APPENDIX C

Montana Department of Environmental Quality

Estimating Natural Attenuation of Nitrate and Phosphorus from On-Site Wastewater Systems

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ABSTRACT

Estimating the fate of nutrients (nitrate and phosphorus) in groundwater from on-site wastewater treatment systems (OWTS) has not been practicable because of the costs and uncertainty associated with determining site-specific degradation rates, groundwater flow paths, and aquifer hydraulic properties. Existing watershed models allow users to specify natural degradation rates for nutrients from OWTS but provide little or no guidance as to how users should determine those rates. A simple spreadsheet analysis, Method for Estimating Attenuation of Nutrients from Septic Systems (MEANSS), has been developed to provide that guidance. MEANSS is designed to estimate the load of nutrients that will migrate to surface waters from OWTS sources. When evaluated against several field studies, a GIS-based nitrate loading model, and a watershed model, MEANSS provided comparable estimates of nitrate and ortho-phosphorus attenuation in the subsurface

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ACRONYMS

ArcNLET	Arc nitrate loading estimation toolkit
MEANSS	method for estimating attenuation of nutrients from septic systems
MDEQ	Montana Department of Environmental Quality
NHD	national hydrography dataset
NRCS	Natural Resources Conservation Service
OWTS	on-site wastewater treatment system
ortho-P	ortho-phosphorus
SWAT	soil and water assessment tool
TN	total nitrogen
TMDL	total maximum daily load
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey

1.0 INTRODUCTION

The state of Montana has been developing numeric nutrient (nitrate and phosphorus) water quality standards over the past decade that are based on: 1) stressor-response studies performed by Montana Department of Environmental Quality (MDEQ) to determine the maximum nutrient concentrations that will maintain algal growth below undesirable levels, 2) a literature review of stressor-response studies, 3) a comparison of nutrient stressor-response thresholds to eco-regionally stratified reference data from Montana (Suplee et al., 2007), and 4) consideration of nutrient ratios (Redfield, 1958). These numeric standards are proposed to replace the current narrative standards and would be set at relatively low concentrations which would be difficult for wastewater treatment systems that discharge to surface water to achieve from both a treatment and financial perspective. Development of Total Maximum Daily Loads (TMDLs) for water bodies that are impaired for nutrients will also be affected by the new numeric standards.

A potentially significant source of nutrients in some watersheds is on-site wastewater treatment systems (OWTS). Predicting the fate of nutrients discharged from OWTS has proven difficult to assess quantitatively. To facilitate implementation of the new nutrient standards, a quantitative screening tool to estimate nutrient loading from OWTS to surface waters was developed. The Method for Estimating Attenuation of Nutrients from Septic Systems (MEANSS) was created to meet this need primarily for use in nutrient trading and for TMDL development.

Nutrient trading has been promoted by the United States Environmental Protection Agency (USEPA) (U.S. Environmental Protection Agency, 2004) to help municipalities and industries that discharge to surface waters to comply with stricter water quality standards that are being implemented or considered in many states. One form of nutrient trading involves connection of OWTS to a permitted wastewater treatment facility. The connection of an OWTS removes a source of nutrients to groundwater and surface water, and then transfers that load to the treatment facility. A wastewater treatment facility receives a nutrient trade credit for each OWTS it connects because a facility with tertiary treatment can typically treat nutrients (particularly nitrate) to a lower concentration than an OWTS. The trade credit gives the treatment facility incentive to connect OWTS by increasing the effluent discharge limit in their wastewater discharge permit. To implement a trade, the load of nutrients from the OWTS that impact surface water must be estimated to determine a proper trade ratio to apply in the wastewater discharge permit. Defining the trade credit value can be difficult because the factors that control soil attenuation rates are both spatially variable and difficult to measure in the subsurface. MEANSS is designed to estimate the natural attenuation of nutrients that occurs as treated wastewater migrates from OWTS through soil and eventually discharges into surface water.

2.0 BACKGROUND

The criteria used to develop MEANSS was: 1) it was easy to understand and use; 2) it was operational using accessible existing information; and 3) it used site-specific information which incorporates factors known to control natural attenuation. MEANSS uses steady-state conditions as a simplifying condition, it does not account for the lag time needed for the treated wastewater from an OWTS to migrate into the receiving surface water (all OWTS included in the model are contributing wastewater to the surface water). Another simplifying condition is that all the treated wastewater is assumed to enter the user-specified receiving surface water, there is no factor included for wastewater that does not discharge to the surface water due to the local groundwater/surface water interactions. However, the user may easily incorporate such a factor into the results of MEANSS in areas where information on the groundwater/surface water interaction is available.

Based on field studies denitrification rates vary over at least three orders of magnitude (McCray et al., 2005); other studies show that denitrification rates can vary considerably even within similar environments (Robertson et al., 1991; Starr and Gillham, 1993). Because of this high degree of variability and because MEANSS does not use site-specific measured rates of attenuation, the results of MEANSS should be used carefully as an approximation of actual attenuation rates on a watershed-scale basis only.

The factors that affect the natural attenuation of nitrate and phosphorus are described here to provide the basis for MEANSS.

2.1 NITRATE

Nitrogen in raw domestic wastewater (in a septic tank) is primarily in the form of ammonia. Disposal of untreated wastewater in a properly constructed and sized drainfield will typically provide sufficient oxygen and naturally occurring bacteria to convert the ammonia to nitrite and then quickly to nitrate. Studies and regulations commonly assume that most or all the nitrogen is converted to nitrate after proper septic tank and drainfield (conventional) treatment (Morgan and Everett, 2005; Heatwole and McCray, 2006; Howarth et al., 2002; Montana Department of Environmental Quality, 2009; Morgan et al., 2007; Toor et al., 2011). Conventional OWTS are not designed to complete the final step of nitrogen treatment, denitrification, which is the conversion of nitrate to nitrogen gas. Nitrogen gas is able to dissipate into the atmosphere and does not have any further impacts to groundwater or surface water. While there may be some minor denitrification in the bio-mat associated with a properly operating drainfield, denitrification primarily occurs after the treated wastewater migrates away from the drainfield.

For denitrification to occur a suitable environment must exist; key factors are adequate temperature (typically above 10 ^{o}C), a food source for the bacteria (that food source is typically carbon, which is related to the soil organic content), an anoxic environment (generally an oxygen range <1-2 milligrams per liter (mg/l)), and the correct bacteria. A riparian zone with shallow groundwater is the most common natural environment that has those conditions (Gold and Sims, 2000; Rosenblatt et al., 2001; Gilliam, 1994; Harden and Spruill, 2008; McDowell et al., 2005). Riparian zones are associated with enhanced potential for groundwater denitrification (Dubrovsky et al., 2010). Studies have identified “micro-sites” of low oxygen in shallow groundwaters (which have often been assumed to be rich in oxygen and therefore poor environments for denitrification) to provide the anoxic environment required for denitrification (Parkin, 1987; Jacinthe et al., 1998; Gold and Sims, 2000). The required bacteria are generally ubiquitous in the environment, and will naturally thrive when the conditions are correct and there is a nitrate source. Although a literature review has not provided any specific lower limit of carbon concentration below which denitrification does not occur, an adequate carbon source is cited as the most common limiting factor for denitrification (Gold and Sims, 2000; Kobus and Kinzelbach, 1989; Rivett et al., 2008). MEANSS accounts for this limiting factor through the use of site-specific soil composition characteristics.

