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

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EXECUTIVE SUMMARY

Estimating the attenuation of nutrients (nitrogen and phosphorus) in groundwater from on-site wastewater treatment systems (OWTS) is difficult because of the costs and uncertainty associated with determining site-specific degradation rates, groundwater flow paths, and aquifer hydraulic properties. Some available methods allow users to specify natural degradation rates for nutrients from OWTS but provide little or no guidance on how to determine those rates. Other methods use a mechanistic approach with numerous variables and equations known to affect nutrient attenuation but are necessarily complex and difficult to use for assessing hundreds or thousands of septic systems as is often needed for regulatory applications. A simple spreadsheet analysis, Method for Estimating Attenuation of Nutrients from Septic Systems (MEANSS), has been developed to provide a relatively simple method with minimal data requirements. MEANSS is an empirical method designed to estimate the load of nutrients that will migrate to groundwater and surface water from OWTS sources. When evaluated against several field studies, a GIS-based nitrogen loading model, a mechanistic model, and a watershed model, MEANSS provided comparable estimates of nitrogen and ortho-phosphorus attenuation below and downgradient of OWTS.

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1.0 INTRODUCTION

In 1990, approximately 25% of the U.S. population used onsite wastewater systems (commonly known as septic systems) to dispose of household wastewater; 1990 was the last year this information was collected by the U.S. Census Bureau (2021). In Montana, usage was 37.5%. The nutrients discharged from onsite wastewater treatment systems (OWTS) can create both groundwater and surface water pollution that affects both human health and aquatic life (Dubrovsky et al., 2010). Determining the amount of those nutrients (nitrogen and phosphorus) discharged from OWTS that eventually impact groundwater and surface water is difficult for many reasons but primarily due to the complex site-specific reactions that control natural attenuation of both nitrogen and phosphorus. Existing methods to estimate that attenuation can be complex, require significant data collection, or use average values that don't account for site-specific conditions. Quantifying OWTS nutrient contributions to groundwater and surface water is important for many facets of environmental and human health protection.

The Montana Department of Environmental Quality (MDEQ) identified a lack of available methods to estimate nutrient (nitrogen and phosphorus) attenuation in the environment that didn't require extensive site-specific subsurface investigation. Due to the difficulty in estimating natural nutrient attenuation rates, the default MDEQ method assumes no attenuation from the point of discharge. That is an environmentally protective assumption but not accurate in most situations. To provide a more accurate estimation of environmental impacts, a relatively simple, yet accurate method was needed to bridge the gap between larger OWTS that are required to obtain a state-issued groundwater discharge permit with long-term monitoring and reporting requirements versus smaller systems that are not monitored once plans are reviewed and approved.

To facilitate implementation of state water quality standards, a quantitative tool to estimate nutrient loading from OWTS to surface waters was developed by MDEQ. The Method for Estimating Attenuation of Nutrients from Septic Systems (MEANSS) was created primarily for use in OWTS permitting, nutrient trading, and for Total Maximum Daily Load (TMDL) development. Siegrist et al. (2005) described the need for a watershed model that uses existing empirical data for tracking the fate of OWTS discharges: "Most research in OWS has been empirical in nature. Much knowledge has been accumulated. The time has come to advance empirical observations into theory of processes involved and incorporate them into a watershed model." MEANSS is not as detailed and accurate as the model provided by Siegrist et al. (2005), but it is an intermediate method that is more accurate than single reduction rates applied to all situations and allows estimation of site-specific nutrient fate with existing and easily accessible site-specific information. MEANSS provides an option to estimate site-specific nutrient fate when more complex methods are not technically or logistically feasible.

MEANSS is not applicable to OWTS discharges that are above typical OWTS influent wastewater nutrient concentrations due to the potential for reduced attenuation percentages at elevated concentrations. Reduced denitrification percentages were noted in streams at higher in-situ nitrogen concentrations (Mulholland et. al., 2008). Using approximate 90th percentile values from cumulative frequency distributions (Lowe, et al., 2007) of septic tank effluent from domestic sources (not including food, medical, or non-medical sources), typical upper limits for total nitrogen and total phosphorus are estimated here as 100 milligrams per liter (mg/L) and 20 mg/L, respectively. Non-medical sources of non-domestic wastewater included schools, day-care centers, gas stations, mobile home parks and hotels/motels (Lowe et al., 2007).

Because nitrogen attenuation in the environment via denitrification is temperature dependent, MEANSS was validated primarily to cold weather climates similar to or in Montana and is thus only applicable to similar cold weather climates.

2.0 METHODS

2.1 DEVELOPMENT

MEANSS was developed using three criteria: 1) easy to understand and use, to allow consistent application by regulatory programs and the regulated public OWTS; 2) utilizes accessible and existing information; and 3) utilizes site-specific information, which incorporates factors known to control natural attenuation of nutrients.

MEANSS uses several simplifying conditions: 1) steady-state conditions are applied which does not account for the lag time needed for the treated wastewater from an OWTS to migrate into the groundwater and eventually into surface water; 2) all treated wastewater is assumed to enter the user-specified receiving surface water. In situations where the wastewater partially or completely by-passes the surface water due to local hydrologic conditions where groundwater does not flow into the surface water, the user may adjust the model results to account for the site-specific groundwater/surface water interactions; and 3) groundwater dilution effects on nutrient concentrations are not accounted for because dilution does not alter the OWTS-related load of nutrients entering groundwater and surface water. Where necessary, separate dilution calculations can be used in conjunction with MEANSS to estimate groundwater or surface water concentrations.

Variations in OWTS treatment efficiency (e.g., unequal wastewater distribution in drainfields) and wastewater influent characteristics are difficult to explicitly address in any estimation of OWTS effluent characteristics because of the variation of typical household wastewater strength and OWTS construction practices. As an empirical model, MEANSS uses its validation to multiple sites with varied OWTS sources to support the estimated nutrient attenuation values use, and thus implicitly accounts for the variations in OWTS influent quality and treatment efficiencies.

Based on field studies, denitrification rates in the vadose and saturated zones vary over at least three to five orders of magnitude (Tucholke et al., 2007; and McCray et al., 2005); other studies show that denitrification rates can vary considerably even within similar environments

(Robertson, Cherry and Sudicky, 1991; Starr and Gillham, 1993). Phosphorus attenuation has been studied less than nitrogen, but also has a high degree of uncertainty (Lombardo, 2006; Lusk, Toor, and Obreza, 2011; Robertson, Schiff, and Ptacek, 1998). Because of this high degree of variability and because MEANSS does not use site-specific measured attenuation rates, it provides an approximation of nutrient loads in groundwater and surface water.

2.2 Factors Controlling Nutrient Attenuation

Although many variables control the natural attenuation of nitrogen and phosphorus discharged from OWTS, those variables that were found in the literature to control attenuation the most –soil type, soil calcium carbonate content, and distance to surface water were used in MEANSS. Minimizing the number of variables was essential to meet one of the three MEANSS development criteria, to keep it simple and easy to use with readily available data. For example, soil temperature is a key factor in the denitrification process that converts nitrate to nitrogen gas. However, temperature is not included as a variable because MEANSS was developed for use in Montana and the annual temperature range throughout much of the state (particularly those areas of high OWTS density) have a similar annual surface air temperature range of 35-45 ^{o}F (see **Figure 2-1**). The nitrogen attenuation values in MEANSS will need to be adjusted and validated for use in significantly different temperature ranges than Montana. The factors that affect the natural attenuation of nitrogen and phosphorus are described here to provide the basis for MEANSS.

