbes. The speed of breakdown is temperature, moisture and carbon/nitrogen ratio dependent. The process begins after the soil thaws. Soil organic matter can yield from 50 pounds ammonuim nitrogen per acre per year at 2% soil organic matter to 250 pounds ammonuim nitrogen per acre per year for 10% soil organic matter. This release generally corresponds to plant need during the growing season.

Plants either use ammonium directly or it is converted to nitrate by bacteria. Once the temperature gets above 50 degrees F, ammonium is converted to nitrate in a week or so. Nitrate is used by plants; it can be denitrified from the top soil when the soil is very wet; it can leach through the soil if not recovered by plant uptake.

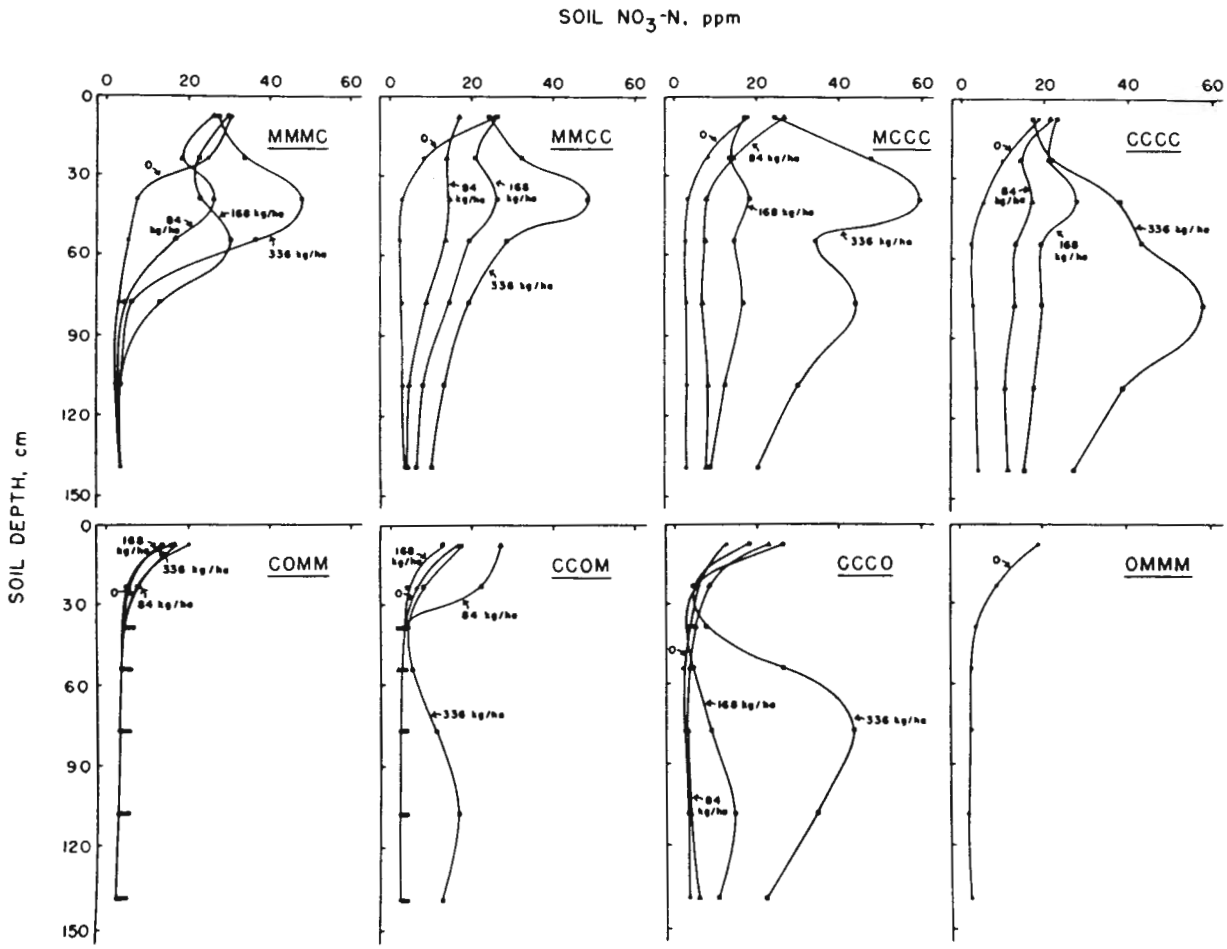
Because the soil dries during the growing season, most movement occurs during the non-growing season except for very sandy soils. On the medium and fine textured soils, plants can recover subsoil nitrate in the next year or two if we reduce nitrogen additions and use deep rooted crops. The more nitrogen that is available, the less efficiently it is recovered by plants. Nitrate left after plant harvest can leach. The bottom line is that natural plant growth is usually nitrogen supply limited. Accordingly, nitrate nitrogen concentrations under forest and unfertilized lawns and other continuously vegetated areas is less than 1 ppm.

Therefore, for groundwater to not experience increasing nitrate concentration, make sure that ammonium and nitrate sources do not exceed plant needs. Make sure that the nitrate is within reach of plant roots.

Bulletin A2519, Soil and Applied Nitrogen by L.G. Bundy is available at UW-Extension offices and gives a synopsis of the nitrogen cycle that has more detail than what we can present today.



On a Plainfield sand, nitrate nitrogen moved about 4 feet with 30 cm of precipitation in 5 weeks after fertilizer application. Additional 14 and 16 cm of precipitation leached nitrate beyond 10 feet. (Olson, et al, 1970, SSSAP p. 450.)



On a Rozeta silt loam, nitrate nitrogen moved about 18 inches per year. Most movement occured between fall and spring. Deep rooted crops that did not receive additional nitrogen fertilizer reduced nitrate nitrogen in the soil profile. (Olson, et al, 1970 SSSAP, p. 451.

Nitrogen Contamination Sources: A Look at Relative Contributions

Byron Shaw - May 1994

As the earlier papers have shown, nitrogen is a fairly complex chemical and can go through a series of chemical/biological transformations. Some of the transformations involve loss to the atmosphere as N_2, NH_3, and N_2O gases which are very difficult to measure and quantify. As a result, it is very difficult to come up with an accurate nitrogen budget.

The numbers used in this paper are based on best available data and existing research information. I have estimated leaching losses of nitrate-N for several land uses to put in perspective the relative contribution of nitrate-N to Wisconsin groundwater. Conservative values were used and may under-estimate the amount of leaching that is actually occurring in many areas.

While statewide numbers are used for many of the inputs to land, there is wide regional and local variations in nitrogen inputs and nitrogen management that result in an equally wide variation in groundwater quality. Data from private, municipal, and monitoring wells, along with groundwater fed streams clearly indicate we have large amounts of nitrate-N reaching groundwater in many areas of Wisconsin. The following discussion covers the major nitrogen inputs to land and groundwater.

Animal Waste

The amount of nitrogen generated as part of animal waste in Wisconsin each year totals 570 million pounds, compared to about 60 million pounds in human waste (Table 1). Being the dairy state has its advantage when it comes to producing organic fertilizer, however, it can be a disadvantage if the manure is not efficiently collected, stored, and spread on crop land with the proper fertilizer credits used. It is estimated that 354 million pounds of manure is applied to cropland, with only 200 million pounds credited for use by row crops annually. The 216 million pounds not applied to cropland is deposited on pasture land, volatilized, or remains in or leaches from barnyards. Leaching and runoff from barnyards can be a significant local source of nitrogen to groundwater. The total nitrogen generated in animal waste is 10 times that in human waste and exceeds by 100 million pounds the total nitrogen removed by crops in Wisconsin (Table 2). Careful distribution and management of this resource could theoretically eliminate the need for commercial fertilizer. It is equally important that both nitrogen and phosphorus be considered when developing programs to best manage animal waste.

Nitrogen Fixation

Nitrogen fixation by legumes is a valuable source of nitrogen for crop production in Wisconsin. A conservative estimate is that 200 pounds/acre of atmospheric nitrogen is fixed annually by 3 million acres of legumes grown in Wisconsin. This totals 600 million pounds/year, most of which remains tied up in vegetation until the land is plowed up for row crops. This is supported by data obtained from monitoring groundwater under alfalfa fields, which show less than 1 mg/l nitrate-N. For the budget on Table 2, one third of these acres were assumed plowed up each year with a 120 pound/acre credit given to the

first year crop. An additional 40 pounds/acre is credited the second year. The total contribution to row crops is estimated to be 160 million pounds/year. These credits vary depending on the quality of the stand and soil type.

Table 1. Nitrogen Inputs to Wisconsin Soils (million pounds/year)

Fertilizer-Agricultural	471
Non-Agricultural	14
Manure	570
Legumes	600
Sludge-Municipal	5.8
Septage-Holding Tanks	3.4
Atmosphere Sources	348
Septic Systems	18
Irrigation	10
Total	**2,040**

Fertilizer Nitrogen

Nitrogen fertilizer sales in Wisconsin were 516 million pounds in 1981 and 486 million pounds in 1991. No direct records are kept for lawn fertilizers, however, about 35,000 tons of non-agricultural fertilizers were sold in 1992. If the non-agricultural fertilizers average 20% nitrogen, then 14 million pounds of nitrogen/year are used on non-agricultural areas. This is 3.1% of the total nitrogen fertilizer use. Assuming most of this is applied to lawns and golf courses, I estimate that 5%, or 0.75 million pounds per year of nitrogen, are lost to groundwater, based on current literature and studies in the central sands. By contrast, losses from agricultural fields to groundwater (from Table 3) total over 165 million pounds. Losses from individual fields and lawns vary significantly depending on management practices and soil types. The averages used in these calculations are derived from data reported in the literature and from N-loss model runs conducted for Wisconsin crops.

Municipal - Human Waste

People generate about 12 pounds of nitrogen/person/year. Of this, approximately 2 pounds/person remains in sludge or septic tanks and 10 pounds/person is discharged to surface water or drainfields. At this rate, the entire Wisconsin population would produce 60 million pounds of nitrogen annually. 2.1 million people are estimated to be served by on site waste treatment, either septic systems (600,000) or holding tanks (50,000). The nitrogen from septic systems frequently leaches to groundwater, except in heavy soil areas where denitrification may occur after nitrogen leaves the drainfield. I have estimated that statewide 80% of the 10 pounds of nitrogen/person/year that enters the drainfield eventually reaches groundwater as nitrate-N. This amounts to 15.4 million pounds/year from drainfield leaching. Most systems we have studied in sandy soil areas lose close to 100% of the nitrogen to groundwater.