Because fine-grained soils are more likely to contain two of the conditions necessary for denitrification, anoxic conditions and carbon, they typically provide better denitrification conditions than coarse-grained soils (Umari et al., 1995; Tesoriero and Voss, 1997; Briar and Dutton, 2000; Mueller et al., 1995). (Anderson, 1998) uses results from several studies to show a correlation (r=0.91) between denitrification rates and soil organic content. Another study (Ricker et al., 1994) uses this relationship to estimate the amount of denitrification beneath drainfields as 15 percent for sandy soils and 25 percent for finer soils. Both (Long, 2014; Roeder, 2008) use soil types to estimate total nitrogen reductions from OWTS discharges. The hydrologic soil groups (HSGs) defined by the Natural Resources Conservation Service (NRCS) are divided into four groups (A, B, C, or D) and are used in the MEANSS tool. Using logistic regression analysis HSG was correlated to probability of groundwater nitrate contamination (Frans, 2000 and Nolan et. al., 2002). The four groups are based on runoff potential of soil during maximum wetness conditions without accounting for effects of freeze conditions, vegetative cover, or slope (Natural Resources Conservation Service, 2007). Although the NRCS uses additional criteria for the HSG designation, the amount of clay is an important part of the designation and generally uses the following criteria. Group A soils have less than 10 percent clay materials. Group B soils have 10 to 20 percent clay materials. Group C soils have 20 to 40 percent clay materials. Group D soils have greater than 40 percent clay materials. The soil types that correspond to the HSGs can be approximated using the United States Department of Agriculture textural triangle (Soil Conservation Service, 1987). Soils with higher clay content tend to have more carbon and thus can provide an environment better for denitrification (these soils also have lower permeability which allows more time for denitrification to occur). MEANSS uses the HSG to determine the relative amount of clay in the soil and varies the estimated amount of nitrate attenuation accordingly.

The absence or presence of hydric soils may also be used to determine a relative rate of denitrification (Gold et al., 2001). Although not specifically included in the MEANSS spreadsheets, when site specific data is available for hydric versus non-hydric soil the user can account for higher nitrate removal rates in soils classified as hydric or use hydric soil presence/absence instead of the hydrologic soil types. Using hydric soil criteria may be particularly useful if the carbon content of site soils do not correlate well with the HSG.

Travel time in the environment (primarily in groundwater) is another factor that has been correlated to denitrification; as nitrate persists in the environment it has more time to encounter conditions conducive to denitrification (Kroeger et al., 2006). However, distance is used in MEANSS instead of travel time because it is easier to measure distances than the three parameters that control groundwater travel time: hydraulic gradient, hydraulic conductivity and effective porosity. Also, increased distances from surface water often allow for deeper groundwater flow which, increase the chances of encountering anoxic conditions that are conducive to denitrification (Dubrovsky et al., 2010). The use of travel time and measured denitrification rates in the groundwater were considered when developing MEANSS. However, that method was not used for three reasons. First, denitrification rates are site-specific and the rates can vary considerably in similar environments (Robertson et al., 1991; Starr and

Gillham, 1993). Second, although several studies include a specific denitrification rate (Kirkland, 2001; Siegrist et al., 2005; McCray et al., 2005) that is based on the median of cumulative frequency distributions of field measured denitrification rates (0.025 day-1); that denitrification rate cannot be correlated to soil type due to variability in the data (McCray et al., 2005). Third, estimating travel time requires collecting site-specific data for hydraulic conductivity, hydraulic gradient, and porosity which is often difficult and expensive to measure accurately.

Review of the existing literature presented above provided three factors that are used in MEANSS to estimate nitrate attenuation: 1) the predominant HSG beneath the drainfield; 2) the predominant HSG in the riparian zone of the receiving surface water; and 3) the distance between the drainfield and the receiving surface water. The numeric reduction values applied to these characteristics are presented in the Methods section.

2.2 PHOSPHORUS

Phosphorus, which has lower mobility than nitrogen, is removed in soils by two primary processes, adsorption and precipitation. The vadose zone is considered the primary location for phosphorus attenuation due partially to the negative soil moisture potentials that pushes the treated wastewater into the finer soil interstices and promotes phosphorus adsorption and precipitation(U.S. Environmental Protection Agency, 2002). Finer-grained soils also tend to retard phosphorus migration more than coarser soils due primarily to their greater surface area that provides more locations for adsorption. The HSG of the predominant soil beneath the drainfield is used to determine the relative amount of fine-grained soil. MEANSS does not distinguish between precipitation and adsorption, it applies a single reduction combining the two processes.

Non-calcareous soils retard the movement of phosphorus more than calcareous soils because calcareous soils commonly maintain neutral pH levels where phosphorus precipitation does not readily occur (Robertson et al., 1998; Lombardo, 2006). Typically, non-calcareous soils are derived from igneous or metamorphic parent rocks. Calcareous soils have been defined (Lombardo, 2006) as those containing more than 15 percent calcium carbonate and non-calcareous soils as those containing less than 1 percent calcium carbonate. MEANSS uses those calcium carbonate divisions to adjust the amount of attenuation occurring in each soil type (see Methods section).

Using similar logic as described for nitrate, distance is used as a criterion for phosphorus attenuation. However, for the same distance a larger amount of reduction is applied to phosphorus than nitrate in MEANSS. Phosphorus is treated differently because treated wastewater plumes with high phosphorus concentrations have been found to extend a relatively short distance from the source, creating high concentrations of phosphorus in soils immediately below drainfields with low levels beyond that location (Makepeace and Mladenich, 1996; Robertson et al., 1998; Gold and Sims, 2000; Reneau, Jr. et al., 1989; Lombardo, 2006).

Riparian areas, where anaerobic conditions often exist and are conducive to denitrification, provide a poor environment for phosphorus reduction. This environment can release some sediment bound phosphorus (Vought et al., 1994), therefore riparian soil conditions are not used in the estimation of phosphorus attenuation as they are for nitrate.

Review of the existing literature presented above provided three factors that are used in MEANSS to estimate phosphorus attenuation: 1) predominant HSG at the drainfield area; 2) calcium carbonate content of the soils in the drainfield area; and 3) distance between the drainfield and the receiving surface water. The numeric reduction values applied to these characteristics are presented in the Methods section.

3.0 METHODS

The parameters used in MEANSS are available through: GIS mapping for distance values; the United States Geological Survey (USGS) National Hydrography Dataset (NHD) set to determine appropriate receiving surface waters; and the NRCS STATSGO2 or SSURGO soils databases to determine the HSG and soil calcium carbonate content. GIS tools can be used to determine the soil characteristics at each drainfield and in the 100 foot riparian buffer (see Figure 1). When available, local detailed soil information should be used to confirm the NRCS soils database information.