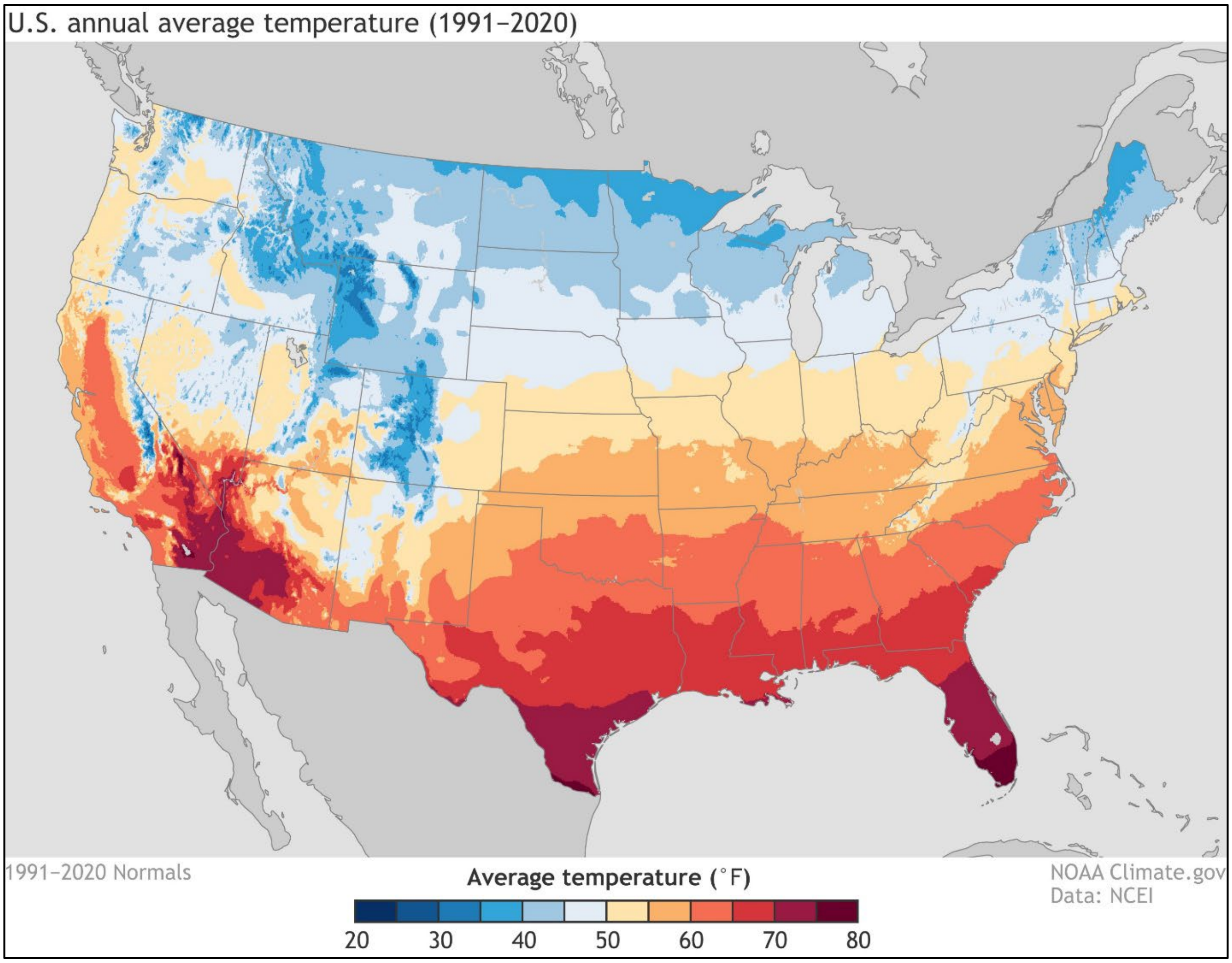

Figure 2-1. Average annual surface air temperatures in the contiguous United States (NOAA, 2021)

2.2.1 Nitrogen Attenuation Factors

Nitrogen in raw domestic wastewater that is discharged to the septic tank is primarily in the form of organic-nitrogen and ammonium-nitrogen, and quickly converts to mostly ammonium in the septic tank (Lowe, et. al., 2009). Disposal of untreated wastewater in a properly constructed and sized drainfield will typically provide sufficient oxygen and naturally occurring bacteria to convert the ammonium to nitrite and then quickly to nitrate. Conventional OWTS are not designed to complete the final step of nitrogen treatment, denitrification, which is the conversion of nitrate to nitrogen gas. Studies and water quality regulations commonly assume that most or all the ammonium in the raw wastewater is converted to nitrate after septic tank and drainfield (conventional) treatment (Geza, Lowe and McCray, 2013; Heatwole and McCray, 2006; Howarth, Burnell, Wicherski, 2002; MDEQ, 2015; Morgan and Everett, July 2005; Morgan. Hinkle and Weick, 2007; Toor, Lusk, and Obreza, 2011). As nitrate is the primary form of OWTS nitrogen that impacts groundwater and surface water, MEANSS is designed to estimate the denitrification rate after wastewater has been treated and discharged from the drainfield. Nitrogen gas dissipates into the atmosphere and does not have any further impacts to groundwater or surface water. While there may be some minor denitrification (approximately 10 percent) immediately below a properly operating drainfield (Costa et al., 2002; Lowe et. al., 2007; Rosen, Kropf, and Thomas, 2006), denitrification primarily occurs after the treated wastewater migrates away from the drainfield.

For denitrification to occur beyond the drainfield, a suitable environment must exist. The necessary factors include: 1) temperature near or above 50 ^oF; 2) an adequate carbon source which serves as food for the bacteria (available carbon is related to the soil organic matter content); 3) an anoxic environment; and 4) and the correct bacteria to utilize the oxygen atoms in the nitrate compound. A riparian zone with shallow groundwater is the most common natural environment that has those conditions (Brady, 1990; Gilliam, 1994; Gold and Sims, 2000; Harden and Spruill, 2008; McDowell et al., 2005; Rosenblatt et al., 2001), but other terrestrial and non-terrestrial environments with those conditions also occur (Seitzinger, 2006). 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 in other environments (Geza, Lowe and McCray, 2013; Gold and Sims, 2000; Jacinthe et al., 1998; Parkin, 1987; Umari et al., 1993). Temperatures above 50 ^oF provide faster denitrification rates, but the process still occurs at lower temperatures (Geza, Lowe and McCray, 2013; Harrison et. al., 2011; McCray et. al., 2010a, Pfenning and McMahon, 1996; and Dawson and Murphy, 1972). 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 organic carbon concentration below which denitrification does not occur, an adequate organic carbon source is cited as the most common limiting factor for denitrification (Starr and Gilham, 1993; Gold and Sims, 2000; Kinzelbach, Schaefer and Herzer, 1989; Rivett et al., 2008). Soil carbon consists of organic carbon (associated with living or once-living organisms) and inorganic carbon (associated with carbonate minerals). Wastewater denitrification is primarily associated with heterotrophic bacteria that utilize organic carbon over inorganic carbon; autotrophic bacteria can utilize inorganic carbon for denitrification but is not a significant component of wastewater denitrification (Rivet et al., 2008; Starr and Gillham, 1993). MEANSS accounts for the soil organic carbon using site-specific soil characteristics from the Natural Resources Conservation Service (NRCS) Soil Survey Geographic Database (SSURGO) database.

Because fine-grained soils are more likely to contain two of the conditions necessary for denitrification—anoxic conditions and organic carbon—they typically provide better denitrification conditions than coarse-grained soils (Tetra Tech, 2016; Geza, Lowe and McCray, 2013; Tucholke et al., 2007; Tesoriero and Voss, 1997; and Mueller et al., 1995). Anderson (1998) uses results from several studies to show a Pearson correlation coefficient (r) of 0.91 between denitrification rates and soil organic content. Otis (2007) uses soil type and drainage potential (divided into four drainage classes) as a factor in estimating denitrification of nitrogen discharged by OWTS; classifying soil types by drainage potential is similar to the method used in MEANSS. The hydrologic soil group (HSG) defined by the NRCS (2007) are divided into four groups (A, B, C, or D) and are used by NRCS to estimate stormwater runoff potential (the soil HSG is determined using SSURGO). The four HSGs are based on runoff potential of soil during maximum wetness conditions without accounting for effects of freeze conditions, vegetative cover, or slope (NRCS, 2007). The HSGs are also based on the saturated hydraulic conductivity of the least transmissive layer in the soil profile (NRCS, 2007). Although the NRCS uses additional criteria for the HSG designation, the amount of clay is 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; and group D soils have greater than 40 percent clay materials (NRCS, 2007). MEANSS uses HSG to account for soil-related anoxic conditions and the relative amount of organic carbon in soils; soil organic carbon content generally increases with clay and silt content (Brady, 1990, and Magdoff and Van Es, 2021). As described above adequate organic carbon is an important factor in denitrification potential. Clay soils also have lower permeability rates which provides two benefits for nutrient attenuation. First, drainfield size is increased in clay soils to account for the slower effluent percolation which increases the amount of soil organic carbon in contact with the effluent. Second, it slows the effluent migration rate, thereby allowing more time for denitrification to occur as compared to a more permeable soil type. MEANSS uses the HSG to estimate the relative amount of clay (and by association organic carbon) in the soil and correlates increased denitrification rates to higher soil organic carbon content.

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, Cole, York, and Valiela, 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. Those three parameters, even when measured using reliable methods, are only estimates of the bulk values across the entire migration distance and include inherent uncertainty. Additional uncertainty is added if travel time is combined with an estimated denitrification rate; McCray, et al. (2005) used published measured denitrification rates in natural soils, but not necessarily specific to OWTSs, to show the rate ranged over nearly three orders of magnitude; Tucholke et al., (2007) provided data indicating the rate ranges over five orders of magnitude. Although correlating nutrient reduction to distances in MEANSS also has its uncertainties, it simplifies data collection over those parameters described above which also have multiple sources of uncertainty.

Assuming that there is no shallow confining layer that limits the effluents vertical migration, increased distances from surface water also allows for deeper groundwater flow which increase the chances of encountering anoxic conditions that are conducive to denitrification (Dubrovsky, et al., 2010). Distance between the discharge location and receiving waterbody is cited as a factor in determining nutrient delivery credits for nutrient trading in National Pollutant Discharge Elimination System Permits (USEPA, June 2009). The use of travel time and measured denitrification rates in the groundwater were considered during the initial development of MEANSS. However, travel time was not used for three reasons. First, denitrification rates are site-specific and 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; McCray et al., 2005; Siegrist et al., 2005) that is based on the median of cumulative frequency distributions of field measured denitrification rates (0.025 day^{-1}), specific denitrification rates could not be correlated to soil type (McCray et al., 2005). The lack of correlation between soil type and definitive denitrification rates may be due to other factors such as available oxygen and temperature that also affect denitrification rates. Third, estimating travel time requires collecting site-specific data for hydraulic conductivity, hydraulic gradient, and porosity. Measuring those three parameters accurately can be logistically difficult, have limited accuracy (particularly for hydraulic conductivity and porosity), and expensive.

