APPENDIX F – SUBSURFACE WASTEWATER TREATMENT SYSTEMS IN THE FLINT CREEK WATERSHED

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

ACRONYMS

Acronym	Definition
DEQ	Department of Environmental Quality (Montana)
EPA	Environmental Protection Agency (U.S.)
GIS	Geographic Information System
MGWPCS	Montana Ground Water Pollution Control System
NRCS	Natural Resources Conservation Service
SWAT	Soil & Water Assessment Tool
TMDL	Total Maximum Daily Load
TP	Total Phosphorus
TPA	TMDL Planning Area
WWTP	Wastewater Treatment Plant

F1.0 INTRODUCTION

This is a technical report prepared to support the nutrient Total Maximum Daily Load (TMDL) process in the Flint Creek Watershed. This report will describe the current septic use locations and associated groundwater information. Information from this report will be used to construct a water quality model which will be used for TMDL source assessment and creating TMDL allocations. The water quality model (Soil & Water Assessment Tool (SWAT)) will assess the potential significance of nutrient loading from all sources within the watershed.

Wastewater is any water that has been adversely affected in quality by anthropogenic influence. It comprises liquid waste discharged by domestic residences, commercial properties, industry, and/or agriculture and can encompass a wide range of potential contaminants and concentrations. In the most common usage, it refers to the municipal wastewater that contains a broad spectrum of contaminants resulting from the mixing of wastewaters from different sources including household, industrial and commercial sources connected to a drainage system and routed to a treatment facility. Wastewater produced from septic systems, both on-site household treatment systems and larger multi-home systems, is the primary focus of this report. Wastewater sources that discharge to surface water are reviewed in a separate report.

F2.0 REGULATORY OVERVIEW

The Montana Department of Environment Quality (DEQ) regulates most subsurface wastewater treatment systems (septic systems) that are installed in Montana. Septic systems that are designed for less than 5,000 gallons per day and are not public systems are reviewed by the Department's subdivision section (if such as system is on a lot that is 20 acres or larger the system is only reviewed by the county). Septic systems that are designed for 5,000 gallons per day or greater are reviewed by the Department's wastewater discharge permit section along with any industrial facilities which discharge to groundwater. The systems reviewed by the wastewater discharge permit section are required to have a valid discharge permit to construct and operate the system (note that systems reviewed and approved by DEQ prior to May 1, 1998 do not need a permit until the system is modified or violates rule or statute). Public septic systems (schools, government buildings, etc.) designed for less than 5,000 gallons per day don't need a discharge permit, but are reviewed by the Department's Public Water Supply Section in a similar manner as those reviewed by DEQ's subdivision section.

All septic system reviews conducted by the Department have three major components: plan and specification review; site suitability; and water quality impact review. The plan and specification review is conducted to insure the design of the system meets the applicable technical standards in the Department's technical design circulars DEQ-2 and DEQ-4. The site suitability review is conducted to insure that the area chosen to dispose of the wastewater (typically through a drainfield or rapid infiltration bed) is suitable to hydraulically dissipate and treat the quantity and quality of the wastewater discharged. The site suitability is primarily based on the local hydrogeology and the soil type. The water quality impact review is to insure that the state waters that will be impacted by the wastewater discharge are not impacted beyond the allowable levels for nitrogen, phosphorus and bacteria.

The water quality impact review for each submitted project is typically completed in accordance with the state nondegradation rules, which applies to any new or increased source of pollutants to high quality state waters since April 29, 1993. Any source that is not subject to those nondegradation rules must comply with the water quality standards.

Under the nondegradation rules nitrogen impacts from septic systems may not cause the nitrate (as N) concentration to exceed 5.0 or 7.5 mg/L at the end of an approved groundwater mixing zone. The 5.0 mg/L limit applies to conventional septic systems. The 7.5 mg/L limit applies to level 2 septic systems (or to any septic system if the background nitrate in the groundwater is between 5 and 7.5 mg/L and that elevated nitrate concentration is primarily from sources other than human waste). When the discharge is close enough to a high-quality state surface water, the nitrate impacts to the surface water may be assessed with respect to the Circular DEQ-7 trigger value for nitrate, 0.01 mg/L. Under the trigger value criterion, the source may not increase the nitrate concentration in the surface water above the trigger value at the end of a surface water mixing zone. Alternatively, for surface water impacts, the source may be reviewed for compliance under the nondegradation narrative standard which requires that the source "… will not have a measurable effect on any existing or anticipated use or cause measurable changes in the aquatic life or ecological integrity." This review does not usually assess cumulative sources over time, only project by project impacts.