Holding tank waste, solids removed from septic tanks, and municipal sludge is often spread on land. If these sources of nitrogen are disposed of improperly, or if they are not included in nitrogen credits they can result in excess nitrate-N leaching. I have estimated 3.8 million pounds of nitrogen in septage and 1.9 million pounds in holding tank waste is

generated annually. No accurate data exists on frequency of pumping and percent that is land applied versus hauled to treatment plants. For purpose of this budget, I have used 60% land applied and estimated that 30% of that would leach to groundwater. I estimated nitrogen present in municipal sludge to be 5.8 million pounds/year, most of which is land spread. Thirty percent leaching of nitrogen from sludge, septage, and holding tank waste gives a total leaching loss of 2.8 million pounds. This is small compared to other sources, but could cause local problems if disposal sites are not carefully chosen and the application rate and distribution are not done correctly. Average nitrogen concentrations in various forms of human waste are; holding tank (137 mg/l), septage (359 mg/l), portable toilets (1,140 mg/l), municipal sludge (40,000 mg/kg dry weight). Septic tank effluent usually has 50 to 100 mg/l total nitrogen.

Table 2. Annual Nitrogen Inputs, Outputs and Residuals for the 9.5 Million Acres of Agricultural Land and 6 Million Acres of Row Crop Land. (Million Pounds/Year and Pounds/Acre)

			Crop Land			
	All Agricultural Land		**Total Inputs**		**Amount Credited[3]**	
	(Mil#/Yr)	**(#/A)**	**(Mil#/Yr)**	**(#/A)**	**(Mil#/Yr)**	**(#/A)**
INPUTS						
Fertilizer	471	50	450	75	450	75
Manure	570	60	354	59	200	33
Legumes	600	63	200	33	160	27
Precipitation	95	10	60	10	0	
Municipal Sludges	5.8	0.6	5.8	1.0	5.8	1.0
Septage and Holding Tank Waste	3.4	0.4	2.0	0.3	2	0.3
Irrigation Water[1]	10	1.1	10	1.7	0	0
Crop Residue			84	14	0	
Soil Mineralization[2]	712	75	450	75	240	40
Total Inputs	**2,467**	**260**	**1,616**	**269**	**1,058**	**176**
OUTPUTS						
Crop Removals	894	94	441	73	441	73
Crop Residues			84	14	0	
Residual Total	**1,573**	**166**	**1,091**	**182**	**617**	**103**

[1] Based on 250,000 acres and 14 mg/l nitrate-N.
[2] Based on 3% organic matter and 2.5% mineralization rate.
[3] Credits based on University recommendations or DNR requirements.

Other Inputs

Precipitation contributes 10 to 15 pounds of nitrogen/acre/year. Nitrogen in dry

deposition is more difficult to measure, but may equal nitrogen in precipitation. These are the only source of nitrogen for forest growth and a major source of nitrogen to many lakes. It is however not a large input to agriculture and is not even included in nitrogen credits.

Irrigation water used in Wisconsin will often have a high nitrate concentration, as nitrate leaching from irrigated cropland is very common. I estimated 10 million pounds of nitrogen are recycled back to the land surface from groundwater annually using 14 mg/l nitrate-N in irrigation water, 12 inches/year of water use on 250,000 acres.

Soil mineralization can be a very important source of nitrogen to crops in Wisconsin. It is estimated that annually, 2.5% of the nitrogen in soil organic matter is converted to plant available forms. I used an average soil organic matter value of 3%, which is on the low end of many agricultural soils. As shown in Table 2, part of the nitrogen is credited by reducing fertilizer requirements. As with many other credits, it is probably not used by a majority of farmers and could reduce fertilizer use.

Crop residue includes materials such as corn stalks, potato and bean stalks or vines that are left on the field after harvest. Nitrogen in crop residue can be viewed as both inputs and outputs from agricultural land, therefore, they were not included in the budgets for amount credited in the last column in Table 2.

Residues from early harvested crops can result in nitrate leaching by decomposition and conversion to nitrate in late summer and fall. Use of cover crops to take up the released nitrogen can help prevent this problem. Residue also contributes to soil organic matter with release of nitrogen in subsequent years.

Outputs

The major output used in the budget shown in Table 2 is from crop removal. Crop residues were not included in the input or output columns except for the total cropland budget. They cancel one another out, and have little to no effect on the residuals or unaccounted for nitrogen. As discussed above, they can affect leaching.

Budget Summary

Table 2 summarizes the total inputs of nitrogen to agricultural lands in Wisconsin and shows UW recommended credits and those required for sludge application. It is obvious that commercial fertilizer, legume credits, and manure make up the majority of nitrogen sources for crop production. At least half of the residual listed in Table 2 is likely reaching groundwater. The leaching losses are estimated in Table 3.

When all inputs are considered, the total is almost three times that removed by crops. The ratios of total input to output for row crops is 3.7 to 1 and even when amounts normally given credit for crop growth are compared to a crop removal a ratio of 2.3 occurs. These data clearly show that agriculture could reduce nitrogen additions to cropland. Much of the problem occurs when farmers fail to take credit for manure and alfalfa, and when manure is produced in large volumes without nearby cropland available for safe use.

Leaching Losses

Table 3 presents relative leaching loses for major crops, septic systems, lawns, and forests. Loss estimates used in this table are conservative and are generally on the low end of what has been observed or predicted by computer model. This table shows the largest loss to groundwater is occurring from row crops. The local mix of land use with their relative inputs is important when determining concentrations of nitrate-N found in groundwater. Septic systems or potatoes, while a small percent of total inputs can be significant sources of high nitrate-N in local areas.

Based on these estimated inputs and credits, it is obvious that agricultural lands receive far more nitrogen than needed for crop growth. Improved crediting and distribution of waste materials could substantially reduce the need for purchased nitrogen fertilizer. This would save farmers large amounts of money and the loss of excess nitrogen to groundwater. Reducing fertilizer purchases by 100 million pounds would save over 20 million dollars per year, and could reduce leaching by 17 pounds/acre or about 8 mg/l.

Table 3. Estimated nitrogen leaching losses to Wisconsin groundwater from major land uses and commonly observed nitrate-N concentrations in groundwater.

Land Use	#Acres or #Systems	Lbs/A/ System Nitrate-N	Total Loss (Mil#)	% of Total	NO3-N Conc in Groundwater (mg/l)	
					Ave	Range
Alfalfa	3 million	2	6	3.2	1	0 - 4
Corn	4 million	30	120	64.2	15	5 - 40
Potatoes	0.069 million	70	4.8	2.6	30	15 - 70
Septic Systems	0.6 million	30	18	9.6	35	10 - 100
Other Ag	2.5 million	15	37.5	20.0	8	2 - 30
Lawns		1.5	0.75*	0.4	5	0 - 20
Forests and Grasslands		<1	<1	<1	<1	0 - 2
Barnyards						10 - 200
Total			**187**			

* Based on 5% of non agricultural fertilizer use.

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"WISCONSIN/FEDERAL GROUNDWATER POLICY AND LAW"

By
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May 10, 1994
U.W. Stevens Point

For those of you unfamiliar with the Office of the Public Intervenor, let me begin by explaining who I am and what our office does.

The Wisconsin Public Intervenor is an office in the Department of Justice expressly charged by state statute with the duty of intervening, wherever necessary, to protect "public rights" in the natural resources of the state. (Sec. 165.07, Stats.) The Public Intervenors are assistant attorneys general and there are two full time and one part time intervenors. We represent "public rights" in the environment. We do not represent the Attorney General nor the Department of Justice. The office is advised by a Citizens Advisory Committee whose members are appointed by the Attorney General and who determine the priorities and cases for the lawyer-intervenors to work on.

The CAC has me working on private sewage issues because of the widespread and irrevocable environmental consequences of septage policy -- both ground water contamination and urban sprawl. Because of our office's interest in septage issues, you can well appreciate that the issue of nitrate contamination -- the subject of this conference -- is one that arose early on.

Nitrates, as well as other septage contaminants, pose threats to human health. Our guest speaker from the California Environmental Protection Agency, who follows me on the podium here, is best suited to describing the human health impacts at stake from nitrates. And while much may be said for the massive nitrate loadings onto our soils from agricultural and homeowner fertilizer practices, it cannot be denied nor ignored that septic systems make an important contribution to our nitrate pollution problem. This is particularly the case where there is a collection of private sewage systems, such as in a subdivision. The U.S. Environmental Protection Agency cites septic tank density as the most important factor influencing groundwater contamination from septic systems. See "Subdivision Impacts on Groundwater Quality" by Shaw et al., July 1993, at 5.

Moreover, this source of nitrate contamination is peculiarly important to those two thirds of Wisconsin's residents who rely on groundwater for their drinking water supply -- and whose drinking water wells may be in close proximity to nitrate contamination from septic systems. So it is not just the mass loading of nitrates statewide onto our soils from agricultural and homeowner fertilizer

use that needs our attention. There is also the site-specific threat of nitrates to our groundwater from localized pollution from septic systems, as Professor Shaw's presentation earlier warned us about.

It has long been acknowledged by the technical experts that: "The buildup of nitrate in groundwater is potentially one of the most significant long-term consequences of on-site sewage disposal practices." See "Predicting Ground-Water Nitrate-Nitrogen Impacts," Hantzsche and Finnemore, 1993 Abstract, at 1. And a 1993 study produced by the U.W. Stevens Point (Shaw, Arnsten, and VanRyswyk) studying subdivisions in the Central Sands area of the state, concluded septic systems "contributed approximately 80 percent" of the nitrates to groundwater, with lawns contributing the remainder. See "Subdivision Impacts on Groundwater Quality," by Shaw, et al., Final Report July 1993, at ii. And studies by the University of Wisconsin-Madison's James. C. Converse show that regardless of the technology -- be it conventional septic systems, mound systems, or aerobic systems -- groundwater standards are exceeded by septic effluent. See "Principles of Pretreatment Systems for On-site Waste Disposal" by James C. Converse, January 1992 (U.W. Wisconsin-Madison).

It is no surprise, then, nor any wonder, that the state regulatory agency in charge of licensing septage systems -- the Wisconsin Department of Industry, Labor and Human Relations -- estimates that "most" of the half a million or so private sewage systems in the state do not meet the groundwater standards, primarily for nitrates. See "DILHR: Groundwater Protection Issues," by DILHR's Michael Corry, et al., November 15-16, 1991, at 1. Yet, and I will talk more about this later, DILHR continues to approve the construction of new systems--only making the current unlawful situation worse--And all of this contamination and legal violation occurring in a state where, as I mentioned before, two-thirds of the states citizens rely on groundwater for their daily drinking water supply. See "Groundwater Quality Monitoring Plan, Fiscal Year 1993," DNR (Dec. 1992), at 2.

But, first, what are the laws that apply here?

WISCONSIN'S GROUNDWATER LAW

In 1984 the Wisconsin Legislature created a comprehensive state regulatory scheme to protect Wisconsin's groundwater. The new law, called the Groundwater Law (1983 Wis. Act 410), created and amended several provisions of the state statutes. The centerpiece of the Act was the newly-created ch. 160 of the statutes entitled "Groundwater Protection Standards."

The general onus of the Groundwater Law is placed on all state agencies that make decisions and take actions that affect groundwater.