Review of the existing literature presented above provided three factors that are used in MEANSS to estimate denitrification: 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 nitrogen reduction values applied to these characteristics are provided in Section 2.3.

2.2.2 Phosphorus Attenuation Factors

Phosphorus, which has lower mobility than nitrogen, is removed from effluent recharge 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 draws the treated wastewater into the finer soil interstices and promotes phosphorus adsorption and precipitation (USEPA, 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 factor 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 (Lombardo, 2006; Lusk, Toor, and Obreza, 2011; and Robertson et al., 1998). Typically, non-calcareous soils are derived from igneous or metamorphic parent rocks. Lombardo (2006) defined calcareous soils 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. The negative relation between soil calcium carbonate and phosphorus adsorption is valid for soils with typical pH values. However, for some unique cases where soils have unusually high pH values (typically above 8 as occurs in small portions of northwestern Montana) phosphorus adsorption in calcareous soils is higher than in similar soils with more neutral pH (Buckman and Nyle, 1969). When there is sufficient site-specific data to demonstrate those high pH soil conditions exist, the user can adjust and increase the amount of attenuation in MEANSS for the calcareous soils (for example, increase the amount of attenuation for calcareous soils (>15% calcium carbonate) to the same attenuation for medium or low calcium carbonate attenuation values in **Table 2-2**). At typical soil pH levels or where the soil pH is unknown the attenuation values in **Table 2-2** should not be adjusted.

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 wastewater plumes with high phosphorus concentrations are less mobile than nitrogen, have been found to extend a relatively short distance from the source, and create high concentrations of phosphorus in soils immediately below drainfields with low levels beyond that location (Gold and Sims, 2000; Lombardo, 2006; Makepeace and Mladenich, 1996; Reneau, Hagedorn, and Degan, 1989; Robertson et al., 1998).

Riparian areas, where anaerobic conditions often exist and are conducive to denitrification, provide a poor environment for soil adsorption and precipitation of phosphorus (Vought, Dahl, Pedersen, and Lacoursiere, 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 soil HSG in the drainfield area; 2) predominant soil calcium carbonate content in the drainfield area; and 3) distance between the drainfield and the receiving surface water. The phosphorus reduction values applied to these characteristics are provided in Section 2.3.

2.3 Determining Parameters Used in MEANSS

The parameters used in MEANSS are available through the following sources: GIS mapping for distance values; the United States Geological Survey (USGS) National Hydrography Dataset (NHD or NHDPlus) to determine appropriate receiving surface waters; and the NRCS SSURGO soils database to determine the HSG and soil calcium carbonate content. GIS analysis tools can be used to determine: distance to surface water; soil characteristics at the drainfield; and soil HSG in the 100 foot riparian buffer. An example site is provided to demonstrate how nitrogen and phosphorus loadings are estimated based on the parameters discussed previously (see **Figure 2-2** and calculations in **Tables 2-1 and 2-2**).

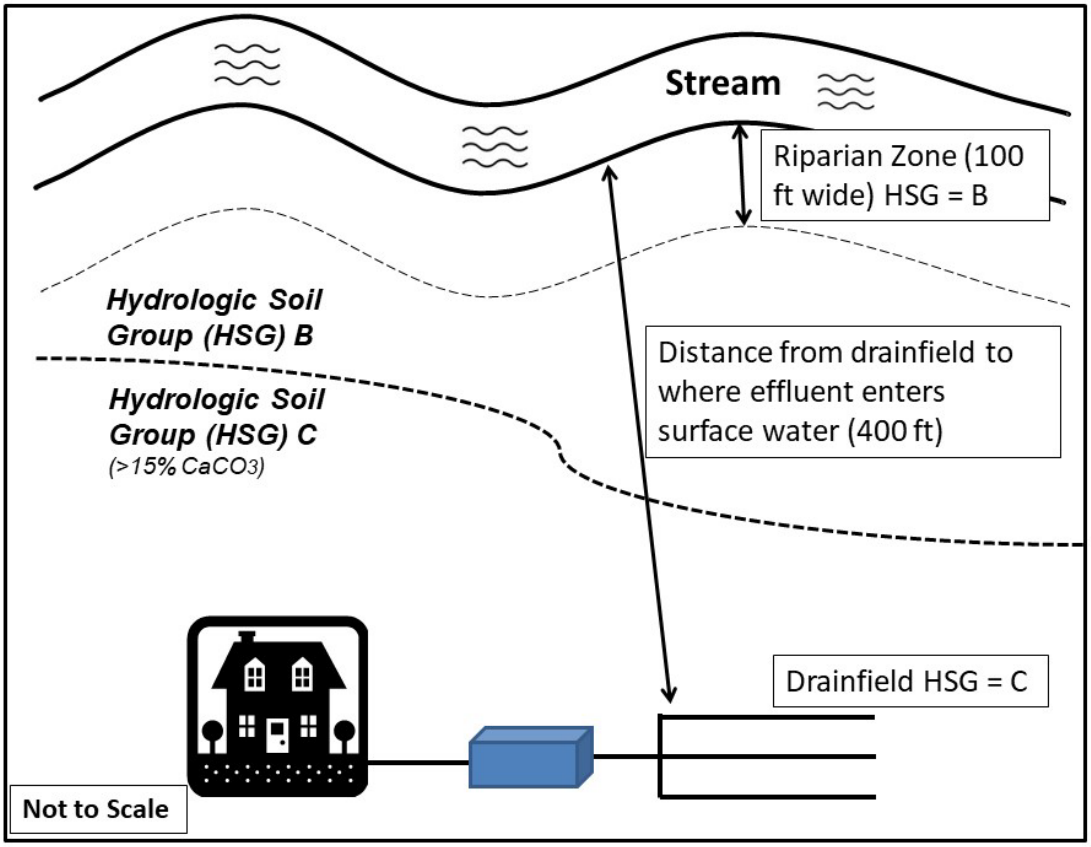

Figure 2-2. Plan view schematic of an example OWTS site showing MEANSS parameters, including hydrologic soil groups, distance to surface-water body, and riparian area.

The distance to nearest receiving surface water (where effluent enters surface water) is often the most uncertain parameter required in MEANSS (see **Figure 2-2**). 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 groundwater flow data is available the horizontal (i.e. map) distance along the flow path from OWTS to surface water is used for the distance value. When site-specific data are not available an option is to assume the shortest distance between the OWTS and the nearest downgradient perennial surface water (as classified in the NHD) for the distance value (see **Figure 2-2**); distance is based on the horizontal (i.e. map) distance between the OWTS and the estimated location where the effluent enters the surface water.

The HSG used in MEANSS is based on the SSURGO classification of the predominant soil type beneath 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 (see **Figure 2-2**). The SSURGO HSG classification is based on the soil layer with the lowest saturated hydraulic conductivity (or other defined soil characteristics when saturated hydraulic conductivity is unavailable) above an impermeable layer or groundwater in the soil profile (NRCS, 2007).

The soil calcium carbonate content used in MEANSS is also based on values in the 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 similar soils nearby that do have a calcium carbonate concentration in SSURGO; or using the medium calcium carbonate concentration (>1% - <15%) value in the phosphorus spreadsheet as a default value.

2.3.1. Nitrate Spreadsheet

The MEANSS spreadsheet for nitrate attenuation is presented in **Table 2-1**.

Table 2-1. Nitrate Attenuation Factors for OWTS Discharges to Soil

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

Notes:

*(1) The total nitrate reduction is the sum of the individual reductions for Category 1 + Category 2 + Category 3. For example (see **Figure 2-2**), a drainfield that is in a hydrologic soil group C soil (20 percent reduction) that drains to a surface water with hydrologic soil group B riparian soil (20 percent reduction) and is 400 feet from the surface water (10 percent reduction) would reduce its nitrate load to the surface water by 50 percent from the original load discharged from the drainfield.*

(2) SSURGO soil data is available via the NRCS web soil survey at: http://websoilsurvey.nrcs.usda.gov/app/HomePage.htm, or can be downloaded as a geodatabase: https://www.nrcs.usda.gov/resources/data-and-reports/soil-data-viewer.

(3) Distance to surface water is one possible endpoint for evaluation of nitrate reduction. Other endpoints within the groundwater flow path may also be used for scoring category 3 (for example, a regulatory groundwater compliance point for the OWTS discharge).