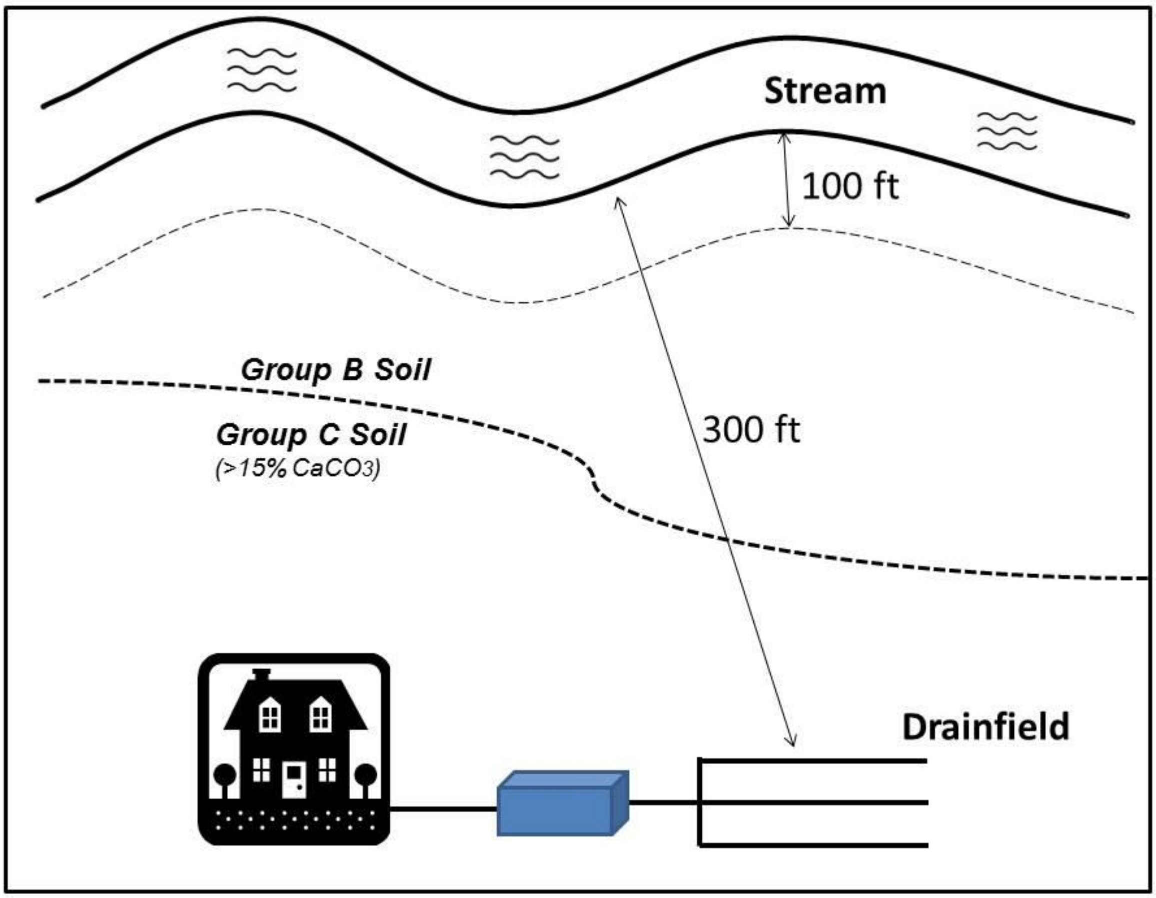

Figure 1 – Schematic of MEANSS parameters

The distance to nearest receiving surface water is typically the most uncertain parameter required in MEANSS. The NHD information can provide the locations of ephemeral, intermittent and perennial streams, but without a detailed groundwater flow map it can be difficult to determine the direction of groundwater movement and where shallow groundwater will intersect surface water. When site-specific data are not available, an option is to use only perennial streams in the NHD data and assume the shortest distance between the OWTS and surface water for the distance value.

The HSG used in MEANSS is based on the NRCS STATSGO2 or SSURGO classification of the predominant soil type at the drainfield and within 100 feet of the receiving surface water. The 100 foot stream buffer is used as the default width to determine predominant soil types in the riparian area.

The soil calcium carbonate content used in MEANSS is also based on values in the NRCS STATSGO2 or SSURGO database. In some areas the calcium carbonate content is not available through those databases. In those cases, users will have to decide how to best estimate this value. Some options include: on-site measurements, use of a supplemental database, estimation of the calcium carbonate content based on the local geology, or using the middle of the three ranges for calcium carbonate concentration in the phosphorus spreadsheet. Less accurate methods would increase the uncertainty of the results.

3.1 Nitrate Spreadsheet

The spreadsheet for nitrate attenuation is presented in Table 1.

Table 1 – Nitrate Attenuation Factors for OWTS Discharges to Soil

	Scoring Category 1	*Scoring Category 2*	*Scoring Category 3*
Percent Nitrate Load Reduction(1)	**Soil Type at Drainfield(2)**	**Soil Type within 100 feet of Surface Water(2)**	**Distance to Surface Water (feet)**
0	A	A	0 – 100
10	B		101 – 500
20	C	B	501 – 5,000
30	D	C	5,001 – 20,000
50		D	20,000+

Table 1 Notes:
(1) The total nitrate reduction is the sum of the individual reductions for Category 1 + Category 2 + Category 3. For example (see Figure 1) a drainfield that is in a group C soil (20 percent) that drains to a surface water with group B riparian soil (20 percent) and is 300 feet from the surface water (10 percent) would reduce their nitrate load to the surface water by 50 percent from the load that is discharged from the drainfield.
(2) Soil descriptions are available via the NRCS web soil survey at: http://websoilsurvey.nrcs.usda.gov/app/HomePage.htm . Once the area of interest (AOI) has been defined information is accessed by selecting the following links: “Soil Data Explorer” – “Soil Properties and Qualities” -- “Soil Qualities and Features” – “Hydrologic Soil Group”. Alternatively, the NRCS Soil Data Viewer program may be used in an external GIS application. Soil Data Viewer is available at: http://www.nrcs.usda.gov/wps/portal/nrcs/detailfull/soils/home/?cid=nrcs142p2_053620 .

3.2 Phosphorus Spreadsheet

The spreadsheet for phosphorus attenuation is presented in Table 2.

Table 2 - Phosphorus Attenuation Factors for OWTS Discharges to Soil

	Scoring Category 1			*Scoring Category 2*
Percent Phosphorus Load Reduction(1)	**Soil Type at Drainfield(2, 3) (CaCO3 <= 1%)**	**Soil Type at Drainfield(2, 3) (CaCO3 >1% and <15%)**	**Soil Type at Drainfield(2, 3) (CaCO3 >=15%)**	**Distance to Surface Water (feet)**
10	A	A	A	0 – 100
20			B	

40		B	C	
50				101 – 500
60	B	C	D	
80	C	D		501 – 5,000
100	D			5,000 +

Table 2 Notes:
(1) The total phosphorus reduction is the sum of the individual reductions for Category 1 + Category 2. For example (see Figure 1) a drainfield that is in a type C soil with greater than 15 percent CaCO3 (40 percent) and is 300 feet from the surface water (50 percent) would reduce their phosphorus load to the surface water by 90 percent from the load that is discharged from the drainfield.
(2) Soil descriptions are available via the NRCS web soil survey at:
http://websoilsurvey.nrcs.usda.gov/app/HomePage.htm . Once the area of interest (AOI) has been defined information is accessed by selecting the following links: "Soil Data Explorer" – "Soil Properties and Qualities" -- "Soil Qualities and Features" –"Hydrologic Soil Group". Alternatively, the NRCS Soil Data Viewer program may be used in an external GIS application. Soil Data Viewer is available at:
http://www.nrcs.usda.gov/wps/portal/nrcs/detailfull/soils/home/?cid=nrcs142p2_053620 .
(3) CaCO3 percent is available via the NRCS web soil survey (or via the Soil Data Viewer) at:
http://websoilsurvey.nrcs.usda.gov/app/HomePage.htm . Once the area of interest has been defined information is accessed by clicking on following links: "Soil Data Explorer" – "Soil Properties and Qualities" -- "Soil Chemical Properties" – "Calcium Carbonate (CaCO3)".

3.3 Model Performance

The accuracy of MEANSS was evaluated by comparing it to the results of four studies that measured nitrate concentrations in groundwater to determine OWTS impacts. Groundwater studies were used instead of surface water studies because surface water nutrient concentrations are typically complicated by nutrient cycling and nutrient uptake which makes it difficult to define the load attributable to a specific source. MEANSS was also compared to another program (Rios, et al., 2012) designed to estimate nitrate surface water loadings from OWTS, Arc Nitrate Loading Estimation Toolkit (ArcNLET). As a final method of validation, a watershed model was created using the Soil and Water Assessment Tool (SWAT) (Arnold et al., 1993)(Arnold et. al., 1993) for a small watershed in Montana. Results from MEANSS were used to produce a model simulation that is calibrated to observed in-stream concentrations of both total nitrogen (TN) and ortho-phosphorus (ortho-P). A lack of adequate phosphorus studies limited the evaluation of MEANSS phosphorus attenuation estimates to the SWAT watershed model.