The foundation of the Groundwater Law is development of, and reliance upon, "numerical standards," see sec. 160.001 (intro.),

Stats., specifically, "preventive action limits" (PAL), and "enforcement standards" (ES) designed to compel the triggering of regulatory action by agencies in response to groundwater contamination before groundwater standards are exceeded (sec. 160.001(2)-(7), Stats.).

The thrust of the law is for all state agencies to ensure their regulatory actions prevent exceedences of the PALs and ESs; to take action when attainment of PALs are threatened so that groundwater does not reach the level of contamination in an ES; and, if either standard is exceeded, the state agencies are required to take action. Secs. 160.23 and 160.25, Stats.

The clear and nondiscretionary duties of agencies -- including DILHR, for example, to protect groundwater include the following:

a. Agencies' duty to review and revise its existing rules to ensure compliance with groundwater standards and to minimize contamination.

(1) Section 160.19(1), Stats., requires that whenever an ES or PAL is established for a substance, "each regulatory agency shall review its rules and commence promulgation of any rules or amendments of its rules necessary to ensure that the activities, practices and facilities regulated by the regulatory agency will comply" with the groundwater law.

(2) Under sec. 160.19(2)(a),

> Each regulatory agency shall promulgate rules which define design and management practice criteria for facilities, activities and practices affecting groundwater which are designed, to the extent technically and economically feasible, to minimize the level of substances in groundwater and to maintain compliance by these facilities, activities and practices with preventive action limits, unless compliance with the preventive action limits is not technically and economically feasible.

(3) Under sec. 160.19(3), Stats., an agency "may not promulgate rules defining design and management practice criteria which permit an enforcement standard to be attained or exceeded . . . ."

(4) Under sec. 160.19(4), Stats., even where an agency has already reviewed or amended an existing rule or promulgated a new rule and yet PALs or ESs are attained, the agency shall review its rules once again and "revise the rules" to "ensure the enforcement standard is not attained or exceeded at a point of standards application . . . ." Sec. 160.19(4)(b), Stats.

(5) Under sec. 160.19(11), Stats., every regulatory agency shall enforce rules with respect to specific sites.

b. Agencies' duty to adopt rules specifying responses to contamination.

(1) Section 160.21(1), Stats., requires for each substance having an ES or PAL, every agency must "promulgate rules which set forth the range of responses which the regulatory agency may take or which it may require the person controlling a facility, activity or practice which is a source of the substance to take if" a PAL or ES is attained or exceeded.

(2) Sections 160.21(2) and (4), Stats., requires each state agency to determine "by rule" the point of standards application for any source of a substance for which an enforcement standard or PAL is established and provide a range of responses if an ES or PAL is exceeded.

c. Agencies' duty to implement site-specific responses to violations of the PALs or ESs.

(1) On a site-specific basis, where a PAL is attained or exceeded, the regulatory agency must implement a response to "ensure" that the ES is not exceeded. Sec. 160.23(1)(c), Stats.

(2) Where standards are violated, the regulatory agency must take responses "in accordance with rules promulgated under s. 160.21."

(3) Where an ES is attained or exceeded, the regulatory agency must take statutorily-prescribed action that includes consideration of background concentrations of "naturally occurring" substances.

And less there be any doubt that the groundwater law applies to private sewage systems, a quick note is in order. Another specific part of the Groundwater Law amended DILHR's general enabling statutes (found in ch. 145) by emphasizing: "The state plumbing code shall comply with ch. 160(the Groundwater Law)." Sec. 145.13, Stats. In fact, a $25 "groundwater fee" is collected for each private sewage system permit to go into an environmental fund for groundwater management. Sec. 145.19(6), Stats.

Bringing this discussion back to nitrates specifically, under the Groundwater Law nitrates receive some special recognition -- not with respect to standard setting nor the duty on agencies to promulgate rules to prevent contamination, but only with respect to enforcement.

Under sec. 160.25(3), if the nitrates standard is attained or exceeded, the regulatory agency is not required to impose a

"prohibition or close a facility" if the following two factors exist: The enforcement standard was attained or exceeded in part or whole because of high background concentrations of the substance and the additional concentration does not represent a "public welfare" concern.

The two standards set under the Wisconsin Groundwater Law are 2 ppm for the "preventative action limit" and 10 ppm for the "enforcement standard." And while these standards have been criticized by some, they have been reviewed by many and have been upheld by the key agencies involved: The Wisconsin Department of Natural Resources and the Wisconsin Department of Health and Social Services advise pregnant women to avoid consuming water with more than 10 ppm. One U.S. Environmental Protection Agency health advisor goes even further and advises all women of child-bearing age to avoid elevated levels of nitrates. See June 24, 1992, letter from EPA's Lee Gorsky to DHSS Dr. Henry Anderson.

A recent editorial article on nitrates by Ron Hennings of the Wisconsin Geological Survey advised that "This is not the time to change the health standard or to relax the rules that protect groundwater." See Sur View, Vol. 13, no. 1, editorial by Ron Hennings. It should be noted that the 10 ppm standard is identical to that of the federal drinking water law. Which brings us to the federal laws applicable.

FEDERAL NITRATE POLICY

In short, there is no national groundwater law equivalent to the Wisconsin Groundwater Law. There is a federal Safe Drinking Water law and it imposes standards, called "maximum contaminant levels" (MCL) for certain substances, including nitrates. See Safe Drinking Water Act, PL> 93-523, as amended. These MCL's, along with the creation of national primary and secondary drinking water regulations, are to be set by the U.S. Environmental Protection Agency; and states can be delegated primary enforcement authority in certain circumstances. A MCL "shall be set at the level at which a known or anticipated adverse effect on the health of persons occur and which allows an adequate margin of safety." See 42 USCS Sec. 300g-1(b0(4). The MCL for nitrates is 10 ppm, the same, you will note, as the Wisconsin enforcement standard for nitrates.

However, the federal law applies only to public drinking water supplies and not to groundwater generally, in contrast to the Wisconsin Groundwater Law. By legal definition, public water supply is one with at least 15 connections or that serves 25 or more persons. Only weeks ago, the community of Fitchburg in Dane County, for example, was required to shut down its public water supply because of nitrate contamination.

DILHR HAS FAILED TO COMPLY WITH THE GROUNDWATER LAW

Others much more expert than our office in the agricultural nitrate contamination situation will speak today about that

important issue. Our office has devoted major efforts over the past decade in trying to prevent groundwater contamination from pollutants from private sewage systems, and this includes nitrates. As I mentioned before, the state regulatory agency in charge of septage licensing admits that "most" of the approximately half of a million private sewage systems in the state do not meet the groundwater standards, and principally for nitrates at that. To date, and in spite of the over decade-old Groundwater Law, DILHR has failed to revise even one single private sewage code to "ensure", as the law requires, that its actions comply with the law.

Numerous efforts have been made by outsiders in the many years following passage of the law to urge, encourage and assist DILHR in complying with the Groundwater Law. For example, the Public Intervenor office has devoted numerous resources since 1984, including serving on three successive advisory committees established by DILHR, including the filing of a formal rule petition to get DILHR to improve its subdivision code, and including numerous memoranda and letters to DILHR urging strategies about how to comply with the law. Many others who serve on these committees do so voluntarily -- and are weary of devoting the time and trips to Madison when, each time, their work product is ignored by the department. Because all of this has been to no avail.

Not only has DILHR refused to promulgate proposed rules that would strengthen its code, DILHR has attempted to further <u>weaken</u> its already inadequate restrictions on private sewage systems.

Very briefly, this is the history. In 1984 the department created its <u>first</u> committee to draft a code to regulate the very <u>large</u> private sewage systems. Raw sewage leaking from a 59-unit motel newly constructed on a large mound system(an above-ground type of septic system) in Door County was one impetus for this effort. While there was a lack of adequate groundwater monitoring at many of the then-existing large systems around the state, for the two systems that had been monitored and studied extensively, the results were alarming. In spite of the fact both systems were loaded "well below" their designed rates, a DNR investigation and report concluded that the large systems did not meet the groundwater quality standards required by state law. <u>See</u> "Investigation of Large Scale Subsurface Soil Absorption Systems," WDNR, February 1989, at p. ii, 13, 16.

It made sense, then, for DILHR to tackle <u>large</u> systems first because DILHR could take advantage of the modeling techniques that exist to predict impacts from large systems; it is cost-effective to regulate large systems; and the health and environmental consequences of failure are presumably greater for large systems than from an individual home.

All of the promises DILHR has made to promulgate this code were broken. While DILHR set a schedule to complete the process by November 1986, it was not met and so the Legislative Audit Bureau

recommended DILHR make this a "high priority". The DILHR-appointed citizen committee not only completed a final draft code for promulgation (ch. ILHR 88), but the entire rule hearing process was then completed by 1988. But DILHR did not adopt the rule and in spite of all its promises, still has not done so today.

Then, faced with the facts that evidence was accumulating that showed that groundwater contamination occurs by the accumulation of individual private systems in subdivisions, DILHR created a second code committee in December 1988 to overhaul its subdivision code(ILHR 85). A February 1989 rule petition by our office and others sought DILHR improvement of its existing review of subdivisions. We offered a methodology and draft code language requiring developers to consider the total nitrate loadings from the accumulation of systems in subdivisions; under the proposed language, DILHR would not approve subdivisions that modeling predicted could contaminate groundwater.

The DILHR committee met and worked on a code for over a year; that is, until a new ILHR division administrator came on board who failed to call any meetings of the committee after mid-1990. It wasn't even until June 1992 that the agency had the courtesy of writing our office, the rule petitioner, to say that it was abandoning this process, albeit with a "promise" to "incorporate revisions" to the subdivision code when the newly-formed ILHR 83 committee met to revise ILHR 83.

So, DILHR created a third committee in 1991 to review its general code for private sewage systems for any individual private system, including an individual homesite -- ILHR 83. In spite of the original timetable for this product set by DILHR for October 1991, a code was not drafted. Later, DILHR promised a circuit court that it would complete a rule draft by December 31, 1992.[1] That promise, too, was broken, and without any notice to the court. Just last month, DILHR produced yet another wishful timetable for having a revised code in effect by January 1996. Don't hold your breath.

It's not that we think the job is easy. In fact, we thing the job is difficult. That is why we urged DILHR to promulgate the large systems and subdivision codes first. But that also doesn't explain, or excuse, why DILHR has not upgraded chs. 88, 85 or 83 to meet the groundwater law, while it has devoted its resources to weakening its policies as it has done three times in the past two years alone.

In 1991 DILHR deleted sec. 83.056, a provision that expressly limited the use of code variances for "mounds" systems to existing,

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[1]See Affidavit of Michael F. Corry, Administrator of DILHR Division of Safety and Buildings, dated August 1992, at p. 3, in State of Wisconsin Public Intervenor v. Wisconsin DILHR, Case No. 92 CV 2826 (Dane Co. Cir. Ct. 1992).

and not new construction, so as to open up new non-conforming sites to variances. (In its comments to DILHR at the public hearing then, the Public Intervenor office stated our position that sec. 145.24, Stats., limits the use of variances to existing systems and not for new systems.)