The reduction percentages in **Table 2-1** were initially chosen to maintain consistency to published values where that information is available as described previously in Section 2.2. The next step involved validating **Table 2-1** to site-specific studies with measured nitrogen attenuation rates related to OWTS discharges as discussed in the results (Section 3). Validating each of the three scoring categories separately in **Table 2-1** to site-specific measured data proved difficult because the data in existing studies often does not lend itself to specifically calibrating each category. For example, using nitrogen groundwater data does not distinguish what portion of the nitrogen attenuation occurs in the soil beneath the drainfield and what portion occurs during the travel time below the soils and in the groundwater. Similarly, surface water nitrogen concentrations do not distinguish attenuation percentages between the three categories in **Table 2-1**. However, several examples do provide an opportunity to validate individual reduction percentages in **Table 2-1**:

- For reductions at the drainfield based on the soil HSG, comparable comparisons include the Soil Treatment Unit Model (STUMOD) (McCray et al., 2010a) and the USGS lysimeter study in Nevada (Rosen et al., 2006); the details are described in the results (Section 3).
- For reductions in the riparian area based on the soil HSG a study of irrigation water denitrification entering the South Platte River is used. The study showed nitrogen reduction due to denitrification between 15-30 percent in groundwater between the source area (fertilized irrigated fields) and water entering the river (McMahon and Bohlke, 1996); for this calculation the dilution effects of river water in the hyporheic zone on the nitrate concentrations were separated from denitrification-related nitrogen reduction using nitrogen isotopes. SSURGO shows the HSG rating of the riparian soils in this region are roughly an equal combination of A and D soils. The estimated reductions using MEANSS (**Table 2-1**) for an area of mixed A soils (0% reduction) and D soils (50% reduction) in the riparian area is 25 percent, which is in the middle of range estimated by McMahon and Bohlke (1996), 15-30 percent. This comparison was not included in the validation results because it is based on nitrogen reduction from an agricultural source of nitrogen, not OWTS.

As an empirical model, the values in **Table 2-1** (and **Table 2-2**, discussed below) are not based on known rate equations for different soil type, temperature, water content, nitrogen concentration, dissolved oxygen concentration, or organic carbon content. Given that denitrification rates vary essentially from 0 to 100% in the environment (Seitzinger, 2006), providing uncertainty ranges for the reduction values provided in **Table 2-1** is not feasible; the empirically estimated values can be adjusted up or down by the user if site-specific and better data are available for any specific application. For any OWTS, a long-term tracer test of wastewater applied via a drainfield would likely provide the most applicable data to estimate attenuation, but logistically is not a feasible solution for the thousands of OWTS applications processed by MDEQ annually.

2.3.2 Phosphorus Spreadsheet

The MEANSS spreadsheet for phosphorus attenuation is presented in **Table 2-2**.

Table 2-2. Phosphorus Attenuation Factors for OWTS Discharges to Soil

	Scoring Category 1			***Scoring Category 2***
Percent Phosphorus Reduction(1)	**Hydrologic Soil Group at Drainfield(2) ($CaCO_3$ <= 1%)**	**Hydrologic Soil Group at Drainfield(2) ($CaCO_3$ >1% and <15%)**	**Hydrologic Soil Group at Drainfield(2) ($CaCO_3$ >=15%)**	**Distance to Surface Water (feet) (3)**
10	A	A	A	0 – 100
20			B	
40		B	C	
50				>100 – 500
60	B	C	D	
80	C	D		>500 – 5,000
100	D			>5,000

Notes:

*(1) The total phosphorus reduction is the sum of the individual reductions for Category 1 + Category 2. For example (see **Figure 2-2**) a drainfield that is in a hydrologic soil group C soil with greater than 15 percent $CaCO_3$ (40 percent reduction) and is 400 feet from the surface water (50 percent reduction) would reduce their phosphorus load to the surface water by 90 percent from the load that is discharged from the drainfield.*

(2) SSURGO soil data is available via the NRCS web soil survey at:
http://websoilsurvey.nrcs.usda.gov/app/HomePage.htm or can be downloaded as geodatabase
https://www.nrcs.usda.gov/resources/data-and-reports/soil-data-viewer

(3) Distance to surface water is one possible endpoint for evaluation of phosphorus reduction. Other endpoints within the groundwater flow path may also be used for scoring category 2 (for example, a regulatory groundwater compliance point for the OWTS discharge).

Similar to the nitrogen attenuation values in **Table 2-1**, the specific phosphorus attenuation reduction values in **Table 2-2** were initially based on trial and error. Phosphorus validation was more difficult than nitrogen due to the lack of available studies with measured phosphorus reductions. Only one example was used (a modeling study of a Montana watershed) to validate the phosphorus reductions. As a result, the uncertainty associated with the phosphorus attenuation estimates is much greater than the uncertainty associated with the nitrogen attenuation estimates.

3.0 RESULTS

MEANSS estimates of nutrient attenuation were validated by comparing it to (a) five site-specific groundwater OWTS nitrate studies, (b) two OWTS nitrate attenuation models and (c) a Soil and Water Assessment Tool (SWAT) watershed model. A lack of adequate existing phosphorus studies that were of sufficient quality to validate MEANSS limited the evaluation of MEANSS phosphorus attenuation estimates to the SWAT watershed model.

The validation sites are summarized in **Table 3-1** and located in **Figure 3-1**. For each site description, the wastewater source (e.g. domestic, commercial, etc.) is identified when that information is available. Non-domestic wastewater sources have a larger range of septic tank effluent concentrations which has the potential to create larger errors in the comparisons to MEANSS because the MEANSS estimates use average nutrient loads consistent with domestic wastewater sources.

Table 3-1. Validation Site Summary

Site ID	Location	No. of OWTS Included in Analysis
1	Lolo, MT	658
2	Missoula, MT	4,861
3	Spanish Springs Valley, NV	4
4	Jordan Acres, WI	26
5	Village Green, WI	45
ArcNLET	Missoula	4,861
STUMOD	N/A	N/A
SWAT	Prickly Pear Watershed, MT	1,010

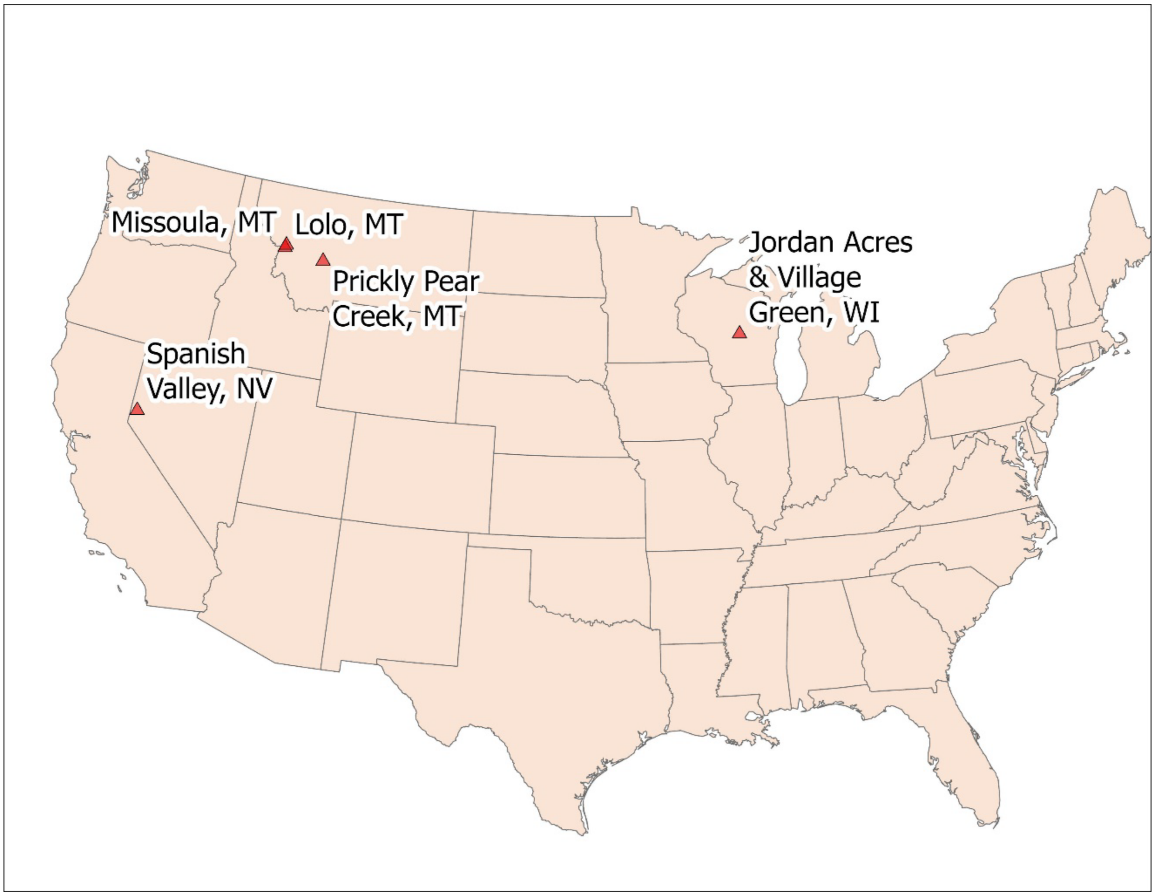

Figure 3-1. Location of MEANSS Validation Sites.