The estimated nitrate and ortho-P loads used for evaluating MEANSS performance are 13.8 and 2.92 kilograms per year (kg/yr) per single family OWTS, respectively. These loads are based on averages of published treated wastewater characteristics (U.S. Environmental Protection Agency, 2002; Montana Department of Environmental Quality, 2009).

3.3.1 Site 1

The study site (Boer, 2002) is a low-density residential area near Lolo, Montana that is 1,600 acres in size and contains over 500 single-family OWTS. The study used site specific data for the hydraulic conductivity, hydraulic gradient, and groundwater nitrate concentrations to estimate the amount of OWTS-related nitrate migrating from the study area. Other potential sources of anthropogenic nitrate noted in the report were domestic lawn fertilizer and a 60 acre septage application site, those sources were calculated to be an order of magnitude less than the potential loading from OWTS and were not evaluated further in the study. After accounting for the natural background concentration of nitrate in the groundwater (estimated as 0.1 mg/l nitrate-N), the total OWTS-related nitrate groundwater load of 13,454 lb/yr was calculated using the information in the study.

3.3.2 Site 2

A study to estimate nitrate loading to the Bitterroot River from OWTS in and around the city of Missoula (Miller, 1996) is used for the second site. The study estimated the groundwater flux based on a groundwater model as 59 cubic feet per second (Miller, 1991). Groundwater concentrations in the study were based on eight groundwater wells and eleven groundwater seep samples collected in August 1995. Only the groundwater seeps were used in this analysis because they were closer to the river and therefore provided a better estimate of the nitrate concentrations entering the river. The average nitrate+nitrite-N concentration of the eleven seeps was 1.04 mg/l (for purposes of this analysis, the nitrate+nitrite concentration is assumed to consist entirely of nitrate). After accounting for the natural background concentration of nitrate in the groundwater (estimated as 0.1 mg/l nitrate-N), the total OWTS-related nitrate groundwater load entering the river of 109,239 lb/yr was calculated for this analysis.

3.3.3 Site 3

In Spanish Springs Valley, Nevada, 38 lysimeters were installed beneath four drainfields to measure the effluent characteristics during soil treatment (Rosen et al., 2006). Three of the drainfields serve single-family homes, the fourth serves a school. All of the drainfields had deep trenches (6.5 to 10 feet deep) due to low permeability soil near the surface. With deep trenches, the fill material essentially becomes the drainfield soil type. State regulations (, 2008) require clean sands and gravels with less than 5 percent fines for the material in absorption trenches. This type of soil was estimated as a B soil because an A soil typically has percolation rates that are too fast for proper effluent treatment, therefore a B soil was used in the MEANSS analysis.

Eighteen deep lysimeters were placed in the native soil beneath the bottom of the drainfield trenches (the deep lysimeters had more consistent results than the shallow ones and are used in this analysis). The deep lysimeters were sampled monthly from July 2004 to January 2006; the median TN concentration was 44 mg/l (for purposes of this analysis it is assumed that the TN measured in the study is entirely nitrate. Using Montana's estimated total nitrogen concentration entering the drainfield of 50 mg/l (Montana Department of Environmental Quality, 2009), the measured nitrate-N reduction after soil treatment in the study is 6 mg/l, or 12 percent.

3.3.4 Sites 4 and 5

Two Wisconsin subdivisions were monitored for nitrate impacts to groundwater (Shaw et al., 1993). Several multi-port groundwater wells were used to measure the three-dimensional extent of nitrate impacts to groundwater from selected portions of the subdivisions. Using a model called BURBS (Hughes and Pacenka, 1985) the authors estimated that approximately 20 percent of the nitrate load measured in the groundwater was from lawn fertilizer use (the relatively higher impacts of lawn fertilizer at these sites as compared to Site 1, is likely due to the higher density of homes in Sites 4 and 5). The study used phosphorus and fluorescence in the multi-port monitoring wells to separate the groundwater being impacted from upgradient sources (deeper water) versus groundwater impacted by the subdivisions. The study calculated low, medium and high groundwater flow rates beneath the subdivisions to determine loading rates. Using the medium flow rates for this analysis, the nitrate groundwater load from the Jordan Acres (26 homes) and Village Green (45 homes) OWTS are 529 and 1,358 lb/yr, respectively.

3.3.5 ArcNLET

MEANSS was compared to a travel time based method of estimating nitrate attenuation from OWTS, ArcNLET. ArcNLET is a GIS-based program that estimates nitrate reduction from OWTS using groundwater velocity rates (calculated from site-specific hydraulic conductivity, hydraulic gradient and porosity), and a user-defined denitrification rate (Rios et al., 2012).

An ArcNLET analysis was completed using the data from evaluation site #2 in Missoula. The same OWTS spatial information used for the MEANSS analysis was also used for ArcNLET. For the ArcNLET analysis the hydraulic conductivity and hydraulic gradient from Miller (1991) were used. The hydraulic conductivity ranged between 2,000 and 14,490 ft/day, the hydraulic gradient ranged from 0.001 to 0.003 ft/ft, and a porosity of 12 percent was estimated using the lower end of the range for sand and gravel aquifers (Driscoll, 1986). The denitrification rate suggested in the ArcNLET documentation, 0.008 day-1, was used. Using those parameters, ArcNLET was used to estimate a total nitrate load to the Bitterroot River of 16,134 kg/yr.

3.3.6 SWAT Model

A SWAT watershed model was prepared for a portion of the Prickly Pear watershed in central Montana (Figure 2). The Prickly Pear watershed was chosen because it has a sufficient number of OWTSs (approximately 1,010) for the size of the watershed (131,200 acres) to create noticeable impacts to stream quality. In addition, there is little industrial or agricultural development in this watershed above the USGS streamflow gage near the town of Clancy that could potentially mask the impacts from OWTS.