Then in 1992 DILHR proposed further code-weakening amendments, including delegating to counties DILHR's regulatory authority over holding tanks and eliminating certain mandatory inspection requirements for mound systems during construction. The Public Intervenor office testified against these rule changes, arguing that instead of weakening its regulatory program over private systems, DILHR ought to be improving its program.

On June 11, 1992, DILHR illegally issued a formal written memorandum to all the state and local agencies charged with enforcing the state's private sewage laws, accompanied by a public press release, announcing the change in its existing policy so as to allow the issuance of variances for new construction of private sewage systems statewide.

So the Public Intervenor office sued DILHR on this policy change, conceded by DILHR to be a "great departure from past practice," See June 11, 1992, DILHR memo at p. 2. The Dane County Circuit Court issued an injunction to prevent DILHR from proceeding to administer its June 11 policy.

The DNR, too, has weighed in on DILHR's actions with a long list of substantial environmental concerns including: groundwater contamination, floodplain development, wetlands, surface water quality, erosion from steep slopes, and land use, stating on one occasion: "The existing code has never been amended to comply with s. 160.19, Stats. Approving mounds systems that are prohibited by the code only expands the noncompliance with the groundwater standards, ..."[2]

Given this history, it is no wonder that the Wisconsin Legislative Audit Bureau, after having investigated DILHR's private sewage programs several times, issued on three different occasions (once in 1987, once in 1989 and again in 1990) numerous criticisms of DILHR's actions -- or more appropriately, failures to act. The following is a typical passage from one of those reports:

> While progress has been made, DILHR lags considerably behind the other state agencies in its efforts to establish the administrative rules necessary to meet its responsibilities under the groundwater law.

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[2]See the July 7, 1992, memorandum from the DNR to the Public Intervenor office.

See "An Evaluation of Groundwater Protection Program," WLAB Report No. 90-28, at 26 (emphasis added); see also: "An Evaluation of The Division of Safety and Building," DILHR, WLAB Report No. 89-29, at 5; and "An Evaluation of The Division of Safety and Buildings Regulatory Decisions," DILHR, WLAB Report No. 87-8, at 6-10.

CONCLUSION

So here we are in 1994, over a decade since the Wisconsin Legislature created the Groundwater Law compelling all agencies of the state to "ensure" their activities do not exceed standards, and DILHR has yet to revise one single private sewage code in order to ensure that even new systems meet the Groundwater Law.

HEALTH IMPLICATIONS OF NITRATE IN DRINKING WATER: PREFACE

During the presentation of Dr. Anna Fan's paper on "Health Implications of Nitrate in Drinking Water", conference participants may have experienced some confusion resulting from terminology, units and state standards described.

Dr. Fan commonly refers to the **maximum contaminant level (MCL) of 45 parts per million (ppm or mg/L) of nitrogen** standard used by the State of California. Here, in the State of Wisconsin, groundwater professionals more commonly refer to **nitrate-nitrogen,** for which the **maximum contaminant level is 10 parts per million (ppm or mg/L).** For practical purposes, 45 ppm of nitrogen and 10 ppm nitrate-nitrogen represent the same concentration of contaminant in water.

HEALTH IMPLICATIONS OF NITRATE IN DRINKING WATER

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INTRODUCTION

In 1987, an evaluation of the nitrate drinking water standard was performed with a primary focus on the effects of nitrate on methemoglobinemia and reproductive /developmental effects (Fan et al. 1987). Since then, additional reports have been available. The present review presents an updated overview and evaluation of the available information on the same health effects of nitrate with an emphasis on data not included in the previous review, which should be used as a compendium to this report.

A literature search did not reveal any reported case of methemoglobinemia occurring in the U.S. resulting from consumption of drinking water containing nitrate at or below the maximum contaminant level (MCL) of 45 ppm (mg/L) nitrate (NO_3) or 10 ppm nitrate-nitrogen (nitrate-N). Higher concentrations, however, were implicated in some cases. Incidences not reported in the earlier review are described below. For epidemiologic studies, all available studies identified are summarized.

METHEMOGLOBINEMIA

A case of infant methemoglobinemia was reported in Petaluma, Sonoma County in 1979 (DWR, 1982). Although the report indicated that samples of the water supply from which the infant's formula had been prepared showed high nitrate levels, the exact concentrations were not specified. Analysis of water samples taken in September and October, 1979 in Northwestern Petaluma Valley, however, revealed that 200 wells (almost 40 percent of the area's groundwater supply) contained nitrate at >45 ppm. The highest value reported for the Valley in 1979 was 367 ppm. A 7-week old infant in South Dakota died with progressive cyanosis due to unrecognized methemoglobinemia; her powdered formula was reconstituted with well water containing 15 times the MCL (Johnson et al. 1988). In Wisconsin, a 6-week old girl was found to have a methemoglobin level of 21.4% after having been hospitalized twice, first for dehydration and vomiting, and later for acute weight loss and limited consumption of formula (MMWR 1993). She appeared "husky" and was afebrile. Her condition was linked to her being given formula mixed with one part water, samples of which showed 9.9 mg/L and 58 mg/L nitrate-N as collected from a reverse-osmosis (R/O) unit and from the well, respectively, during the infant's hospitalization.

REPRODUCTIVE AND DEVELOPMENTAL TOXICITY

Animal Studies

Two earlier reviews (Fan et al. 1987, 1992) showed that nitrate and nitrite produced adverse reproductive effects in experimental animals, but these occurred mostly at extremely high levels of exposure, and most of the studies were conducted with nitrite. Nitrate is converted to nitrite in the

body and the latter is responsible for the formation of methemoglobin in the blood. The MCL is protective against the reproductive effects. Sodium and potassium nitrate and nitrite did not show teratogenic effects (birth defects) when tested in rats, mice, rabbits and hamsters.

Sodium nitrite ($NaNO_2$)administered in drinking water to Long-Evans rats during pregnancy and lactation severely affected erythropoietic development, growth, and mortality in their offspring, but did not cause methemoglobinemia or any detectable level of nitrite in the offspring's blood plasma (Roth et al. 1987). Female rats were maintained on 0, 0.5, 1, 2 or 3 g/liter through pregnancy and lactation. At birth, no differences existed between offspring in the treatment and control groups. Thereafter, pups of dams treated with 1, 2 and 3 g/liter experienced hypochromic, microcytic anemia, and had a significant difference in weight gain and mortality rates at 2 and 3 g/liter. Pups of dams receiving 0.5 g/liter $NaNO_2$ showed reduced mean corpuscular volumes (MCV). The authors considered 0.5 g/liter $NaNO_2$ to be at or near the NOEL. During the lactational period, both fluid consumption and weight gain were significantly lower for treated dams than for controls. Cross-fostering indicated that the influence of gestational treatment manifested in lower birthweights on days 1 through 8 postpartum, while postnatal treatment effects predominated on days 11 through 18 postpartum. Gestational exposure alone had little effect on hematological parameters.

Roth and Smith (1988) administered sodium nitrite at 0.5, 1, 2, or 3 g/L in drinking water to Long-Evans rats during pregnancy and gestation and observed maternally mediated toxicity in the neonates. A dose-related increase in both mortality and growth retardation in pups was seen in rats given 2 and 3 g/L. The animals progressively became severely anemic. At 0.5 g/L, the only effect seen was decreased MCV's at day 16 postpartum which was not seen at day 20. This level appears to be the NOEL. At a MCL of 45 ppm nitrate, and assuming a 5% conversion of nitrate to nitrite, the amount of nitrate ingestion from 2 liters of water would be more than 1000 times less than this NOEL. Severe iron deficiency in pups of dams fed drinking water with 2 and 3 g/liter $NaNO_2$ during lactation was also observed. The authors concluded that dams consuming drinking water containing 2 and 3 g/liter $NaNO_2$ had a reduced capacity to supply iron in their milk, although they were apparently able to transfer sufficient iron to the fetus during gestation, and that nitrite-associated toxicity in the pups was a result of severe iron deficiency.

Human Studies

Several epidemiologic studies have attempted to evaluate the reproductive and developmental effects of nitrate. These include the studies by Knox (1972) in the United Kingdom, Super et al. (1981) in Africa, Scragg et al. (1982) and Dorsch et al. (1984) in South Australia, Arbuckle et al. (1988) in New Brunswick, Canada, and Bove et al. (1992) in New Jersey, U.S.

Knox (1972) studied maternal consumption of cured meat containing nitrate and nitrite and the prevalence of anencephalus in a retrospective study in England and Wales. Information was obtained from statements of monthly still births and infant deaths due to anencephalus, and total births for the period of 1961-67. There was a statistically significant correlation between the per capita consumption of nitrate/nitrite -cured meats and both spatial and temporal variations in the prevalence of anencephalus (a central nervous system, or CNS, defect). However, quantitative information on nitrate or nitrite intake was not known.

Super et al. (1981) studied Namibian infants in Africa and found increased deaths during infancy in births born to mothers from high nitrate areas. There was no correlation between nitrate

area and prematurity and stillbirths. The findings represented a statistical association, but there was no evidence of a causal relationship.

Scragg et al. (1982) and Dorsch et al. (1984) investigated the association of congenital malformation with maternal drinking water supply (containing nitrate) in a case control study conducted in the Mount Gambier region of South Australia. All cases of congenital malformations recorded from 1951-1979 at the general hospitals were identified. There was a total of 43 CNS cases among a total of 218 cases. The cases were matched on an individual basis with "seemingly normal" control babies by hospital, maternal age, and date of birth. Water source was recorded as lake, rainwater, or groundwater. Findings suggested an association of water containing nitrate levels of 5 ppm and higher with neural tube defects. A three-fold risk of malformation of the CNS and musculoskeletal system was associated with consuming water with nitrate at 5-15 ppm, and a four-fold risk if the water contained >15 ppm nitrate.

In this study, several observations would argue against a positive association. The variety of malformation categories observed would argue against a single factor (RR = 4.1, 5-15 and >15 ppm). When analyzing specifically for dose or organ system, the CNS defect had a RR = 3.5. In the multivariate analysis, 15 ppm gave a RR = 4.9 for malformation. However, nitrate was not included in the model. No similar analysis was done for CNS effect. When "groundwater" exposure was mentioned, nitrate was not specified. There were questions as to when and how the exposure data on nitrate were determined. The contributions of both water supply and mother's residence seemed to have an effect, and we cannot exclude that other unidentifiable materials or environmental variables might be correlated with the water supply. There might also be present other unidentifiable spatial variables independent of the water supply, or other factors associated with both residence and water supply. The sample sizes were small (CNS cases/ all congenital malformation = 43/218) and limited data were available on individuals. Overall, it is premature to interpret the suggestive finding exclusively in terms of water nitrate exposure.