For evaluating MEANSS performance, the nitrate and ortho-phosphorus (ortho-P) loads discharged from a single family OWTS were estimated to be 13.8 and 2.92 kilograms per year (kg/yr), respectively, based on averages of published treated wastewater characteristics (USEPA, 2002; MDEQ, 2015).

The validation results are summarized in **Table 3-2** in Section 3.2.

3.1 Validation Sites

3.1.1 Site 1 (Lolo, Montana)

This study site (Boer, 2002) is a low-density residential area near Lolo, Montana (**Figure 3-1**). It covers 2.51 square miles (mi^2) and the study estimated OWTS loads equivalent to 658 single family homes. 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 dispersed livestock waste, domestic lawn fertilizer, and a 0.24 km^2 septage application site; these sources were calculated to be an order of magnitude less than the potential loading from OWTS at approximately 220 and 20 kg/yr of nitrate. Boer (2002) used average nitrate concentrations (50 mg/L) and an estimated average per household flow rate

(150 gpd) to characterize the OWTS-related loads to groundwater in the study area. The study results showed a small increase of the nitrate load in groundwater flowing out of the study area (7,008 kg/yr), which includes any background nitrate load, compared to the nitrate loading directly from OWTS (6,810 kg/yr). Using that mass balance information and chloride/nitrate ratios, Boer (2002) concluded that little or no denitrification was occurring in the study area. However, the initial OWTS loadings used by Boer (2002) are 75% of the average nitrate loading rate used by MDEQ, the MDEQ rates are more consistent with summaries of measured literature load rates (Lowe et. al., 2007) than those used by Boer (2002). If the MDEQ loading rates are used for the 658 OWTS in the Boer (2002) study it would provide an initial OWTS nitrate loading of 9,080 kg/yr, and a 23% nitrate attenuation rate compared to the measured groundwater loads (7,008 kg/yr) flowing out of the study area. The estimated nitrate reduction of 23% occurs despite the lack of local soil conditions more conducive to denitrification such as silty or poorly-drained soils (the study area consists primarily of well-drained loamy soils with predominantly A and B HSG classifications), which indicates even well drained soils can provide conditions favorable to a limited amount of denitrification.

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 at the time of the study in 2001, 558 homes. The analysis was run without using the second nitrate scoring category (soil type within 100 feet of surface water) because the groundwater data was primarily from wells not within the 100-foot riparian buffer. The cause of the difference between the number of homes in the 2008 database (558 homes) and the number of homes used in the study (658 homes) is unknown. The discrepancy is accounted for as described below.

MEANSS was used to calculate an average nitrate reduction in the study area of 41.5 percent based on the location of the 558 homes in the county septic database. If that percent reduction is applied to the 658 homes used by Boer (2002), the total OWTS-related nitrate load in groundwater flowing out of the study area is estimated at 5,312 kg/yr. The MEANSS load is 76 percent of the OWTS-related groundwater nitrate load measured by Boer (2002), 7,008 kg/yr.

3.1.2 Site 2 (Missoula, Montana)

A study to estimate nitrate loading to the Bitterroot River (**Figure 3-1**) from OWTS in and around the city of Missoula (Miller, 1996) is used for the second site. The study estimated the annual groundwater flux in the shallow aquifer based on a groundwater model as 144,432 cubic meters per day, which was about 10% lower than the annual median of the measured groundwater flux (158,592 cubic meters per day) (Miller, 1991; Miller, 1996), Average groundwater nitrate+nitrite-N concentrations in the study (1.2 mg/L) were based on eight groundwater wells and eleven groundwater seep samples collected in August 1995 (Miller, 1996). Only the groundwater seeps, which were close to the Bitterroot River, are used here to calculate the OWTS nitrogen load in groundwater. The Bitterroot River is at the downgradient end of the shallow aquifer, the groundwater seeps provide the best estimate of groundwater nitrate concentrations after migrating through most of the aquifer before discharging to the river and thus provided a more definable endpoint to evaluate nitrate reduction for scoring category 3 in **Table 2-1**. The average nitrate+nitrite-N concentration of the eleven seeps was

1.04 milligrams/liter (mg/L); for purposes of this analysis, the nitrate+nitrite-N 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 measured OWTS-related nitrate groundwater load near the river was 49,551 kg/yr.

The types of OWTS in the study area were not characterized (Miller, 1996). The study area was within the city of Missoula which is primarily residential wastewater but likely also has a significant component of non-residential sources (food, commercial, medical, schools, etc.) and therefore may have higher average nitrate concentrations than residential-only wastewater. The MEANSS analysis used the 2008 database provided by the Missoula Valley Water Quality District to extrapolate the number of OWTS that were contributing treated wastewater to the river in 1995, 4,315; accounting for multiple-family OWTS the equivalent number of single-family homes was 4,861. The analysis was run without using the second nitrate scoring category in **Table 2-1** (soil type within 100 feet of surface water) because the groundwater data was primarily from groundwater seeps not within the 100-foot riparian buffer.

MEANSS was used to calculate a nitrate reduction of 43.7 percent, which provides a groundwater nitrate load of 37,767 kg/yr from the 4,861 homes. The MEANSS load is 76 percent of the amount estimated by Miller (1996), 49,551 kg/yr. The error would be about 10 percent higher if the annual median measured ground water flow volume values were used, instead of the modeled flow volumes, to calculate the load.

3.1.3 Site 3 (Spanish Springs Valley, Nevada)

In Spanish Springs Valley, Nevada (**Figure 3-1**), 38 lysimeters were installed beneath four drainfields to measure the effluent characteristics during soil treatment (Rosen, Kropf and Thomas, 2006). Three of the drainfields serve single-family homes, the fourth serves a school. All of the drainfields had deep trenches (2 to 3 meters deep) with imported fill material due to low permeability soil near the surface. The lysimeters were installed in pairs, with one shallow lysimeter less than 1 meter deep within the trench fill material and one deep lysimeter immediately below the trench fill material (approximately 2-3 meters deep).

With deep trenches, the fill material essentially becomes the drainfield soil type. State regulations (Nevada Division of Environmental Protection, 2017) require clean sands and gravels with less than 5 percent fines for the material in drainfield trenches. However, local county regulations for sand lined trenches (Washoe County Health District, 2013) applicable to this site require an effective size (D_{10}) corresponding to fine or medium sands, with the remaining 10% consisting of fines. This type of fill soil was estimated as a B soil for this study comparison.

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 total nitrogen (TN) concentration was 44 mg/L (for purposes of this analysis it is assumed that the TN measured in the study is entirely nitrate). The deep lysimeter data was highly variable from 2.7 to 837 mg/L (the highest value was from a lysimeter located within a horse corral), therefore the study results used median data to minimize the effect of the outliers (Rosen, Kropf, and Thomas, 2006).

The study didn't measure the septic tank nitrogen concentrations at the four sites to determine influent concentration, but rather relied on published studies to estimate a range of typical septic tank effluent concentrations of 62 mg/L and 50 mg/L. Based on those values and the study results (44 mg/L TN in the deep lysimeters) the estimated percent TN reduction is 31% or 12%, respectively. The MEANSS analysis was run only using the first scoring criteria for nitrate reduction in **Table 2-1** (soil type at the drainfield) because the study measured the nitrate reduction beneath the drainfield and therefore could not be used to estimate attenuation along groundwater flow paths or attenuation near downgradient surface water.

MEANSS was used to estimate a 10 percent nitrate reduction based on B-type soils in the drainfield. The study estimated the influent TN concentration from published values as 50 and 62 mg/L (Rosen, Kropf and Thomas, 2006), corresponding to a percent reduction in the deep lysimeters of 12 and 31 percent, respectively. Using those values, the MEANSS estimate is either 83 or 32 percent, respectively, of the measured reductions.

3.1.4 Sites 4 and 5 (Jordan Acres and Village Green developments, Wisconsin)

Two separate subdivisions in Wisconsin (**Figure 3-1**) were monitored for nitrate impacts to groundwater (Shaw, 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 spreadsheet model, the authors estimated that approximately 20 percent of the nitrate load measured in the groundwater was from lawn fertilizer use (21 percent for Jordan Acres site and 18 percent for the Village Green site). The study used phosphorus and fluorescence in the multi-port monitoring wells to separate the groundwater being impacted from upgradient sources (deeper water) from groundwater impacted by the subdivisions. The study calculated low, medium, and high groundwater flow rates beneath the subdivisions to determine loading rates, and only used a portion of each subdivision in the loading analysis based on where downgradient multi-port wells could be installed to measure the groundwater nitrate concentrations. Using the medium flow rates in the study, and after accounting for the site-specific lawn fertilizer portion of the nitrate groundwater load, the calculated OWTS-related nitrate groundwater loads downgradient from the Jordan Acres (26 homes) and Village Green (45 homes) were 237 and 631 kg/yr, respectively.

MEANSS was run without using the second nitrate scoring category in **Table 2-1** (soil type within 100 feet of surface water) because the groundwater data was collected prior to entering any riparian zones.

MEANSS was used to estimate a 19.6 percent nitrate reduction at Jordan Acres (site 4) which is equal to a groundwater load of 289 kg/yr; the MEANSS estimated load is 122 percent of the measured load, 237 kg/yr. MEANSS was used to estimate a nitrate reduction of 18.4 percent at

Village Green (site 5) which is equal to a groundwater load of 508 kg/yr; the MEANSS estimated load is 80 percent of the measured load, 637 kg/yr.