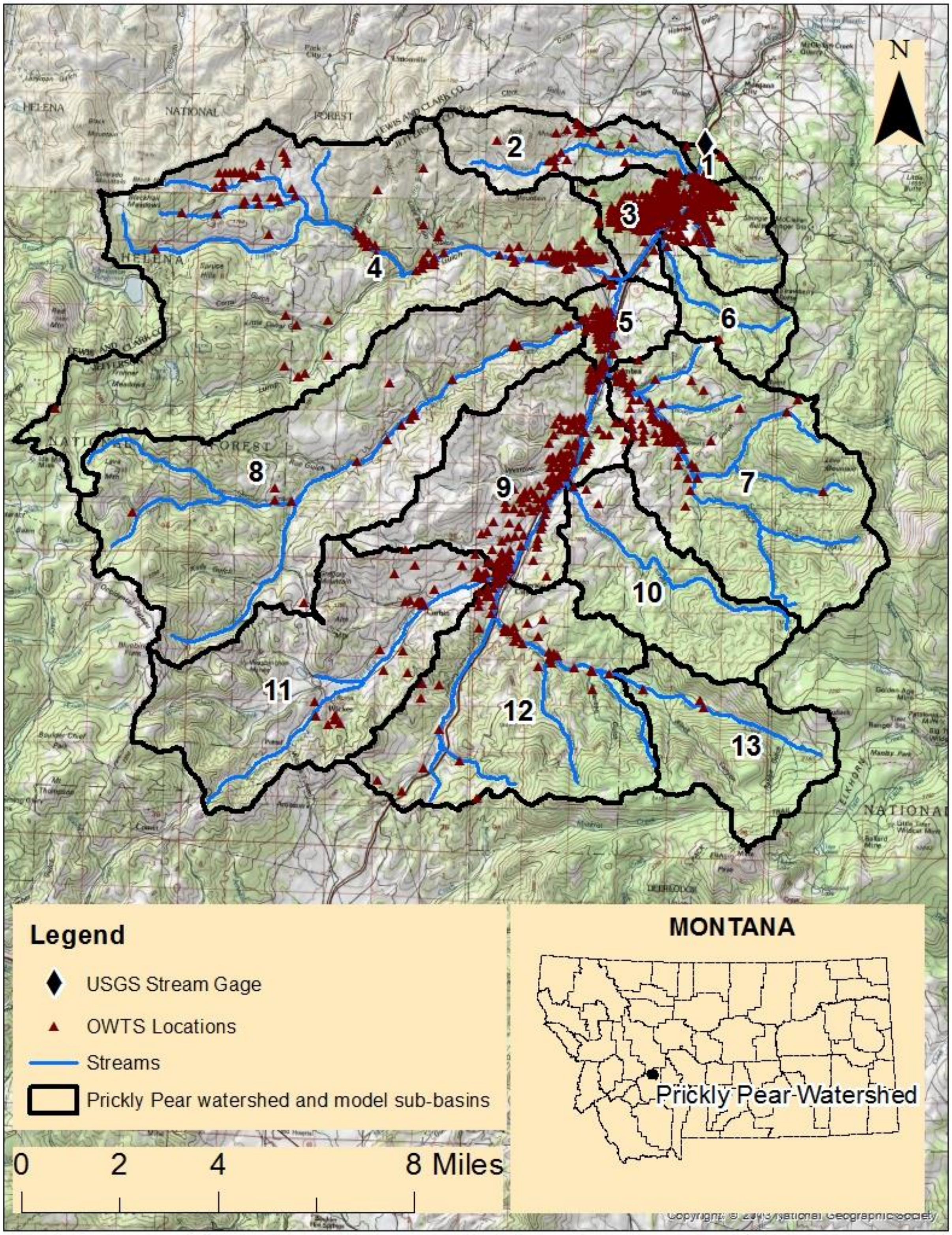

Figure 2 – Prickly Pear watershed

The SWAT model was developed using available information for elevation, landuse/landcover, soils and streamflow. The hydrology was calibrated to daily streamflow values measured at the USGS Prickly Pear near Clancy MT gage (06061500) which was also used as the outlet for the model. The calibration period was 1992 through April 2013, daily measured streamflow was available for 82 percent of the calibration period. Daily error statistics of relative error, coefficient of determination, and Nash-Sutcliffe coefficient of efficiency were -9.0 percent, 0.76 and 0.76, respectively. All three statistics indicate a good match between measured and simulated streamflow values.

SWAT contains a OWTS biozone algorithm designed to simulate nitrogen, phosphorus, bacteria and biological oxygen demand discharges from septic tank effluent (Jeong et al., 2011). In-stream ortho-P and TN observed concentrations were compared to the SWAT biozone results and to the MEANSS results (for this comparison the MEANSS results were added to a SWAT simulation that did not include any OWTS system discharges). The in-stream data consisted of 20 samples collected from 1999 through 2003 by the USGS, and three cold-weather samples (February, March and April) collected by MDEQ in 2013. The winter samples were collected for this study to determine in-stream concentrations while in-stream nutrient cycling was at a minimum, however based on the limited sampling the cold weather sample concentrations were not noticeably different than those collected during warmer months.

4.0 RESULTS

4.1 SITE 1

The MEANSS analysis used a 2008 database provided by the Missoula Valley Water Quality District to extrapolate the number of single-family homes that existed in 2001, 558 homes. The analysis was run without using the third nitrate parameter (soil type within 100 feet of surface water) because the groundwater data was primarily from wells not within the 100 foot riparian buffer. MEANSS was used to calculate a nitrate reduction of 41.5 percent, which provides a total nitrate load entering the river of 9,956 lb/yr from the 558 homes. The MEANSS load is 74 percent of the load estimated from the thesis (Boer, 2002), 13,454 lb/yr.

4.2 SITE 2

The MEANSS analysis used the 2008 database provided by the Missoula Valley Water Quality District to extrapolate the number of single-family OWTS that were contributing treated wastewater to the river in 1995, 4,315 homes. The analysis was run without using the third nitrate parameter in Table 1 (soil type within 100 feet of surface water) because the groundwater data was primarily from springs not within the 100 foot riparian buffer. MEANSS was used to calculate a nitrate reduction of 43.7 percent, which provides a nitrate load entering the river of 74,095 lb/yr. The MEANSS load is 68 percent of the amount estimated from the report (Miller, 1996), 109,239 lb/yr.

The city of Missoula uses a dry well disposal system for stormwater runoff. Much of this stormwater is likely mixed in the groundwater with OWTS discharges and therefore included with the estimated nitrate load (Miller, 1996). An accurate calculation of the stormwater nitrate load was not possible with the available information, but is estimated as several thousand lb/yr (stormwater includes nitrate from various sources such as lawn fertilizer runoff and atmospheric deposition). If that estimate is accurate, the nitrate loads estimated with MEANSS would provide a better match with the measured nitrate loads.

4.3 SITE 3

The MEANSS analysis was run only using the first criteria for nitrate reduction in Table 1 (soil type at the drainfield) because the study measured the nitrate reduction immediately beneath the drainfield. MEANSS was used to estimate a 10 percent nitrate reduction based on B-type soils in the drainfield, which is 83 percent of the measured reductions, estimated at 12 percent.

4.4 SITES 4 AND 5

MEANSS was run without using the third nitrate parameter in Table 1 (soil type within 100 feet of surface water) because the groundwater data was not collected within the 100 foot buffer of surface water. MEANSS was used to estimate a 19.6 percent nitrate reduction at Jordan Acres which is equal to a groundwater load of 637 lb/yr; the MEANSS load is 120 percent of the measured load, 529 lb/yr. MEANSS was used to estimate a nitrate reduction of 18.4 percent at Village Green which is equal to a groundwater load of 1,120 lb/yr; the MEANSS load is 82 percent of the measured load, 1,358 lb/yr.

4.5 ArcNLET Comparison

The ArcNLET program estimated the nitrate load into the Bitterroot River as 35,568 lb/yr, the MEANSS estimate is over double that load, 74,095 lb/yr. The estimate of aquifer parameters can have a significant effect on the ArcNLET results. If, for example, the high end of the published porosity values were used instead of the low value (25 percent instead of 12 percent) the ArcNLET program estimates a reduced nitrate load of 22,600 lb/yr. The choice of the denitrification rate can also have significant impacts to the ArcNLET results. For example, if a denitrification rate of 0.025 day-1 is used instead of 0.008 day-1 the ArcNLET nitrate estimated load is reduced to 8,832 lb/yr. A denitrification rate of 0.025 day-1 is based on the median value from a cumulative frequency distribution of measured natural denitrification rates (McCray et al., 2005).