In a case-control study in New Brunswick, Canada, Arbuckle et al. (1988) investigated the relationship between maternal exposure to nitrates in drinking water and the risk of delivering an infant with a CNS malformation. The study included 130 cases, each matched with two controls. Water samples were obtained during the study from the women's residences during pregnancy. Historical data for nitrate levels in the area revealed a slight tendency toward higher nitrates from June through October, but only small fluctuations on a year-to-year basis during the study period. The authors assumed from these data that water samples collected for this study provided a fairly good estimate of the relevant exposure measurements. Fluoride, chloride, and sulfate concentrations in the water samples were also determined. These variables along with mother's age and birthplace, birth order, and water source were analyzed using a mathematical model, and significant interaction terms were retained. According to the logistic model, the relationship between nitrate exposure and CNS defects was modified by the water source. Compared to a baseline nitrate level of 0.1 ppm, exposure to nitrate levels of 26 ppm from private well water sources was associated with a moderate, but not statistically significant, increase in risk (risk odds ratio = 2.30; 95% CI = 0.73-7.29). Nitrate levels in public or spring water, however, showed a slightly negative relationship with CNS birth defects. When data were subdivided into high- and low-prevalence regions, nitrate exposure showed a slightly positive trend but did not appear to be significantly associated with either region. The authors noted that the increased risk seen associated with well water requires further study using a larger case series and a larger population of exposures to nitrate levels exceeding 5 ppm.

A cross-sectional study was conducted in New Jersey to evaluate the relationship between potential residential exposure to contaminated drinking water and adverse reproductive outcomes (Bove et al. 1992). Chemicals evaluated included various organic chemicals, trihalomethanes, and nitrate. The study included 75 towns in four counties; each town had nine to 3,063 births in a year. The study populations received groundwater, surface water, or a mixture of the two. Water sampling data were obtained from monitoring data on public systems (1985-1988) and from water companies which sampled two to four times a year. Vital records and New Jersey Birth Defects Registry provided information on birth and infant/fetal death certificates, and maternal residence for the entire pregnancy and first trimester. The study population included 81,055 singleton live births and 599 singleton fetal deaths; the control included 52,334 births.

No information was provided on drinking water habits or exposure to other potential risk factors such as occupational exposures, smoking, or medication during pregnancy. Information was included on some potential risk factors such as the number of prenatal visits, maternal age, race and education, parity, and previous pregnancy loss. Thirteen outcomes were evaluated. These included birth weight as a continuous variable among "term" birth, SGA, term low birth weight, prematurity, very low birth weight, stillbirths, all surveillance anomalies, all CNS effects, neural tube defects, all cardiac defects, major cardiac defects, ventricular septal defects, and oral clefts. The variable outcome relationships were observed to evaluate the null hypothesis. The study found a decreased prevalence of very low birth weight, and a suggested association with neural tube defects (OR = 1.82), primarily among multiple defect cases. When adjusted for total trihalomethanes, OR = 2.72 (CI = 1.3 - 5.6). The authors concluded that "...the positive associations found in this study do not provide sufficient evidence to make the claim that these contaminants cause adverse reproductive outcomes at the levels commonly found in public drinking water systems, and the scarcity of other toxicological research on the reproductive effects of these water contaminants prevents us from making such claims ...", and that the "association should be taken seriously and investigated further."

CURRENT WATER STANDARD

The current drinking water standard (MCL) of 10 ppm nitrate is enforced by the State. Under certain circumstances, the maximum limit may be set up to 20 ppm for adults. Several considerations would tend to support the argument that the current MCL is too low: 1) there were no documented health problems, especially in adults, at low NO_3-N concentrations; 2) the methemoglobinemia cases in the U.S. were reported at nitrate levels higher than the MCL; 3) a study in the U.S. failed to observe elevated methemoglobin concentrations in Illinois children aged 1-8 years old who drank water containing 22-111 ppm nitrate-N; and 4) only one death occurred and at a nitrate level >10 times the MCL. However, other considerations that would argue that the MCL is too high include the following: 1) among the worldwide epidemiologic data on which the MCL was based, nitrate concentration was not known in all cases; 2) death had occurred in other countries at nitrate levels lower than the MCL; 3) the severity of effect (death) would argue that a conservative approach should be taken.

One major consideration is that uncertainties exist in the data base used for the development of the MCL. There could be yet unidentified differences among the studies and countries which provided data on methemoglobinemia and developmental effects that could contribute to the differences in the findings reported. For example the nutrition status may be different among varying populations; vitamin C can help to reduce the risk of methemoglobinemia from nitrate and

some populations or countries get a higher vitamin C intake level than others. In addition, there is a possibility that methemoglobinemia is more likely associated with nitrate plus bacterial contamination of the water, a condition which favors the conversion of nitrate to nitrite and occurrence of diarrhea, and infant diarrhea can increase the risk. Furthermore, some methemoglobinemia cases that were attributed to nitrate were not substantiated. A particular concern is the finding of methemoglobinemia at a lowest observed effect level with no clear delineation of a no-observed-effect level. Overall no uncertainty factor was applied in deriving the MCL in order to account for the uncertainties in the data base.

SUMMARY

Recent epidemiologic data have suggested an association between developmental effects in offspring and the maternal ingestion of nitrate from drinking water, but a definite conclusion on the cause and effect relationship cannot be drawn. Animal experimental data have shown reproductive toxicity associated with high exposure levels to nitrate or nitrite, which are not likely to be encountered in drinking water. The current MCL is protective of such potential reproductive effects. No teratogenic effects were observed in animals tested. Several cases of methemoglobinemia have been reported in infants in the U.S. using water containing nitrate at levels higher than the MCL, but none at or lower than the MCL. The infants are identified as the sensitive population. No uncertainty factor was applied in deriving the MCL in order to account for the uncertainties that exist in the data base.

Acknowledgment. The author thanks Valerie Steinberg for her assistance in preparing the final manuscript.

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Agricultural Best Management Practices for Nitrogen

Scott J. Sturgul[1]

Improving the efficiency of nitrogen (N) fertilizer applications has both economic and environmental benefits. Applications of N fertilizer in excess of crop need increase the potential for nitrate additions to groundwater. Economically, N removal from the crop root zone is a lost production expense to a grower.

N Application Rate

The most important practice for environmentally and economically sound N management is determination of the appropriate N application rate. University of Wisconsin N fertilizer recommendations for corn are based on the yield potential, texture, and organic matter content of a specific soil series. Yield goal estimates, which were often over-optimistic and led to excessive N applications, are no longer a direct component of N recommendations for corn. The N recommendations are the maximum amount of N needed to obtain economically optimum yields.

To maximize N efficiency, the base N recommendations need to be adjusted for any nitrogen contributions from nonfertilizer resources, such as previous legume crops, manure applications, or land-applied organic wastes. Both economic and environmental benefits can result if the nutrients supplied by these resources are assessed and commercial fertilizer applications are reduced accordingly. Legume crops, such as alfalfa, clover, and soybeans, can supply substantial amounts of N to a following crop. Alfalfa has the potential to provide all the N needed by a corn crop that follows it in a rotation. Similarly, manure can supply crop nutrients, including nitrogen, as effectively as commercial fertilizers and in amounts that can meet the total N requirement of corn. Proper assessment of N credits from legumes, manure and organic wastes is essential in preventing N applications in excess of crop needs.

Recently, new soil tests have been developed for assessing residual soil nitrate, i.e., crop-available nitrate that has remained in the soil profile from one growing season to the next. These two tests, the preplant soil nitrate test and the pre-sidedress soil nitrate test, allow N recommendations for corn to be fine-tuned to site-specific conditions that can influence N availability. If either of these tests indicate substantial soil nitrate, N applications can be reduced or, in some cases, eliminated.

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Sources of N

Many different forms of N fertilizer are available. All can be equally effective if properly applied, although one form may have advantages over another under certain conditions. For example, urea needs to be incorporated to prevent volatilization loss of the N, anhydrous ammonia needs to be injected and should be not be applied to wet soils.

Timing of N Applications

The timing of N fertilizer applications can significantly affect N use efficiency. The period of time between N application and crop uptake is an important factor influencing crop utilization of N and the amount of N lost through leaching or other processes. Ideally, supplying N just prior to the crop's greatest demand (for corn in Wisconsin, this would be mid-June through July) would maximize the efficiency of N applications. For spring seeded crops, sidedress (post-planting) and preplant N applications increase nitrogen use efficiency by 10-15% over fall N applications. For both agronomic and environmental reasons, fall applications of N fertilizers are not recommended on sandy soils, shallow soils over bedrock, or poorly drained soils. The preferred technique for N applications on these soils would be a sidedress or split application(s). If preplant applications on these soils must be made, ammonium forms of N treated with a nitrification inhibitor should be used.

Nitrification Inhibitors

Nitrification inhibitors are used with ammonium forms of N fertilizers on soils where the potential for leaching or denitrification is high. Nitrification inhibitors slow the conversion of ammonium to nitrate. Since leaching and denitrification occur through the nitrate form of N, maintaining fertilizer N in the ammonium form should reduce N losses. Nitrification inhibitors are likely to improve N efficiency when used with preplant N applications on sandy or poorly drained soils and when fall N applications are made on medium-textured, well-drained soils. However, preplant or sidedress N applications are usually more effective than fall-applied N even with the use of a nitrification inhibitor. Use of a nitrification inhibitor with preplant N applications medium-textured, well-drained soils or with sidedress applications on any soil type is not likely to improve N use efficiency.

A more detailed discussion of agricultural management practices for protecting water resources from nitrate (and phosphorus) contributions is contained in the UWEX publication A3557 *Nutrient Management: Practices for Wisconsin corn production and water quality protection*. This publication is included with your conference proceedings.

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DESIGN AND OPTIMIZATION OF TWO RECIRCULATING SAND FILTER SYSTEMS FOR NITROGEN REMOVAL

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Nitrogen entering a conventional septic system is typically in the organic-N and ammonium-N forms. A properly functioning septic tank will remove approximately 10% of the influent organic nitrogen which is stored in the sludge (Laak et. al., 1981). In the septic tank, settling and ammonification occur, resulting in effluent containing primarily ammonium-N (USEPA, 1980; Canter and Knox, 1986). Most of the ammonium is biologically converted to nitrate as the wastewater moves through the unsaturated soil beneath the crust of the soil absorption system. The high solubility of the nitrate anion allows it to move freely with percolating water in the unsaturated zone and within the groundwater.

When used on sandy soils, on-site waste disposal systems have resulted in significant additions of nitrate-N (NO_3^--N) concentrations to groundwater (Ritter et al., 1988; Robertson et al., 1991; Shaw et al., 1993; Shaw and Turyk, 1992; Walker et. al., 1973). With average nitrate-N concentrations of 30 to 70 mg/l down gradient of drainfields, there is increasing interest in devising means whereby household waste disposal can be accomplished without impacting groundwater above the 10 mg/l nitrate-N standard.

The following is a summary of the results obtained in a study of two Recirculating Sand Filters installed on private residences in central Wisconsin. One of the Recirculating Sand Filters was installed on a mound system, the other on a conventional gravity system. Samples from the septic tank, sand filter, dosing chamber, and monitoring wells adjacent to the drainfields are collected at least monthly, and analyzed for the nitrogen series, BOD, pH, and chlorides. Phosphorous and volatile organic chemicals are analyzed seasonally.