Under the nondegradation rules phosphorus discharges from septic systems must be adsorbed in the soils for at least 50 years before discharging to any high quality state surface water. When the discharge cannot meet the 50-year breakthrough criterion, the phosphorus impacts to the surface water may be assessed with respect to the Circular DEQ-7 trigger value for inorganic phosphorus, 0.001 mg/L. Under the trigger value criterion, the source may not increase the inorganic phosphorus concentration in the surface water above the trigger value at the end of a surface water mixing zone. Alternatively, the source may be reviewed for compliance under the nondegradation narrative standard which requires that the new source "… will not have a measurable effect on any existing or anticipated use or cause measurable changes in the aquatic life or ecological integrity." This review does not usually assess cumulative sources over time, only project by project impacts.

F3.0 Groundwater Waste Source Characterization

Sources of wastewater in the Flint Creek Watershed include municipal, industrial, public facility, multi-family and individual household wastewater treatment systems.

F3.1 Industrial Wastewater Treatment Systems

There are two industrial Montana Ground Water Pollution Control System (MGWPCS) groundwater discharge permits located in the watershed for the Sugar Loaf Wool Carding Mill located near Hall and for the Contact Mining Company located southeast of Phillipsburg (**Figure F-1**). Industrial groundwater discharges are required to obtain a MGWPCS permit regardless of the discharge volume.