4.6 SWAT Model

Incorporating the MEANSS loading estimates into the SWAT model showed that the lack of seasonal variation in MEANSS results created unreasonably large ortho-p and TN values in the winter months during baseflow conditions. To provide better seasonal variation for the MEANSS results, it was assumed that groundwater contribution to streams varied proportionally with stream flow (i.e. there is a higher volume of groundwater contribution, and OWTS effluent, to streams during the spring when groundwater elevations are higher than during other months of the year when ground water elevations recede). The annual loads estimated using MEANSS were proportionally divided on a monthly basis (although variable by month, the sum of the monthly loads remained equal to the MEANSS annual load) to match the monthly variation of streamflow at the USGS streamflow gage. Accounting for seasonal variation resulted in good calibration to the measured data for both TN and ortho-P. Statistical summaries of the simulated and observed concentration data are based on 23 sample dates (19 dates for the load values) and are provided in Figure 3. Time series plots of the observed versus simulated results are presented in Figure 4. A SWAT model was also run by replacing the MEANSS results with the biozone algorithm that is included with SWAT to predict phosphorus and nitrogen leaching from OWTS. The SWAT biozone results showed TN concentrations an order of magnitude lower than the measured in-stream concentrations. For phosphorus SWAT only calculates the amount of phosphorus that is discharged from the soil profile, it does not simulate additional phosphorus migration into groundwater and surface water. The amount of phosphorus discharged from OWTS that is estimated to enter surface waters (247.6 lb/yr) in the MEANSS results is similar to (114 percent) of the phosphorus estimated to leach from the bottom of the soil profile (216.7 lb/yr) in the SWAT results. Additional reductions in the SWAT phosphorus loads would be expected if SWAT simulated the migration and attenuation of phosphorus in groundwater as it migrates into surface waters.

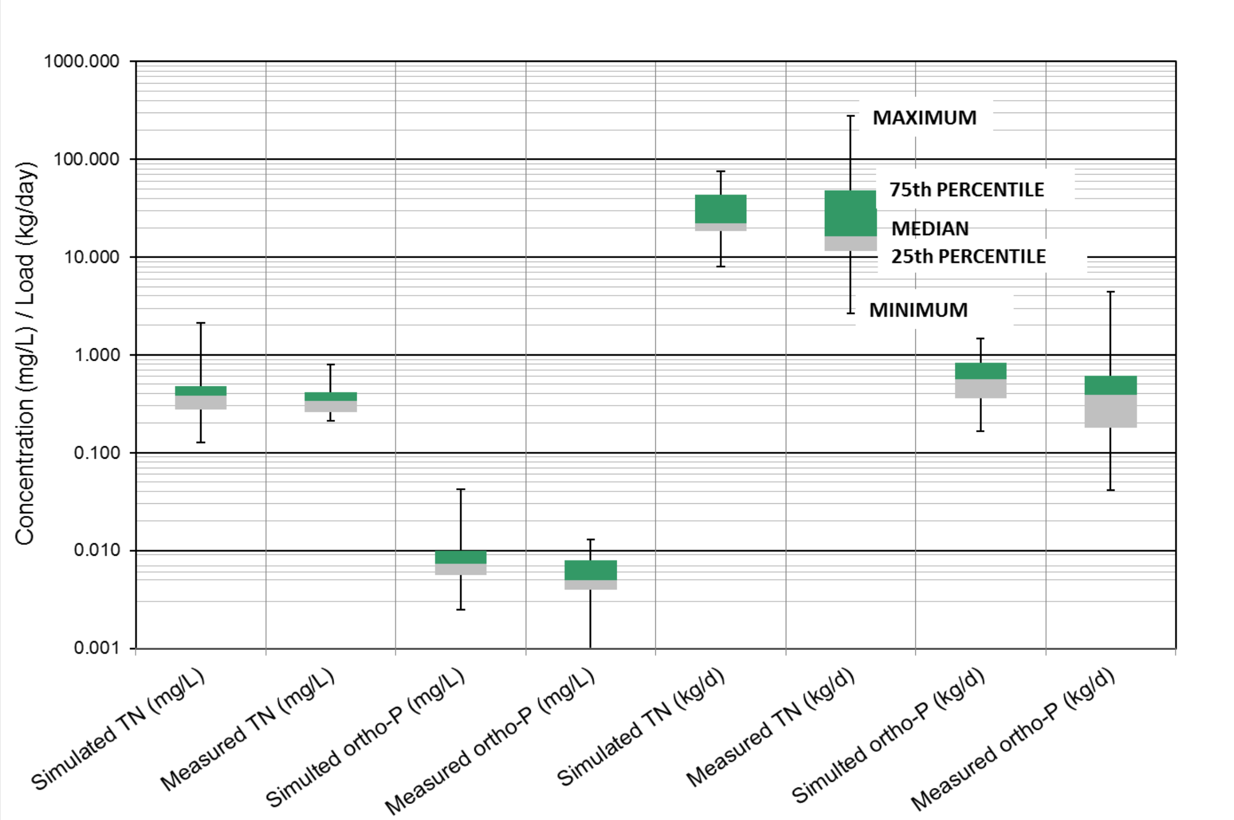

Figure 3 – SWAT model calibration results for Total Nitrogen and Ortho-Phosphorus. Based on 23 dates of observed in-stream concentrations and 19 dates of observed in-stream loadings.

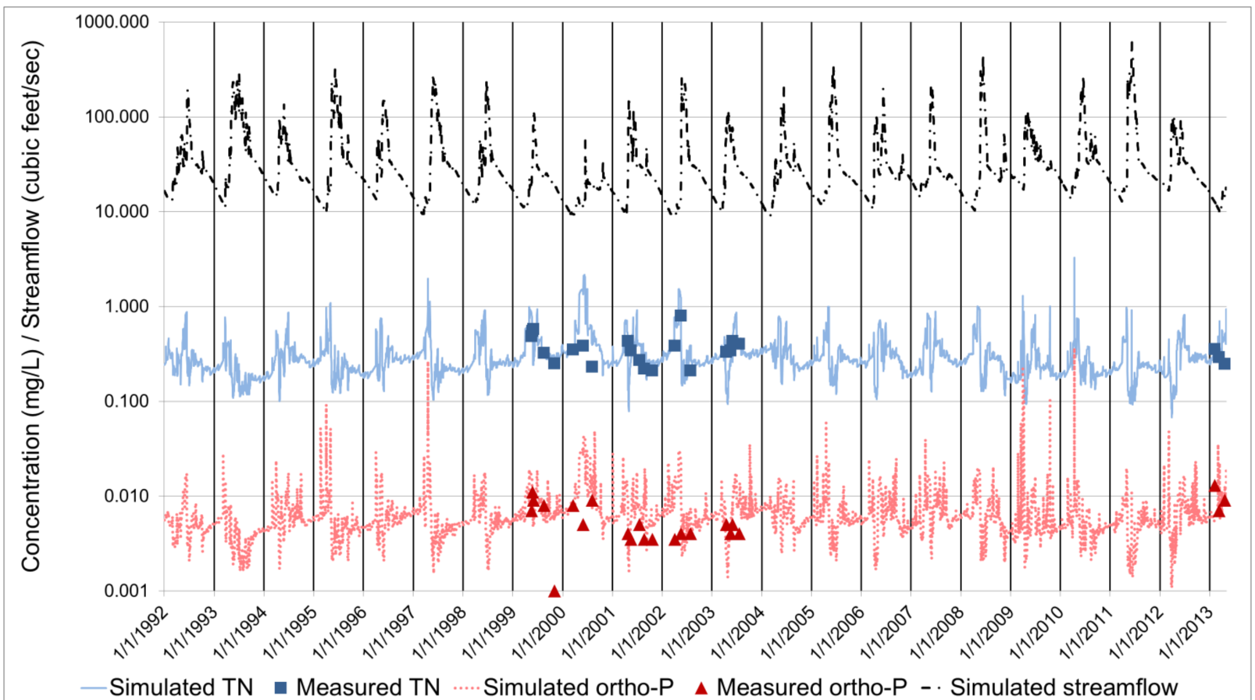

Figure 4 –Time series graph of simulated and observed concentrations for total nitrogen (TN) and ortho-phosphorus (ortho-P).

5.0 DISCUSSION

The model performance as compared to five sites where nitrate was measured in the groundwater indicate MEANSS is within 32 percent of field measured values, which is a good result for a screening tool particularly when accounting for the high degree of uncertainty in trying to measure and partition nitrate loads in groundwater from OWTS discharges.