A detailed analysis of the groundwater downgradient of a single OWTS (site REE) was conducted in the Jordan Acres site using wells with multiple sampling ports at different depths (Shaw, 1993). Using chloride as a conservative tracer, nitrate/chloride ratios indicate a 20% reduction in nitrate in the wells 38 meters downgradient of the OWTS compared to wells immediately downgradient of the OWTS. Although the difference between the upgradient and downgradient ratios was determined to not be statistically significant (Shaw, 1993), it still suggests a small amount of denitrification may have occurred that is similar to the average reduction predicted by MEANSS for the Jordan Acres site, 19.6 percent.

3.1.5 ArcNLET (Site #2, Missoula, Montana)

An analysis with the Arc Nitrate Loading Estimation Toolkit (ArcNLET) (Rios et. al., 2013) was used with the data from site #2 (Missoula, Montana). 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. ArcNLET would not run to completion with over 4,315 OWTS locations in this study area, therefore for purposes of this analysis ArcNLET was run with 10% of the actual OWTS (431). ArcNLET results were then extrapolated to the total OWTS in the study area.

The same OWTS spatial information used for the MEANSS analysis was also used for ArcNLET. For the ArcNLET analysis, the hydraulic conductivities and hydraulic gradient from Miller (1991) were used; the hydraulic conductivity included three zones with values of 610, 1,070 and 1,370 meters/day, the hydraulic gradient ranged from 0.001 to 0.003 meters/meter from east to west across the study area based on a 1986 groundwater flow map (Miller, 1991). A porosity of 25 percent was estimated using the upper end of the range for sand and gravel aquifers (Driscoll, 1986). The denitrification rate suggested in the ArcNLET documentation, 0.008 1/day, was used. Using those parameters, ArcNLET was used to estimate a total nitrate load to the Bitterroot River of 19,023 kg/yr.

The ArcNLET results are highly dependent on using accurate aquifer parameters and denitrification rates. For example, if the low end of published porosity values for the site soils were used instead of the high end (12 percent instead of 25 percent), the ArcNLET load estimate nearly doubles which would significantly increase the comparative error to MEANSS.

All the MEANSS nitrate scoring categories were used for this comparison of loads entering the Bitterroot River. The ArcNLET program estimated the nitrate load into the Bitterroot River as 19,023 kg/yr. The MEANSS estimate is 17,956 kg/yr, which is 94 percent of the ArcNLET calculated load.

The MEANSS estimated load in this comparison is less than the MEANSS estimated load for comparison to the Miller (1996) data for this same site (see site #2 results in Section 3.1.2) because the nitrogen load for this comparison includes all three of the MEANSS scoring categories (see **Table 2-1**), the MEANSS estimate for site #2 only included two of the scoring criteria.

3.1.6 Soil Treatment Unit Model (STUMOD)

MEANSS was compared to a mechanistic model, STUMOD, that calculates nitrogen reduction below the drainfield in the vadose zone (McCray et al., 2010a). To provide comparable results, only the soil type at the drainfield category in MEANSS (see **Table 2-1**) was used in the comparison. The twelve soil types included in McCray et al. (2010b) were compared, the HSG for each of those soil types was estimated for comparison purposes.

The nitrogen reduction values for STUMOD were estimated from cumulative probability graphs of Monte Carlo simulation results for a deep water table (McCray et al., 2010b). The STUMOD results were based on the following parameters:

- Hydraulic loading rate (i.e. drainfield application rate): 2 cm/day;
- Frigid/cryic temperature range (0-8 °C) – comparable to Montana temperature ranges (see **Figure 2-1**);
- Percent nitrogen reduction estimated at 120 cm soil depth;
- Standard drainfield effluent of 60 mg/L as ammonium-nitrogen and 1 mg/L as nitrate-nitrogen; and
- 50 percent value on the cumulative frequency distribution (i.e. half the simulations showed greater reductions and half the simulations showed lesser reductions of nitrogen).

The nitrogen reduction for twelve soil types using STUMOD and MEANSS is shown in **Figure 3-2**. The soil types in **Figure 3-2** are those used by McCray et al. (2010b). The HSG values in parenthesis (A, B, C and D) in **Figure 3-2** are approximations of HSG because the HSG classifications are not defined for the generic soil types used. MEANSS both under and over-estimates the nitrogen reduction compared to STUMOD. A 60 percent nitrate reduction for sandy clay soils using the STUMOD model (**Figure 3-2**) was 4 times higher than that for soils with the next highest reduction (15 percent for 3 other soils); the cause of the reduction rate spike is unknown and was not matched by the MEANSS reductions for sandy clay soils.

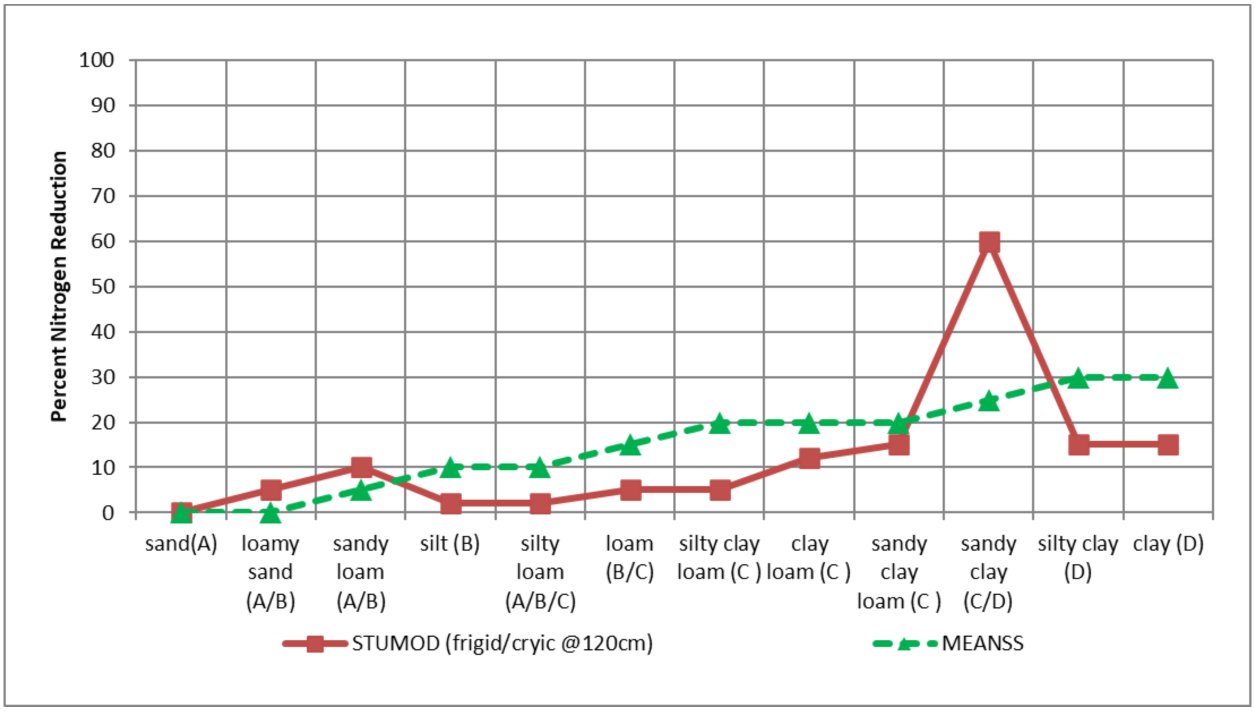

Figure 3-2. Comparison of STUMOD and MEANSS Nitrogen Reduction in Vadose Zone below the Drainfield.

3.1.7 SWAT Model (Prickly Pear Watershed, Montana)

The final validation method was a watershed model created using SWAT (Arnold, Allen, and Bernhardt, 1993). OWTS loading values from MEANSS were incorporated into the SWAT simulation and calibrated to observed instream concentrations of both TN and ortho-P. For this comparison the SWAT OWTS biozone algorithm, which is designed to simulate nitrogen, phosphorus, bacteria, and biological oxygen demand discharges from septic tank effluent (Jeong et al., 2011), was not used and replaced with the MEANSS estimates.

The Prickly Pear watershed in central Montana (**Figure 3-3**) was chosen for this project because it has a sufficient number of OWTSs (1,010) for the size of the watershed (531 km^2) to create noticeable impacts to stream water 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 or overwhelm the impacts from OWTS. The watershed is primarily a residential area, therefore the effluent for nearly all the OWTS in the study area is likely residential strength.