RECIRCULATING SAND FILTER DESCRIPTION

Two Recirculating Sand Filters (RSF) were installed on private residences in the summer of 1992 to evaluate their ability to reduce nitrogen loading to groundwater from on-site waste disposal systems. The two sites chosen are located in the north central section of Portage county in the Town of Hull. The two study sites were chosen from single family homes which already had groundwater monitoring wells in place, contaminant plumes well identified, high nitrate-N concentrations occurring in the groundwater contaminant plumes and homeowner cooperation. Site #1 had a conventional septic system with a 3,785 liter (1,000 gallon) concrete septic tank which feeds a 15.8 by 3.7 m conventional drainfield. Site #2 had an existing pressurized mound system consisting of a 3,785 liter (1,000 gallon) septic tank, a one chambered, 3,785 liter (1,000 gallon) dosing chamber and a 20.4 by 8.5 m mound system.

The two denitrification systems installed involve using a Recirculating Sand Filter (RSF) with a built in rock storage area similar to that described by Sack et al. (1988). The systems were designed to remove nitrogen via denitrification in the septic tank following nitrification in the sand filter. Effluent from the RSF was recirculated to the septic tank where an adequate carbon source and anaerobic

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conditions enabled bacteria to denitrify most of the nitrate-N to nitrogen gas. The only notable differences between the two systems is that at Site #1 septic tank effluent flows by gravity to the bottom of the RSF whereas at Site #2 septic tank effluent is applied to the top of the RSF.

A 7,571 liter (2,000 gallon) septic tank was used to house the various components of the sand filter system. The major components included a collection system at the bottom, 39.4 cm (15.5 in) of 2.5 to 3.8 cm (1 to 1.5 in) diameter limestone, 7.6 cm (3 in) of pea gravel, 58.4 cm (23 in) of a 1.8 mm effective size sand with a uniformity coefficient of 1.4, a pump chamber, and a distribution system on the top of the sand filter. Two cm (3/4 in) of treated plywood and 5 cm (2 in) of polystyrene foam were placed around the RSF in order to allow the septic tank enclosing the RSF to be placed deeper in the ground. This was done to allow septic tank effluent to flow by gravity into the RSF and also to help maintain heat throughout the winter. To allow for easy access, the top of the RSF is at land surface and is covered with a three piece insulated plywood cover. A cross section of the RSF can be seen in Fig. 1.

The temperatures of the systems throughout the winter was a primary concern as both nitrification and denitrification are temperature dependent. The optimum temperature range for nitrification has been reported as 18 to 35°C with nitrification ceasing below 5°C (Shammas, 1986). Crites et al. (1981) reported the minimum temperature for denitrification in land treatment systems is 2 - 5°C.

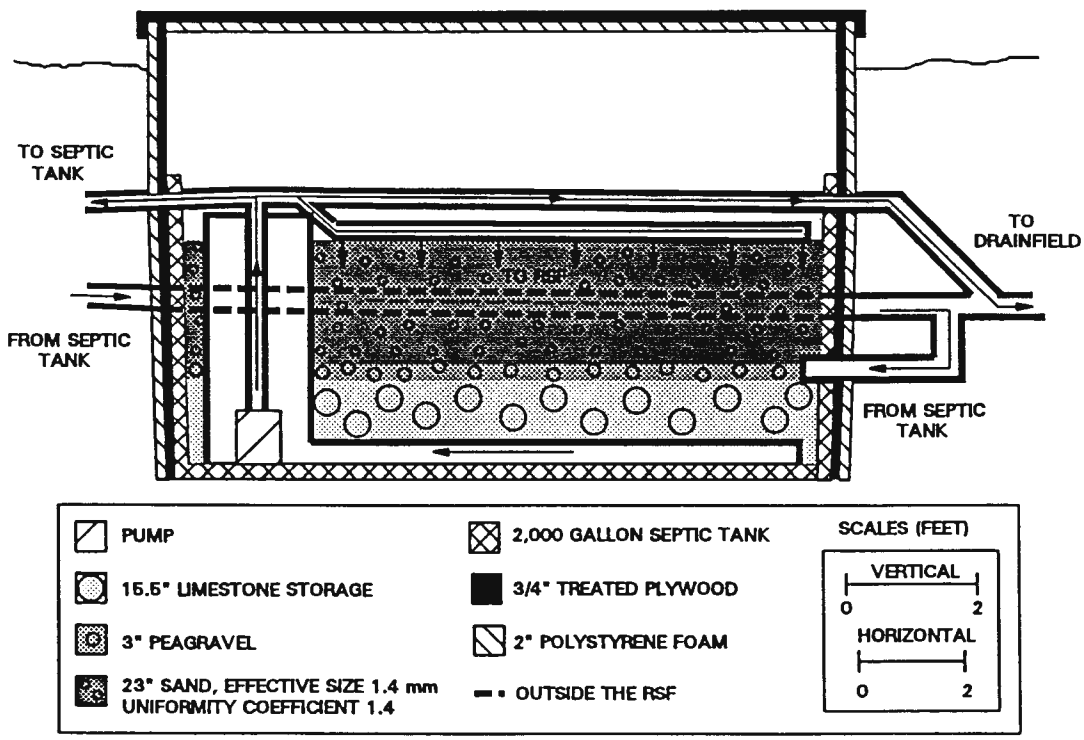

Figure 1. Cross Sectional View of RSF at Site 1.

RESULTS AND DISCUSSION

Wastewater Characteristics

Table 1 shows a summary of the average and percent reductions of several water quality characteristics of the sewage effluent which was applied to the drainfield and mound systems. Total Nitrogen reductions of 70 and 60 percent were obtained for Site 1 and Site 2 respectively and BOD_5 concentrations were reduced by 95% at both sites. Total Phosphorous concentrations were also reduced by at least 60%

however phosphorous removal is expected to decrease as the number of adsorption sites become saturated. The large differences in wastewater characteristics are attributed to large differences in per capita water usage between the two sites. These differences also had an effect on the final concentrations discharged to the drainfield and mound system despite similar treatment efficiencies.

Table 1. Summary of average and percent reductions for several water quality characteristics of sewage effluent applied to the drainfield and mound system.

	Applied to Conventional Drainfield			Applied to Mound System		
	Before RSF installed	After RSF installed		Before RSF installed	After RSF installed	
	AVG. mg/l	AVG. mg/l	RED. %	AVG. mg/l	AVG. mg/l	RED. %
BOD_5	270.6	5.7	97.9	126.5	6.8	94.6
COD	447.5	74.5	83.4	349.0	34.3	90.2
NO_2+NO_3-N	<0.2	15.3	NA	<0.2	3.7	NA
NH_4^+	62.8	7.0	88.8	22.1	6.3	71.5
TKN	76.6	8.1	89.5	28.7	7.8	72.8
TOTAL NITROGEN	76.6	23.0	70.0	28.8	11.5	60.1
TOTAL PHOSPHOROUS	9.0	2.4	73.8	4.0	1.5	62.5
ALKALINITY	381.4	172.2	54.8	138.7	154.5	- 11.4
TOTAL HARDNESS	154.5	179.2	- 16.0	52.0	106.7	- 105.2
CHLORIDES	47.5	33.3	30.0	22.5	18.4	18.2

(-) Indicates a percent increase

Nitrification

At Site 1, oxidation of organic-N and ammonium-N resulted in an average of 15.3 mg/l of nitrate-N in the pump chamber of the RSF. Despite temperatures as low as 3.1°C in the RSF during the winter months at Site 1, nitrate-N concentrations in the pump chamber never decreased below 10 mg/l throughout the study period.

At Site 2, oxidation of organic-N and ammonium-N resulted in an average of 4.7 mg/l of nitrate-N in the pump chamber of the RSF. Temperatures within the RSF at this site were significantly higher throughout the winter months with temperatures decreasing to only 12.3°C. The greater water usage at this site is most likely the primary reason for the higher temperatures. Despite the greater temperatures, only 4.7 mg/l of nitrate-N was found within the RSF at this site as compared to the 15.7 mg/l found at Site 1. The lower concentration of nitrate-N produced within the sand filter has been attributed to the

lower initial nitrogen concentrations found within the wastewater at this site.

Denitrification

At Site 1, 912 liters/day of RSF effluent were pumped to the septic tank for denitrification. Accounting for the forward flow of 352.4 liters/day and the amount directed from the RSF to the septic tank, the theoretical retention time was approximately 3 days. Within the septic tank, an average of 15.3 mg/l of nitrate-N was reduced to 0.4 mg/l for a 97% removal rate. Despite temperatures as low as 4.1°C, denitrification did not appear inhibited.

At Site 2, accounting for the forward flow from the house of 1,694 liters/day and the amount pumped from the RSF to the septic tank of 1,816 liters/day, the theoretical retention time within the septic tank was approximately 1 day. It should be noted that the amount of effluent which has been pumped back to the septic tank for denitrification has been increased in an effort to improve the percent removal at this site. Within the septic tank, an average of 4.7 mg/l of nitrate+nitrite-N was reduced to 0.1 mg/l for a 97% removal rate. Temperature did not appear to be a limiting factor as the recorded minimum temperature of 13.7°C is well over the reported minimum temperature of 5°C required for denitrification. Mass balance calculations indicate additional nitrogen removal is occurring within the sand filter which supplements that occurring by denitrification in the sand filter.

Miscellaneous Results and Observations

No major problems were encountered at either site throughout the study period. At Site 2, a biological mat did appear at the surface of the RSF which caused some ponding of the effluent. However, raking the mat to a depth of about 15 cm (6 in) alleviated the problem for about 6 months after which raking was again necessary. Occasionally the holes within the distribution located on top of the RSF would become plugged in which case it was necessary to run a brush through the distribution lines. It is suggested that a filter be placed around the pump located in the RSF to prevent it from receiving the larger suspended solids.

Presently, the pumping intervals and lengths and the flow rates to the top of the sand filter, to the septic tank, and to the drainfield or mound system are being varied to determine the optimum performance of the RSFs. The systems are also being evaluated for their ability to remove Volatile Organic Compounds (VOCs).

CONCLUSIONS

1. Total Nitrogen removals of at least 60 to 70 percent can be obtained through the use of a system with this design using simple pump and flow regulation equipment.

2. Temperature does not appear to severely limit nitrification and denitrification despite the cold weather encountered during Wisconsin's winter.

3. To eliminate the larger suspended solids, a filter at the septic tank outlet and around the pump located within the pump chamber of the RSF would be desirable thus reducing the need to periodically clean the distribution system located on top of the RSF.

4. Applying septic tank effluent to the top of the RSF greatly increases the development of a biological mat on the sand filter. The mat can be broken up simply by raking the upper portion of the sand filter. It is unknown if allowing the wastewater to enter the RSF at the bottom of the system also results in the development of a biological mat.