Comparison with ArcNLET provides insight into how much variation can occur in estimating attenuation rates even when using a range of reasonable hydrogeologic parameters. When aquifer parameters and denitrification rates are well constrained, the ArcNLET program can likely provide more accurate values than MEANSS. But as the accuracy of site-specific aquifer parameters and denitrification rates decrease, MEANSS becomes a useful option.

Finding existing studies with adequate ortho-P data from OWTS proved to be a limiting factor in assessing MEANSS performance against existing data. Fortunately, the SWAT watershed model provided an adequate assessment method for ortho-P. The SWAT model also provided validation for the third nitrogen scoring category (soil within 100 feet of surface water), which was not possible using groundwater data in the five site-specific studies presented.

6.0 CONCLUSIONS

MEANSS was developed as an easy to use and cost-effective tool to estimate nitrogen and phosphorus loadings to surface waters from OWTS. Comparison to five sites, where nitrate loading from OWTS was measured in the groundwater, provided MEANSS estimates at a range of 68 to 120 percent of the measured loads. Comparison to a more quantitative GIS-based program (ArcNLET) shows the high degree of variability in estimating nitrate attenuation when aquifer properties are not well constrained. Using the SWAT watershed model on a small watershed provided good results when MEANSS was compared to observed instream concentrations for both TN and ortho-P.

Future work on MEANSS may include evaluation of how to better distribute loads seasonally to match the variation in groundwater flows that control the delivery of nutrients to surface water. Additional comparisons to measured data may be conducted as sites with adequate data become available.

Despite the good results presented, MEANSS only provides an estimate of highly complex and spatially variable processes that occur in the subsurface. For certain applications, the level of uncertainty inherent with MEANSS may be acceptable to the user. However, for other applications that require a higher degree of accuracy MEANSS may not be an appropriate tool.

7.0 REFERENCES

Anderson, Damann L. 1998. Natural Denitrification in Groundwater Impacted by Onsite Wasterwater Treatment Systems. In: Proceedings of the Eighth National Symposium on Individual and Small Community Sewage Systems. American Society of Agricultural Engineers (ASAE); 336-345.

Arnold, Jeffrey G., Peter M. Allen, and Gilbert Bernhardt. 1993. A Comprehensive Surface-Groundwater Flow Model. *Journal of Hydrology.* 142(1-4): 47-69.

Boer, Brian R. 2002. Septic-Derived Nutrient Loading to the Groundwater and Surface Water in Lolo, Montana. M.S. Missoula, MT: University of Montana.

Briar, David W. and DeAnn M. Dutton. 2000. Hydrogeology and Aquifer Sensitivity of the Bitterroot Valley, Ravalli County, Montana. Helena, MT: US Geological Survey. Water Resources Investigations Report 99-4219, DEQ Contract # 260102.

Driscoll, Fletcher G. 1986. Groundwater and Wells, 2nd ed., St. Paul, MN: Johnson Screens.

Dubrovsky, Neil M., Karen R. Burow, Gregory M. Clark, JoAnn M. Gronberg, Pixie A. Hamilton, Kerie J. Hitt, David K. Mueller, Mark D. Munn, Bernard T. Nolan, Larry J. Puckett, Michael G. Rupert, Terry M. Short, Norman E. Spahr, Lori A. Sprague, and William G. Wilber. 2010. Nutrients in the Nation's Streams and Groundwater, 1992 - 2004. Reston, VA: U.S. Geological Survey. Circular 1350.

Frans, L.M., 2000, Estimating the probability of elevated nitrate (NO2 + NO3 - N) concentrations in ground water in the Columbia Basin Ground Water Management Area, Washington: U.S. Geological Survey Water-Resources Investigations Report 00-4110, 24 p.

Gilliam, J. W. 1994. Riparian Wetlands and Water Quality. *Journal of Environmental Quality.* 23: 896-900.

Gold, Arthur J., Peter M. Groffman, Kelly Addy, Dorothy Q. Kellogg, Mark Stoldt, and Adam E. Rosenblatt. 2001. Landscape Attributes As Controls on Ground Water Nitrate Removal Capacity of Riparian Zones. *Journal of the American Water Resources Association.* 37(6): 1457-1464.

Gold, Arthur J. and J. Thomas Sims. 2000. Research Needs in Decentralized Wastewater Treatment and Management: A Risk-Based Approach to Nutrient Contamination. In: National Research Needs Conference Proceedings: Risk-Based Decision Making for Onsite Wastewater Treatment. United States Environmental Protection Agency, and National Decentralized Water Resources Capacity Development Project; May 19, 2000; St. Louis, Missouri. Palo Alto, CA: EPRI.

Harden, Stephen L. and Timothy B. Spruill. 2008. Factors Affecting Nitrate Delivery to Streams From Shallow Ground Water in North Carolina Coastal Plain. United States Geological Survey Scientific Investigations Report.

Heatwole, K. K. and John E. McCray. 2006. Modeling Potential Vadose-Zone Transport of Nitrogen From Onsite Wastewater Systems at the Development Scale. *Journal of Contaminant Hydrology.* 10

Howarth, Robert W., Barry Burnell, and Bruce Wicherski. 2002. Nutrient-Pathogen Evaluation Program for On-Site Wastewater Treatment Systems. Boise, Idaho: Idaho Department of Environmental Quality.

Hughes, B. F. and S. Pacenka. 1985. BURBS: A Simulation of the Nitrogen Impact of Residential Development on Groundwater. Ithica, New York: Center for Environmental Research, Cornell University.

Jacinthe, P., Peter M. Groffman, Arthur J. Gold, and A. Mosier. 1998. Patchiness in Microbial Nitrogen Transformations in Groundwater in a Riparian Forest. *Journal of Environmental Quality.* 27: 156-164.

Jeong, J., C. Santhi, Jeffrey G. Arnold, R Sirinivasan, S. Pradhan, and Kyle F. Flynn. 2011. Development of Algorithms for Modeling Onsite Wastewater Systems Within SWAT. *American Society of Agricultural and Biological Engineers.* 54(5): 1693-1704.

Kirkland, S. L. 2001. Coupling Site-Scale Fate and Transport With Watershed-Scale Modeling to Assess the Cumulative Effects of Nutrients From Decentralized Onsite Wastewater Systems.: Department of Geology and Geological Engineering: Colorado School of Mines.

Kobus, Helmut and Wolfgang Kinzelbach. 1989. Contaminant Transport in Groundwater: Proceedings of the International Symposium on Contaminant Transport in Groundwater, Stuttgart, 4-6 April 1989: AA Balkema.

Kroeger, Kevin D., Marci L. Cole, Joanna York, and Ivan Valiela. 2006. Nitrogen Loads to Estruaries From Waste Water Plumes: Modeling and Isotopic Approaches. *Ground Water.* 44(2): 188-200.

Lombardo, Pio. 2006. Phosphorus Geochemistry in Septic Tanks, Soil Absorption Systems, and Groundwater. Newton, MA: Lombardo Associates, Inc.

Long, T. 2014. Methodology to Predict Nitrogen Loading From On-Site Sewage Treatment Systems. In: 8th Northwest On-Site Wastewater Treatment Short Course. Seattle, Washington: University of Washington, Department of Civil Engineering.

Makepeace, Seth V. and Brian Mladenich. 1996. Contribution of Nearshore Nutrient Loads to Flathead Lake. [Place unknown]: CSKT Natural Resources Department. TMDL Grant #X99818401-0.

McCray, John E., S. L. Kirkland, Robert L. Siegrist, and Geoffrey D. Thyne. 2005. Model Parameters for Simulating Fate and Transport of On-Site Wastewater Nutrients. *Ground Water.* 43(4): 628-639.