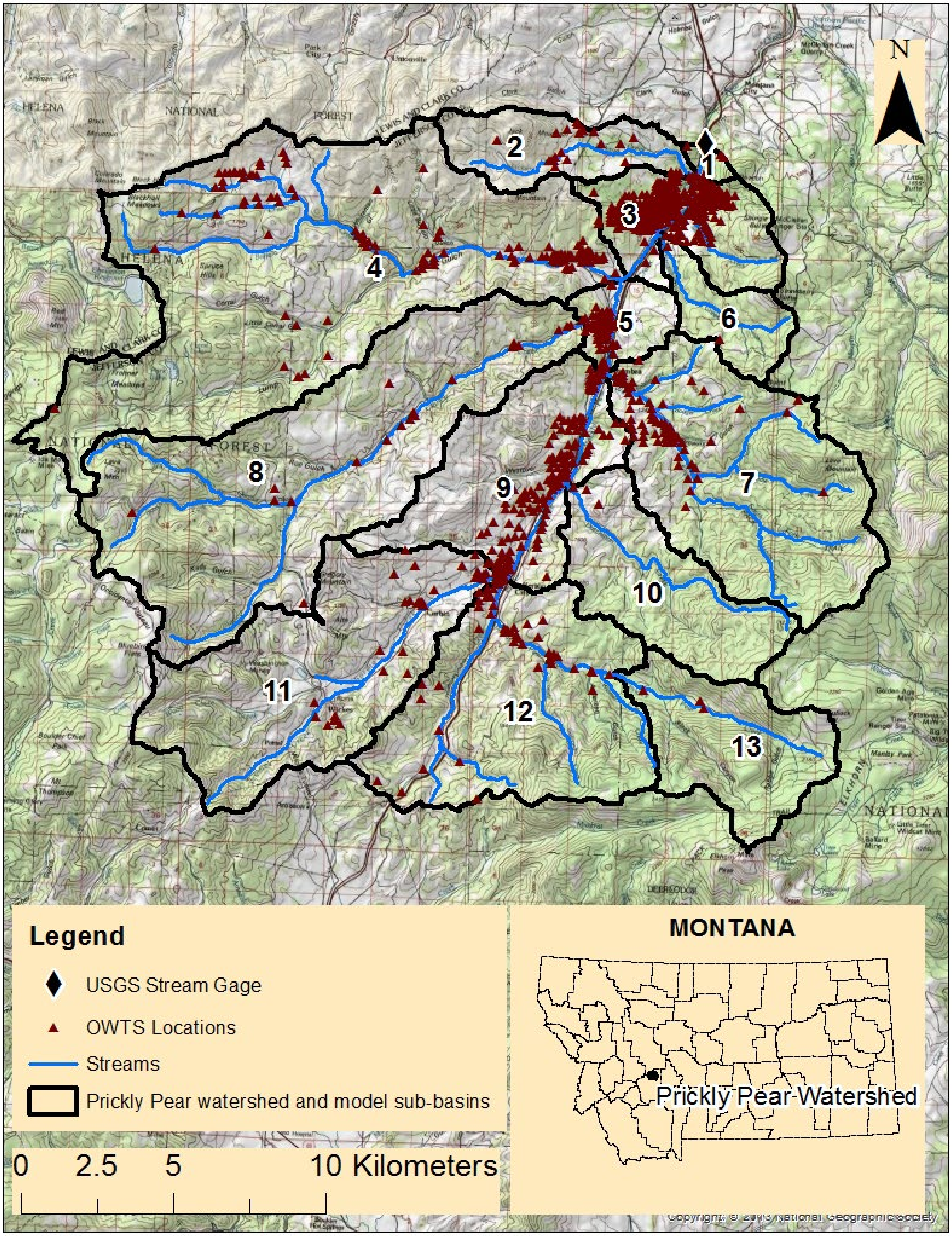

Figure 3-3. Prickly Pear watershed used for SWAT model.

The SWAT model was developed using available information for elevation, land use/landcover, soils and streamflow. The SWAT model daily streamflow predictions were compared and 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 streamflow calibration period was 1992 through April 2013. Daily measured streamflow was available for 82 percent of the calibration period. The error statistics between the simulated and measured daily streamflow values 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 daily streamflow values.

The instream water quality calibration data consisted of 20 TN and ortho-P 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 instream concentrations while instream 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.

Incorporating the steady-state MEANSS loading estimates into the daily time step SWAT model showed that the lack of seasonal variation in MEANSS results created unreasonably large TN and ortho-P values in the winter months during baseflow conditions. To provide better seasonal variation for the MEANSS results, it was assumed that OWTS TN and ortho-P contributions to streams varied proportionally with streamflow. This assumes a higher volume of groundwater contribution (and corresponding OWTS effluent contributions) to streams during the spring when groundwater elevations and velocities are higher than during other months of the year when groundwater elevations recede. The annual loads estimated using MEANSS were proportionally divided on a monthly basis (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. Adjusting the MEANSS results on a monthly basis produced better validation results. However, because MEANSS is not designed as a transient estimator of nutrient attenuation, it is more accurate when used to estimate steady-state or annual reductions.

Adjusting the MEANSS results to account for monthly variation resulted in good calibration to the measured data for both TN and ortho-P as shown in **Figures 3-4** and **3-5**. Statistical summaries of the simulated and observed concentration data are based only on the 23 sample dates (19 dates for the load values) and are provided in **Figure 3-4**. The simulated median TN and TP loads to Prickly Pear Creek using SWAT and MEANSS on those 19 dates were 37 and 38 percent larger than the measured loads, respectively. Time series plots of the observed versus simulated results are presented in **Figure 3-5**. **Figure 3-5** includes one anomalously low ortho-P (as P) measured sample (0.001 mg/L) on 11/5/1999, which is not matched by the simulated results. The cause of the low measured ortho-P result is unknown, total phosphorus or TN results for the same date (11/5/1999) were not anomalously low compared to other sample dates.

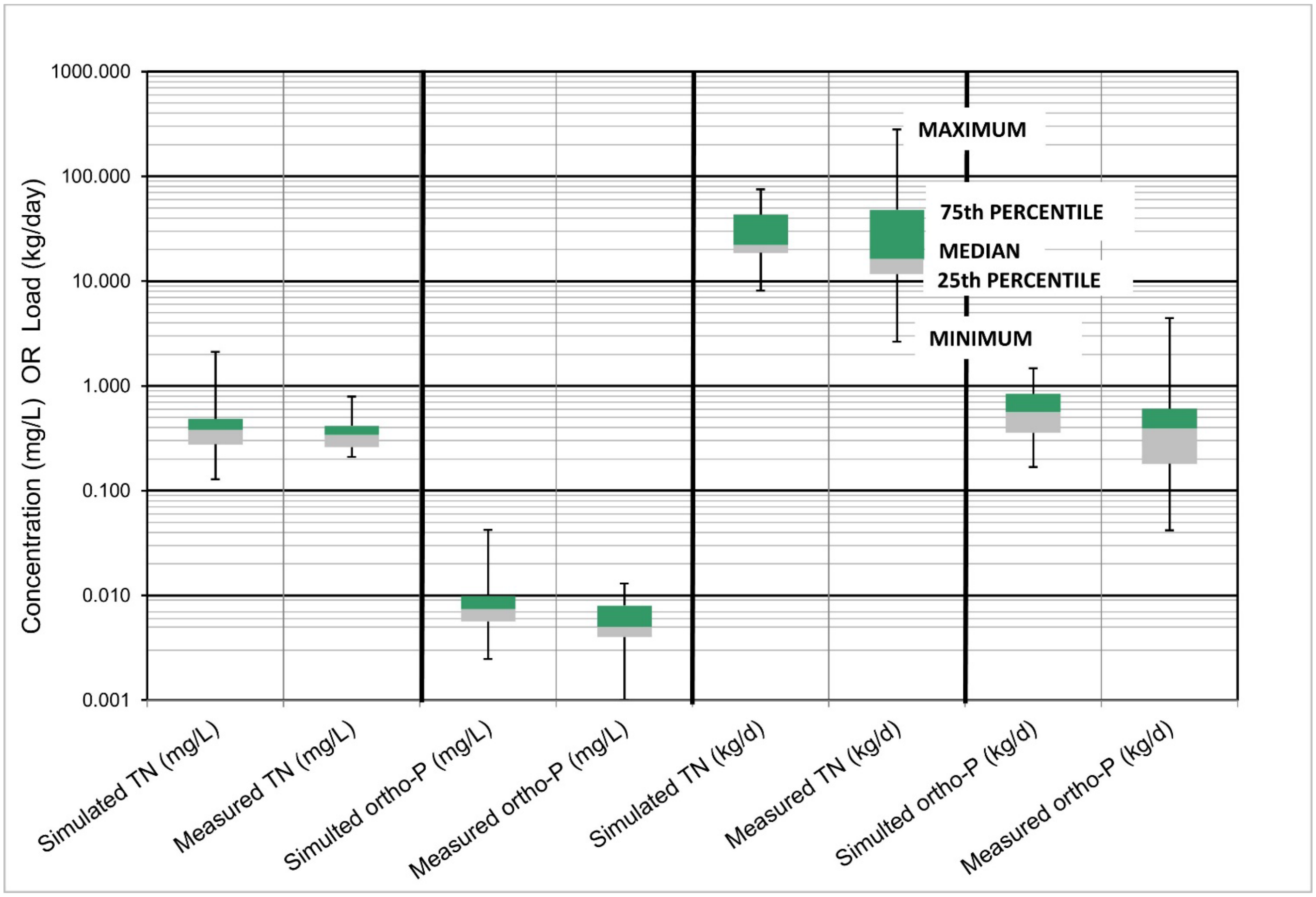

Figure 3-4. SWAT model calibration results for Total Nitrogen (TN) and Ortho-Phosphorus (ortho-P) using MEANSS to estimate TN and ortho-P OWTS loadings from OWTS. Based on 23 dates of observed instream concentrations and 19 dates of observed instream loads in Prickly Pear Creek.