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Options for Controlling Nitrogen Inputs to Groundwater from Onsite Waste Disposal

Byron Shaw
October, 1993

Nitrogen in domestic wastewater has been shown to cause Nitrate-Nitrogen concentrations in groundwater to exceed drinking water standards. Concentrations of 30 to 70 mg/l down gradient of drainfields are common. There is increasing interest in devising means whereby household waste disposal can be accomplished without impacting groundwater above the 10 mg/l nitrate standard.

Nitrogen generation per capita averages about 10 pounds per person per year and averages 60-70 mg/l in the effluent from a typical household septic tank. Most of this nitrogen is transported to groundwater from present waste disposal practices. The organic nitrogen and ammonia present in the waste water are convered to nitrate under the drainfield and leached to the underlying groundwater.

The following is a summary of the wastewater disposal options available to prevent nitrate in groundwater from exceeding the 10 mg/l standard. Two mechanisms are available to achieve this goal:

- Dilution of the Nitrogen in Wastewater.
- Removal of nitrogen from the wastewater.

1. Dilution can be achieved by:

   a. use of larger lots with suitable soils and minimal fertilizer use can achieve the needed volume of groundwater recharge having sufficiently low nitrogen concentration to achieve the needed dilution. Dispersal of waste to achieve mixing with this recharge water is the major remaining obstacle to prevent contaminant plumes from exceeding groundwater standards. The use of Nitrogen-Water Mass Balance Models such as BURBS are appropriate to evaluate lot size needed for soils with varying groundwater recharge rates. The use of this option has the negative effect of requiring large lot sizes (2 or more acres) and therefore would result in more urban sprawl.

   b. Dilution of waste with low NO_3 groundwater prior to discharge to soil absorption system is also possible but would hydraulically overload soil absorption systems and be expensive to pump the dilution water if it was even available. The use of water conserving devices actually results in a more concentrated wastewater and, therefore, a higher concentration in contaminant plumes.

2. The removal of nitrogen from wastewater can be accomplished by

   a. treatment prior to discharge to soil absorption systems or
   b. use of the soil as part of the treatment system.

These options are briefly outlined below:

a. Pretreatment for nitrogen removal prior to discharge largely involves either toilet waste separation and treatment or whole house denitrification systems. Other options such as ion exchange, reverse osmosis or distillation are possible but are generally more useful for water treatment than for waste treatment.

   Separating toilet waste (black water) from all other waste (grey water) can result in 70 to 80% of nitrogen being separately collected for treatment or disposal. This waste can be directed to holding tanks for pumping and disposal elsewhere. Problems with holding tank pumping and maintenance are however significant.

   Composting or incinerating toilets are devices used to treat blackwater and may have application to some homes where people are willing to install, use and maintain these systems.

   Treating whole house or black water waste by use of a denitrification system for nitrogen removal involves systems where the nitrogen in the waste is first oxidized to nitrate by use of aeration or sand filters (equation 1) then the nitrate must be reduced to nitrogen gas (equation 2) in a chamber that lacks oxygen and has adequate organic matter for the denitrifying bacteria. Recirculation back to the septic tank or to an additional tank for this final nitrate reduction are both being used. The use of an additional tank following the sand filter or aeration device may require the addition of an organic substance such as alcohol to insure an adequate energy source for bacterial action to convert nitrate to N_2 gas.

   Equation 1: Organic N + NH_4^+ + O_2 - →NO_3^-N + H^+ + H_2O
   Equation 2: $4NO_3^-$ + $4H^+$ + 5 Organic Carbon → $2N_2\uparrow$ + $5CO_2$ + $2H_2O$

b. The use of soils as part of the treatment system for nitrogen removal is possible by either denitrification or uptake by vegetation.

   1. Denitrication. The main mechanism available for nitrogen removal by soils is denitrification. This process needs very specific environmental condition for it to work efficiently. I believe however these conditions do occur in some soil types in Wisconsin. The following are soil criteria needed for denitrification:

      a. Oxidising conditions near effluent lines for conversion of NH_4^+ to NO_3^-. This could be in the sand layer under a mound system or one to three feet of well drained soil under the drainfield.

      b. The aerated, nitrified waste must then encounter a soil layer that is both anaerobic (lacks oxygen) yet contains sufficient organic material for the denitrifying bacteria to use as an energy source.

The combination of the above soil characteristics do not exist in many areas in Wisconsin; however, it may be feasible to design systems using native soils to achieve the needed conditions. More research is needed to identify where denitrification is occurring at present with current septic systems.

Soil conditions where it may be possible to achieve nitrification and denitrifcation include:

1. Mound systems over poorly drained soils where there is sufficient organic matter in the soil or waste to serve as carbon source for denitrification once the effluent passes through the sand layer where nitrogen should be converted to nitrate.

2. Soils where well drained conditions occur over a poorly drained subsoil. There may not be enough organic matter reaching the subsoil for oxygen consumption and to aid bacterial growth and therefore denitrification may not occur.

3. Constructed sites with sand over high organic matter and low permeability soil. A peat layer or layer of sulfur or pyrite under three feet of sand may accomplish the combination of nitrification and denitification.

2. Discharge of septic tank effluent or nitrified sand filter or aeration system effluent to the land surface or in shallow trenches may result in vegetation using the nutrients in waste water during summer months. These may be appropriate for summer only residences. The long Wisconsin winters limit the usefulness of land disposal or even the use of constructed wetlands for nitrogen removal. Surface disposal would also require disinfection to prevent disease problems.

The only other option currently available involves use of holding tanks that need to be pumped and the contents properly disposed of. Current problems with management of holding tanks do not make them an environmentally safe option in my opinion.

There is adequate evidence that various denitrification systems using sand filter will remove 70% plus of nitrogen from domestic wastewater to warrant their adoption in Wisconsin. More evaluation is needed of which areas need nitrogen removal systems to protect groundwater and which areas are adquately protected by natural denitrification.

LAND USE PLANNING

Charles P. Kell
Portage County Planning and Zoning

Portage County's programs to protect groundwater by land use control techniques began with the preparation of a Comprehensive Groundwater Management Plan for the County in 1988. The plan represented a thorough assessment of potential pollution sources throughout the County and a review of management techniques and options available to protect groundwater. These options include preventative/regulatory programs, preventative/remedial programs, and educational programs. The preventative/regulatory programs area is the primary area of land use planning/control focus, including the following:

1. Municipal and County Wellhead Protection Ordinances adopted under zoning authority, as provided by Wisconsin Act 410, made effective by the Wisconsin Legislature on May 11, 1984. In Portage County, Wellhead Protection Ordinances have been adopted by the County, City of Stevens Point and the Village's of Whiting, Plover, Junction City and Amherst.

2. Zoning Ordinance provisions, authorized by Wisconsin Act 410, that include use restrictions, lot size/density requirements, natural area requirements and site plan review standards.

3. Subdivision Ordinance requirements that prohibit development in contaminated locations, provide requirements for groundwater/well monitoring, and include provisions for community/subdivision water systems.

4. Adoption of a County Well Delegation Program that addresses the siting of new wells and abandonment requirements for old, unused wells.

5. Administration of the County Private Sewage Ordinance to insure proper on-site waste system construction and siting and the prohibition of holding tanks for new construction.

6. Regulation of septage waste land application, especially in wellhead protection areas.

7. Agricultural Waste Facility Construction Ordinance to insure proper siting and construction quality of manure storage facilities.

WATER TREATMENT: WHEN ALL ELSE FAILS

Charles P. Kell
Portage County Planning and Zoning Deparment

The cost of not protecting groundwater aquifers from pollution, including nitrogen sources is very evident for two communities in Portage County.

The Village of Whiting nitrate removal plant was constructed in 1991 to serve the Village's population of 1,785 residents at a cost of $669,909. Whiting's sytem uses a chemical anion exchange process that works similar to a water softener. The plant has three vertical pressure anion exchange fixed bed units that are four feet in diameter and eleven feet high. The Village mixes approximately 60% treated water with 40% untreated water to lower the nitrate levels from 16 ppm down to water that averages 6.5 ppm distributed to the public.

The Village of Plover's nitrate treatment plant uses a continuous absorption ion exchange process consisting of rotating ion exchange beds in a carousel. Construction cost for the plant was approximately $2,000,000. This system serves a population of 9,000 people today and is designed to serve an ultimate population of 18,000. The system can treat up to 3,600,000 gallons per day at nitrate levels in excess of 15 ppm. Annual operating costs for the plant are estimated at $50,000.

CASE STUDY OF NITRATE CONTROL: THE STEVENS POINT, WHITING, PLOVER WELLHEAD PROTECTION PROJECT

by

William Ebert, Project Manager

The Stevens Point, Whiting, Plover (SWP) Wellhead Protection Project is a USDA Hydrologic Unit Area project that focuses on non-point source pollution to a 106 square mile aquifer in Central Portage County, Wisconsin. This aquifer is the groundwater recharge area for the municipal wells serving over 40,000 people in the communities of Stevens Point, Whiting, Plover and outlying areas. The soils are sandy and the depth to groundwater is between 10 and 20 feet. Contamination from nitrates and pesticides is our primary concern.

The history of groundwater contamination in this area dates back to the late 1970's, when in 1979, Whiting's municipal water exceeded the 10 ppm NO3-N standard. They were forced to shut down their municipal well and begin purchasing water from Stevens Point. A year later, the potato insecticide, aldicarb, began showing up in private wells.

Of the 65,000 acres in the SWP area, 25,900 acres are cropland and 25,100 are woodland and wildlife areas. The remaining 14,000 acres are made up of roads, commercial, residential, pasture, idle land and surface water. Of the 25,100 acres of cropland, about 10,000 acres are under center pivot irrigation and the rest is dryland.

The association and distribution of land uses within the SWP area is not uniform, however. Most of Stevens Point's recharge area is forest, wetlands and dryland cropland. On the other hand, most of Whiting and Plover's recharge is irrigated and dryland cropland.

The main goal of the SWP Project is to encourage and facilitate voluntary adoption of Best Management Practices (BMP's) to reduce the risk of nonpoint source pollution from nitrates and pesticides. The three most pertinent BMP's are Integrated Crop Management (ICM), manure management and storage facilities, and lawn and garden nutrient and pesticide management.

In an attempt to determine how well we are protecting or improving municipal groundwater quality through the implementation of these BMP's, specifically ICM, we have selected the area within the 15 year time of travel (TOT) of the Plover recharge area for our case study area. This area was selected because it is representative of the entire recharge area and beyond this, wellhead protection areas get more indefinite. The 15 year TOT is an area we can feel comfortable with.

The questions that we wished to evaluate are:

1. What is the steady-state nitrate concentration at the Plover wells (again, using the 15 year TOT) with and without ICM?

2. If our goal is to get groundwater below the 10 ppm NO3-N standard, how much nitrate do we need to remove from the system?

3. If we need to remove more nitrate than ICM can, what other options do we have?

We can answer these questions by implementing an expensive groundwater monitoring program or (since we don't have that much money) we can estimate results by using a "mixing model". Total nitrate going into groundwater divided by total groundwater recharge will give us the steady-state nitrate concentration at the municipal wells.