McDowell, Will, Christine Brick, Matt. Clifton, Michelle Frode-Hutchins, John Harvala, and Karen Knudsen. 2005. Septic System Impact on Surface Waters: A Review for the Inland Northwest. Sandpoint, Idaho: Tri-State Water Quality Council.

Miller, Ross. 1991. A Numerical Flow Model of the Missoula Aquifer: Interpretation of Aquifer Properties and River Interaction. M.S. Missoula, Montana: University of Montana.

-----. 1996. Missoula Wastewater Facilities Plan Update: Appendix B: Bitterroot River Nitrogen Loading From On-Site Wastewater Disposal Systems. Missoula, MT: Land & Water Consulting, Inc.

Montana Department of Environmental Quality. 2009. How to Perform a Nondegradation Analysis for Subsurface Wastewater Treatment Systems (SWTS) Under the Subdivision Review Process. Helena, MT: Montana Department of Environmental Quality.

Morgan, David S. and R. Everett. 2005. Simulation-Optimization Methods for Management of Nitrate Loading to Groundwater From Decentralized Wastewater Treatment Systems. St. Louis, Missouri: National Decentralized Water Resources Capacity Development Project. WU-HT-03-07.

Morgan, David S., Stephen R. Hinkle, and Rodney J. Weick. 2007. Evaluation of Appraoches for Managing Nitrate Loading From On-Site Wastewater Systems Near LaPine, Oregon. U.S. Geological Survey. USGS Scientific Investigations Report 2007-5237.

Mueller, David K., Pixie A. Hamilton, Dennis R. Helsel, Kerie J. Hitt, and Barbara C. Ruddy. 1995. Nutrients in Ground Water and Surface Water of the United States: An Analysis of Data Through 1992. Denver, CO: U.S. Geological Survey ; Earth Science Information Center.

Natural Resources Conservation Service. 2007. National Engineering Handbook - Part 630 Hydrology: United States Department of Agriculture.

Nevada Department of Conservation & Natural Resources. 2008. Guidance Manual (1.0): On Site Sewage Disposal Systems for Commercial Applications.

Nolan, B.T., Hitt, K.J., and Ruddy, B.C., 2002, Probability of nitrate contamination of recently recharged groundwaters in the conterminous United States: Environmental Science & Technology, v. 36, no. 10, p. 2138–2145.

Parkin, Timothy B. 1987. Soil Microsites As a Source of Denitrificaiton Variability. *Soil Science Society of America Journal.* 51: 1194-1199.

Redfield, Alfred C. 1958. The Biological Control of Chemical Factors in the Environment. *American Scientist.* 46: 205-221.

Reneau, R. B., Jr., C. Hagedorn, and M. J. Degan. 1989. Fate and Transport of Biological and Inorganic Contaminants From On-Site Disposal of Domestic Wastewater. *Journal of Environmental Quality.* 18: 135-144.

Ricker, John A., Norman N. Hantzsche, Barry Hecht, and Howard Kolb. 1994. Area-Wide Wastewater Management for the San Lorenzo River Watershed, California. In: Proceedings of The Seventh National Symposium on Individual and Small Community Sewage Systems. American Society of Agricultural Engineers; 355-367.

Rios, J. F., M Ye, L. Wang, P. Z. Lee, H. Davis, and R. W. Hicks. 2012. ArcNLET:A GIS-Based Software to Simulate Groundwater Nitrate Load From Septic Systems to Surface Water Bodies. *Computers and Geosciences.* 52: 108-116.

Rivett, Michael O., Stephen R. Buss, Phillip Morgan, Jonathan W. N. Smith, and Chrystina Bemment. 2008. Nitrate Attenuation in Groundwater: A Review of Biogeochemical Controlling Processes. *Water Research.* 42(16): 4215-4232.

Robertson, William D., John A. Cherry, and Edward A. Sudicky. 1991. Ground-Water Contamination From Two Small Septic Systems on Sand Aquifers. *Ground Water.* 29(1): 82-92.

Robertson, William D., S. I. Schiff, and C. J. Ptacek. 1998. Review of Phosphate Mobility and Persistence in 10 Septic System Plumes. *Ground Water.* 36(6): 1000-1010.

Roeder, Eberhard. 2008. Revised Estimates of Nitrogen Loadings to Groundwater From Onsite Wastewater Systems in the Wekiva Study Area. Tallahassee, Florida: Florida Department of Health.

Rosen, Michael R., Christian Kopf, and Karen A. Thomas. 2006. Quantification of the Contribution of Nitrogen From Septic Tanks to Ground Water in Spanish Springs Valley, Nevada. U.S. Geological Survey. USGS Scientific Investigations Report 2006-5206.

Rosenblatt, Adam E., Arthur J. Gold, Mark H. Stoldt, Peter M. Groffman, and Dorothy Q. Kellogg. 2001. Identifying Riparian Sinks for Watershed Nitrate Using Soil Surveys. *Journal of Environmental Quality.* 30: 1596-1604.

Shaw, Byron, Peter Arnsten, and William VanRyswyk. 1993. Subdivision Impacts on Groundwater Quality. Stevens Point Wisconsin: University of Wisconsin.

Siegrist, Robert L., John E. McCray, J. Bagdol, S. Van Cuyk, Kathryn S. Lowe, P. J. Lemonds, H. Smith, R. Goldstein, C. Chen, L. Weintraub, and J. Rada. 2005. Quantifying Site-Scale Processes and Watershed-Scale Cumulative Effects of Decentralized Wastewater Systems. St. Louis, Missouri: National Decentralized Water Resources Capacity Development Project. WH-HT-00-27.

Soil Conservation Service. 1987. Soil Mechanics Level 1, Module 3, USDA Textural Classification Study Guide: United States Department of Agriculture.

Starr, Robert C. and Robert W. Gillham. 1993. Denitrification and Organic Carbon Availability in Two Aquifers. *Ground Water.* 31(6): 934-47.

Suplee, Michael W., Arun Varghese, and Joshua Cleland. 2007. Developing Nutrient Criteria for Streams: An Evaluation of the Frequency Distribution Method. *Journal of the American Water Resources Association.* 43(2): 453-472.

Tesoriero, Anthony J. and Frank D. Voss. 1997. Predicting the Probability of Elevated Nitrate Concentrations in the Puget Sound Basin: Implications for Aquifer Susceptibility and Vulnerability. *Ground Water.* 35(6): 1029-1039.

Toor, Gurpal S., Mary Lusk, and Tom Obreza. 2011. Onsite Sewage Treatment and Disposal Systems: Nitrogen. University of Florida - Institute of Food and Agricultural Sciences. SL348.

U.S. Environmental Protection Agency. 2002. Onsite Wastewater Treatment Systems Manual. Cincinnati, OH: U.S. Environmental Protection Agency, Office of Water. EPA/625/R-00/008.

-----. 2004. Water Quality Trading Assessment Handbook: Can Water Quality Trading Advance Your Watershed's Goals? Washington, DC: United States Environmental Protection Agency, Office of Water. EPA Contract 68-W-02-048.

Umari, Amjad M. J., Peter Martin, Roy A. Schroeder, Lowell F. W. Duell, Jr., and Ronald G. Fay. 1995. Potential for Ground-Water Contamination From Movement of Wastewater Through the Unsaturated Zone, Upper Mojave River Basin, California. U.S. Geological Survey. U.S. Geological Survey Water-Resources Investigation Report 91-4137.

Vought, Lena B. M., Jonas Dahl, Carsten L. Pedersen, and Jean O. Lacoursiere. 1994. Nutrient Retention in Riparian Ecotones. *AMBIO: A Journal of the Human Environment.* 23(6): 342-348.