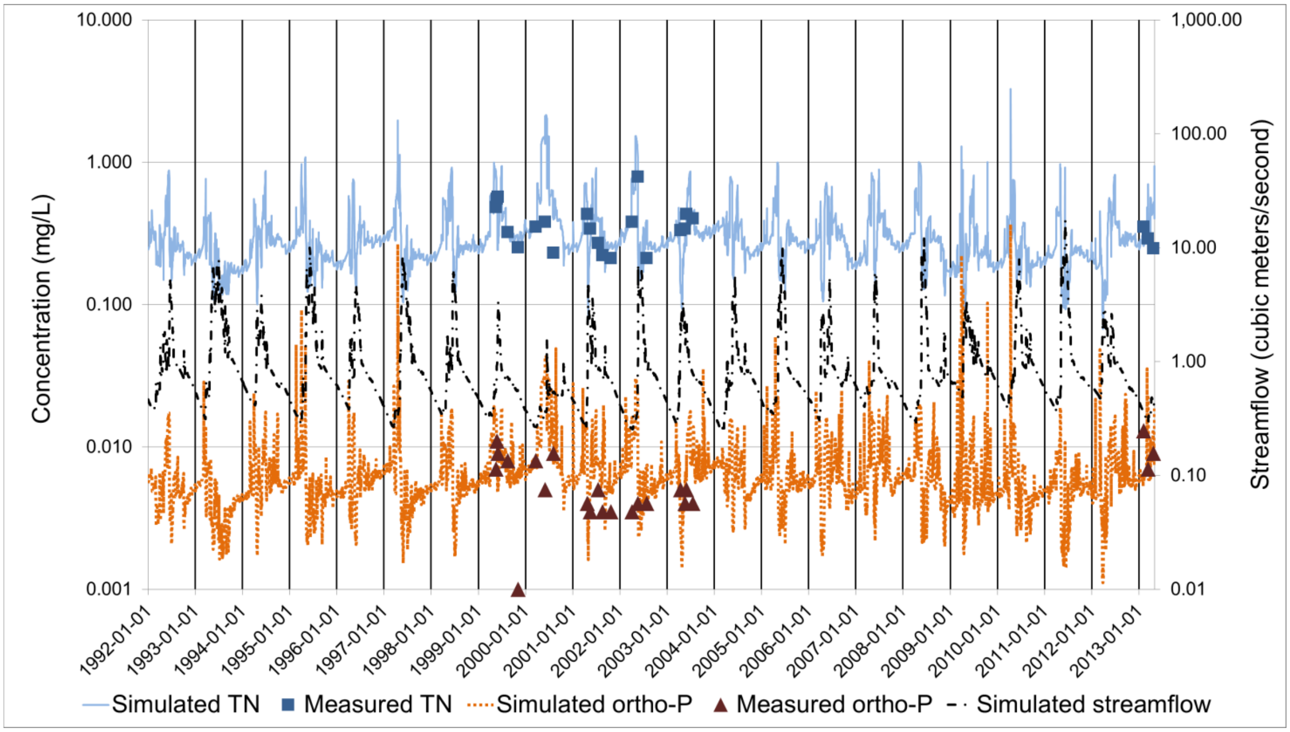

Figure 3-5. Time series graph of SWAT simulated and observed concentrations for total nitrogen (TN) and ortho-phosphorus (ortho-P) using MEANSS to estimate TN and ortho-P OWTS loadings to Prickly Pear Creek.

3.2 Results Summary

In **Table 3-2** the validation results are presented as the percent difference between the validation site load and the load estimated by MEANSS was calculated as:
(OWTS $LOAD_{MEANSS}$ – OWTS $LOAD_{CASE\ STUDY}$) ÷ (OWTS $LOAD_{CASE\ STUDY)}$) x 100.
The absolute average percent difference between MEANSS estimated loads and the loads estimated via other methods (not including STUMOD results from **Figure 3-2**) in **Table 3-2** is 28%.

Table 3-2. Validation Results

Site	Parameter	OWTS Load (or Percent Reduction) Estimated from Site Information	OWTS Load (or Percent Reduction) Estimated from MEANSS	% Difference
1	Total Nitrogen	7,008 kg/yr	5,312 kg/y	-24
2	Total Nitrogen	49,551 kg/yr	37,767 kg/yr	-24
3	Total Nitrogen	12% / 31% reduction	10% reduction	-17 / -68
4	Total Nitrogen	237 kg/yr	289 kg/yr	22
5	Total Nitrogen	637 kg/yr	508 kg/yr	-20
ArcNLET	Total Nitrogen	19,023 kg/yr	17,956 kg/yr,	-6
STUMOD	Total Nitrogen	Variable	Variable	See **Figure 3-2**
SWAT	Total Nitrogen	16.3 kg/d	22.4 kg/d	37
SWAT	Ortho-P	0.39 kg/d	0.54 kg/d	38

4.0 UNCERTAINTY

The uncertainty associated with MEANSS estimates is largely associated with the unknown environmental parameters that control nutrient attenuation, which include soil type, soil temperature, nitrogen concentration, soil dissolved oxygen concentration, soil organic carbon content, soil moisture content and soil $CaCO_3$ concentration. Those parameters and conditions largely control the environmental attenuation of both nitrogen and phosphorus. The uncertainties are reduced through the validation and comparison to measured attenuation values. Further reductions of uncertainty can be accomplished by continuing to validate MEANSS against measured data in various environments.

Additional uncertainty is associated with soil characteristics based on the resolution of the SSURGO database. Some of that uncertainty is reduced because soil-test pits are required as part of the MDEQ review at most OWTS sites; those test pits can be used to ground-truth the soil types indicated by SSURGO and adjusted to the correct soil map unit with the correct HSG and $CaCO_3$ percentages.

Another source of uncertainty is the nutrient effluent quality of OWTS, as those concentrations can influence the estimate of nutrient effluent loadings from each OWTS and influence the nutrient reduction rate. Published OWTS effluent ranges based on literature search (Lowe et. al., 2007) shows a narrower range of septic tank effluent nutrient concentrations when only domestic wastewater is considered (and thus less uncertainty) compared to the ranges when commercial, medical, food and industrial sources are included. Validation of MEANSS was based primarily on comparisons to domestic source OWTS and will have less uncertainty when used on domestic wastewater sources as compared to the other wastewater sources listed that have a higher potential for non-typical effluent concentrations. Therefore, it is recommended that MEANSS only be used for residential strength wastewater where the septic tank TN and TP effluent are measured or anticipated to be equal or less than 100 mg/L and 20 mg/L, respectively.

5.0 DISCUSSION

The model performance as compared to five sites where nitrate was measured in the groundwater indicate MEANSS is within 24 percent or less of field measured values, except for the much higher 68 percent error associated with one of the two reduction estimates for site #4. Overall, the results indicate a reasonable match to measured reductions particularly when accounting for the high degree of uncertainty associated with measuring and partitioning nitrate loads in groundwater from OWTS discharges in each of the validation studies.

Comparison to ArcNLET provided nearly identical results, but also provides insight into how much variation can occur in estimating attenuation rates even when using a range of reasonable hydrogeologic parameters. For example, if the low end of published porosity values for the site soils were used instead of the high end, (12 percent instead of 25 percent) the ArcNLET program load estimate nearly doubles. When aquifer parameters and denitrification rates are well constrained, the ArcNLET program can provide more accurate values than MEANSS. But as the accuracy of site-specific aquifer parameters and denitrification rates decrease, MEANSS becomes a useful option.

Comparing vadose zone reductions in MEANSS to a mechanistic model (STUMOD) showed large percent differences in denitrification for some soils, however the percent denitrification rates were generally low (less than 15%) which amplified small differences in the two models when those differences were compared in terms of relative percent differences. Some of the errors may be caused by the estimated HSG categories used in MEANSS based on the STUMOD soil descriptions.

Finding existing studies with adequate ortho-P data from OWTS proved to be a limiting factor in validating the MEANSS performance for ortho-P. Fortunately, the SWAT watershed model provided an adequate assessment method for ortho-P. The results showed a reasonable match between MEANSS and measured ortho-P loads, the MEANSS estimate was 38% larger. The nitrate comparison was similar, the MEANSS estimate was 37% larger.

6.0 CONCLUSIONS

MEANSS was developed as an easy to use and cost-effective tool to estimate nitrogen and phosphorus loadings to groundwater and surface waters from OWTS. The results show that MEANSS generally provides good estimates of nutrient loading to groundwater and surface water when compared to several studies and to other methods for estimating nutrient loads from OWTS. The data requirements for MEANSS are minimal compared to other methods and the parameters are readily available using existing soil databases and GIS tools.

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 would provide further validation and may be conducted as sites with adequate data become available.

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