We're using the Nitrogen Leaching and Economic Analysis Package (NLEAP) model, developed by the UDSA, Agricultural Research Service (ARS), Fort Collins, Colorado. NLEAP accounts for uptake of crops and leaching. It also allows for volatilization and denitrification losses, but these should be small components of the nitrogen budget in these sandy conditions.

The acreages of major land uses within the Plover 15 year TOT are as follows:

Irrigated ag.	506 acres
Nonirrigated ag.	240
Forest	340
Urban	132
Unsewered residential	35
Grass/brush	14
Roads	25
TOTAL	1292 acres

We can assign a nitrate loading rate to groundwater based on land use. Natural systems are very conservative of nitrogen, so we assign a zero nitrate-N leaching rate to Forest and Grass/brush. Further, for the type of development that has transpired in the Plover 15 year TOT, we assign zero to Urban and Roads.

Therefore, the land uses that we define nitrate-N loading rates for are irrigate ag., nonirrigated ag., and residential.

Septic systems are the major nitrate contributor in unsewered residential areas. We counted 20 residences with septic systems in the area and assumed 4 residents per household. The nitrate loading rate is 10 lbs per persons per year, so the residences account for 800 lbs of nitrate-N per year to groundwater.

For the ag component, we used the NLEAP model to calculate the amount of nitrate-N that would leach to groundwater under both conventional (CON) and ICM practices.

In the nonirrigated ag land use, we could only find 40 acres of the 240 that appeared to be actively farmed. This 40 acres is in a corn-oats-5 years alfalfa rotation, and these are the numbers that NLEAP generated for nitrate-N leaching below the root zone.

NLEAP PREDICTED NITRATE-N LEACHING (LBS/ACRE)
UNDER DRYLAND CORN-OATS-HAY

Crop	CON	ICM
Corn	41	31
Oats	15	12
Hay	0	0

Average Annual Leaching (40 ac):

CON	728
ICM	559

The nitrate loading under irrigated ag is somewhat more difficult since each operator has their own rotation. Our estimate is that the following number of acres are grown of each crop per year.

ACRES OF IRRIGATED CROP PER YEAR

Crop	Acreage
Sweet Corn	205
Potato	128
Snap bean/pea	91
Hay	41
Field corn	27
Oats	14

We again used the NLEAP model and these are the numbers it generated for nitrate-N leaching below the root zone for irrigated ag.

NLEAP PREDICTED NITRATE-N LEACHING (LBS/ACRE)
UNDER IRRIGATED CROPS

Crop	CON	ICM
Sweet Corn	87	55
Potato	138	120
Snap bean/pea	72	55
Hay	0	0
Field corn	69	46

We then multiply the predicted lbs/acre nitrate-N leaching below the root zone by the acres of each crop under irrigated ag. to get the total predicted nitrate-N loading below irrigated ag.

ACRES AND NLEAP PREDICTED NITRATE-N LOADING (THOUSANDS OF LBS) UNDER IRRIGATED CROPS

Crop	Acreage	CON	ICM
Sweet corn	205	17.8	11.3
Potato	128	17.6	15.4
Snap bean/pea	91	6.5	5.0
Hay	41	0.0	0.0
Field corn	27	1.8	1.2
Oats	14	0.2	0.2

Next, we summarized the nitrate-N loading by all contributing land uses.

SUMMARY OF NITRATE-N LOADING BY LAND USE (LBS)

	CON	ICM
Irrigated ag	44,200	31,600
Nonirr. ag	730	560
Unsewered residential	800	800
TOTAL	45,652	33,000

This summary can be expressed on a per-acre basis.

SUMMARY OF NITRATE-N LOADING BY LAND USE (LBS/ACRE)

	CON	ICM
Irrigated ag	87	63
Nonirr. ag	18	14
Unsewered residential	23	23

This summary can also expressed as a percent of the total loading.

PERCENT OF NITRATE LOADING

Irrigated ag	96.6%
Nonirrigated ag	1.6
Unsewered residential	1.8

Next, we calculated the total annual recharge, which when divided into the predicted total nitrate-N loaded will give us an estimate of the steady state nitrate-N at the Plover wells.

GROUNDWATER RECHARGE

Land use	Acres	Recharge rate	Recharge (acre-ft)
Irrigated ag	506	0.50	253
Other	786	0.83	652

TOTAL 905 Acre-ft/yr

STEADY STATE NITRATE-N AT PLOVER WELLS =

$$\frac{\text{Mass nitrate-N loaded}}{\text{Recharge}}$$

ICM	14 mg/l
CON	19 mg/l

Since both resultant steady state predictions are above the 10 mg/l standard, we asked ourselves "What else could be done under an ICM and CON scenario to yield water below the standard?"

HOW MUCH ADDITIONAL N LOADING REDUCTION TO MEET STANDARD?

ICM: 9520 LBS/YR
CON 21,000 LBS/YR

A FEW WAYS TO REDUCE LOADING BY 9520 LBS/YR BEYOND ICM

1. Convert 109 acres of irrigated land to forest or grassland.
2. Change irrigated rotation to S. corn, potato, snap bean and 2 years hay.
3. Convert 220 acres of irrigated land to 2-acre lot unsewered residential.

A FEW WAYS TO REDUCE LOADING BY 21,000 LBS/YR WITHOUT ICM

1. Convert 240 acres of irrigated land to forest or grassland.
2. Change irrigated rotation to S. corn, potato, snap bean and 4 years hay.
3. Convert 313 acres of irrigated land to 2-acre lot unsewered residential.

In conclusion, I want to emphasize that this exercise makes several assumptions and uses average rainfall and climatic data when predicting nitrate-N loading to the aquifer. In addition, there are several other ways to reduce loading to numbers that would yield water below the standard. It takes long-term monitoring money to validate computer models, such as this one. Until then, we can only estimate what is actually going on out there. The main reasons for this exercise were to quantify the impact of ICM and to open our minds toward the big picture.

The Second Million Pounds

Dr. Fred Madison
University of Wisconsin-Madison
Wisconsin Geological and Natural History Survey

One of the major ways we're attempting to deal with the question of nutrient management and ground and surfacewater is through watershed projects. The Big Springs project in northeastern Iowa represents one of the first watershed projects and has been considered to be very successful. It also provides an excellent case example of how we are failing in our mission to actually protect ground and surfacewater.

Big Springs is a 103-square mile watershed that is entirely agricultural. There are no other land uses. The geology of the area is composed of fractured dolomite with an underground flow system, appearing much like a maze of sink holes. There are no surfacewater streams. Ultimately, all of the water discharges out of a big hole in the rock which drains the entire watershed.
Ninety-five percent of the farmers in the watershed operate corn, alfalfa, livestock operations, much like what you find in Wisconsin. The average farm is about 290 acres in size. Not surprisingly, the management of nitrogen is their number one problem.

The Big Springs project began in 1981 initially as a research effort and later became a demonstration project in 1986. Its goals were to help farmers implement and improve management practices and to improve the efficiency and profitability of their operations. By 1992, when the demonstration phase was completed approximately $3.1 million had been invested in various cost-sharing practices with farmers.

Two hundred and twenty-nine farmers were surveyed in detail over the course of the project. Results showed the three to five year average corn yield in the basin to be about 130 bushels per acre, normally requiring nitrogen applications of about 160 pounds per acre. Instead farmers were shown to be applying an average of 240 pounds per acre, consistent with the over-application of fertilizer we see today throughout much of our state.

Interestingly, optimum economic yields were shown in the demonstration project during the years of 1987, '88 and '90 when less than 150 pounds of nitrogen per acre were added through the combination of legumes and/or purchased nitrogen. Seeking to reduce nitrogen applications, project staff worked to implement the standard packages of animal waste management practices, manure pits, concrete barnyards, cropland packages, some cropland reduced tillage, integrated crop and pest management. The institutional linkages between the University Extension, Soil Conservation Service, Agricultural Soil Conservation Service, Iowa Geological Survey and Iowas Department of Natural Resources were outstanding and truly

effective in getting educational programs and practices out on the land. So effective was the project, that it claimed a reduction in nitrogen inputs into the basin of one million pounds.

However, a closer look at nitrate nitrogen data in the basin between the years of 1982 and 1990 really don't show nitrate nitrogen levels varying with statistical validity any more than would normally be expected. The Big Springs project and all the projects we're doing here in Wisconsin are designed to improve water quality. In fact, these data suggest that very little has happened to impact water quality and that to truly accomplish our water quality goals we have to reduce nitrogen another million pounds.

One of the major problems is that farmers have been recalcitrant to change their practices even when they are made aware of the distinct economic advantage to do so. In 1985, a Big Springs survey revealed field application rates of purchased nitrogen to be about 50 pounds per acre on corn following alfalfa. In 1991, another survey showed the average application rate to be 91 pounds per acre, despite six years of intensive education recommending that no nitrogen be applied following alfalfa.

Our success here in Wisconsin is not any better. The Farm Practices Inventory (FPI) utilized by Professor Peter Nowak and Robin Shephard shows farmers' range of applications are between 0 and 1,600 pounds of nitrogen per acre for corn with an average of about 203 pounds. Recommended applications are 160 pounds per acre. Results also show that many individuals are failing to do any nitrogen crediting for manure applications whatsoever, and that only 4 to 5% of farmers are accurately crediting legumes.

Educational efforts like the nutrient and pest management program that help farmers to weigh manure spreaders and actually provide farmers with a sense of how much nitrogen is going on the land through manure alone are very important. Additional work is needed in this area to help change farmers' notions that manure is something simply to be hauled away, and that it is in fact a substantial source of nutrients. Likewise, considerable attention needs to be devoted to helping more farmers accurately credit legumes. Attempts to address both have been focused in the Narrows Creek Watershed, where the FPI survey helped pinpoint the specific educational needs of farmers in that area.

Another important concept is that of fertilizer efficiency. A Canadian study performed at the University of Gwelth looking at no till and conventional till production of corn found an interesting pattern. Data comparing corn yield with nitrogen applications as well as dollars spent on fertilizer, showed that beyond a critical point the amount of return for money spent begins to drop off. In other words, plants may only be capable of utilizing so much nitrogen. The Canadian researchers further suggest that what is needed is an "environmentally acceptable" fertilization rate that recognizes where yield increments begin to decline.

This concept is an interesting one and worthy of exploration. As we search for new directions in agriculture, we need to begin looking at it as a system and not just crops. The question is can you take the money that would be freed up by putting on 100 lbs. of nitrogen versus 150 lbs. and invest it someplace else in the enterprise and net a better economic return? Perhaps.

The nutrient management question is an interesting one and truly the challenge ahead. Most watershed projects have implemented many engineering projects, but haven't solved many problems. We're at a point now where we can begin to see that engineering solutions are less important than addressing the behavioral practices of farmers. Though we may not have all the answers yet to solve all of the problems, it is clear that in order to achieve our water quality goals we have got to find a way to address that second million pounds of nitrogen.