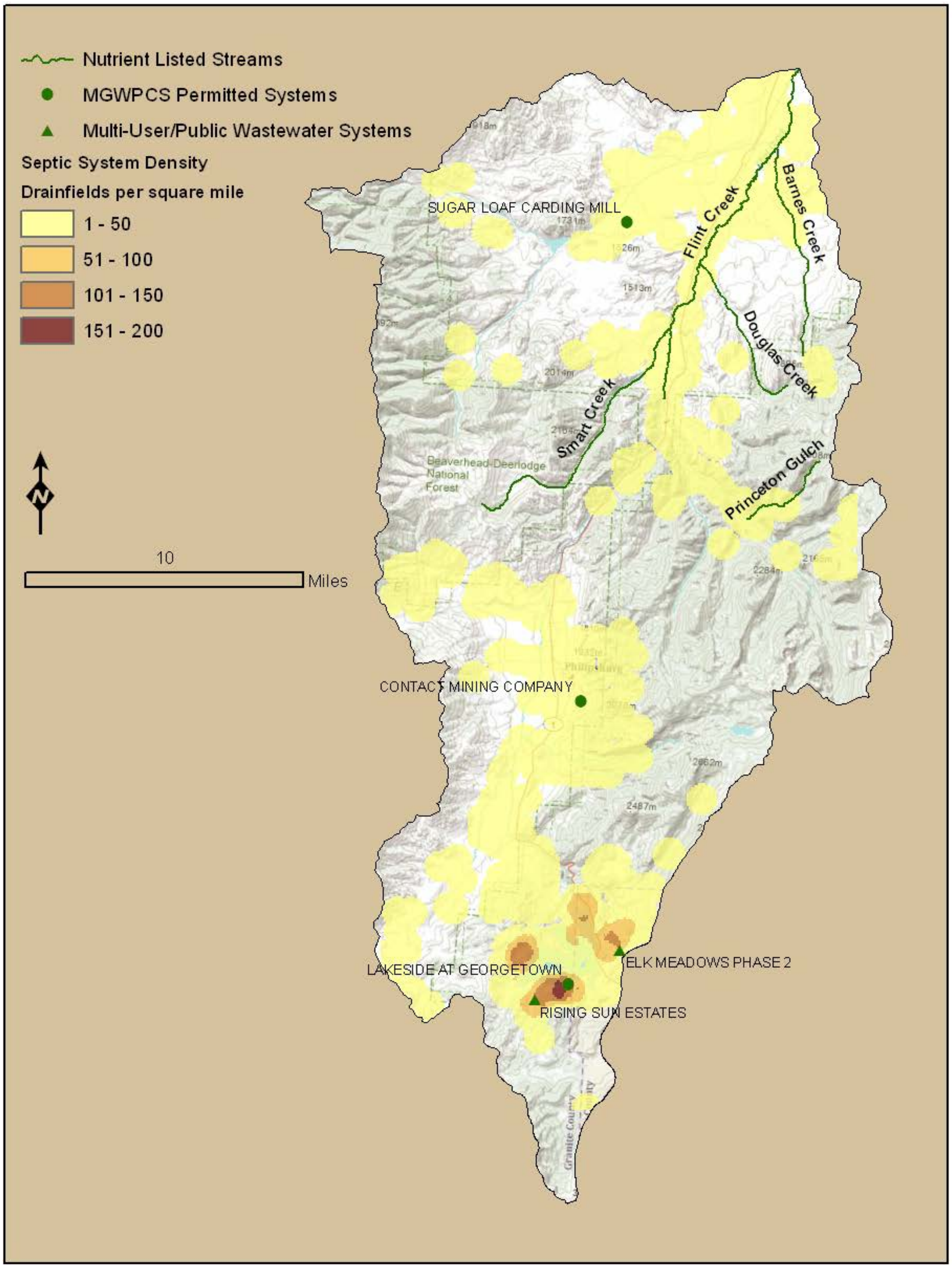

Figure F-1. Wastewater Contributions to Groundwater

F3.2 Municipal Wastewater Treatment Systems

The town of Phillipsburg is the only municipal sewered system that discharges wastewater within the TMDL Planning Area (TPA). The Phillipsburg system is discussed in the wastewater treatment plant (WWTP) point sources technical report for this project. The Philipsburg wastewater treatment system consists of a 2-celled facultative lagoon that discharges directly to Flint Creek and is therefore required to have an active Montana Pollutant Discharge Elimination System permit.

F3.3 Permitted, Public, and Multi-User Wastewater Treatment Systems

There is one active or pending MGWPCS discharge permit for human waste disposal within the TPA. That is for the Lakeside at Georgetown subdivision near Georgetown Lake (**Figure F-1**) that serves 22 single family homes. A MGWPCS permit is required for wastewater systems discharging to groundwater that are designed to treat 5,000 gallons per day or more.

Public wastewater treatment systems have at least 15 service connections or regularly serve at least 25 persons daily for any 60 days or more in a calendar year. Multi-user wastewater treatment systems have 3–14 service connections and don't regularly serve 25 or more persons for any 60 days in a calendar year. Multi-user and public wastewater systems that discharge to groundwater are not regulated via the MGWPCS unless they: 1) are designed to treat 5,000 gallons per day or more; or 2) are aerobic package plant systems, mechanical treatment plants, and nutrient removal systems, which require a high degree of operation and maintenance or systems which require monitoring pursuant to Administrative Rules of Montana 17.30.517(1)(d)(ix). The DEQ Subdivision Review Section database records three multi-user or public wastewater treatment systems that were approved since 2000 in the watershed (**Figure F-1**); one of those is Lakeside at Georgetown subdivision discussed above. The other two systems serve a total of 44 single family homes or condominiums, and are also in the vicinity of Georgetown Lake (**Figure F-1**). Records are not available for public or multi-user systems approved prior to 2000.

F3.4 Household Wastewater Treatment System Estimates

Using Geographic Information System (GIS) data, DEQ estimates that there are 1,623 septic systems in the Flint Creek watershed. DEQ reached this estimate using GIS layers of structures in Granite and Deer Lodge Counties. The counties had previously contracted a GIS consultant (MaPS, Inc.) to develop these layers to support emergency responders. Locations and type of structure were established by field mapping and aerial photograph interpretation, as of summer 2009. For DEQ, the location and type of structure are the relevant attributes.

DEQ reduced the dataset to structures within the watershed boundary, and of the following types: apartment, cabin, house, mobile home. Commercial, civic or other public facilities were excluded on the assumption that their wastewater systems serve the same population as the residences. DEQ recognizes that a subset of the residential structures in the watershed are seasonal or vacation homes, however there are no available data on occupancy.

The city of Philipsburg is sewered and served by a WWTP. Dick Hoehn of the Philipsburg Public Works Department provided a map of the sewer system. Using this, DEQ manually deleted those structures within the corresponding area.

F3.4.1 Distribution and Growth

Yearly septic permit approval rates in the watershed on parcels less than 20 acres could not be determined from the available databases. However, as an analogy to approximate population growth rates in the watershed, the number of subdivision lots under 20 acres approved in each state fiscal year since 1990 are shown in **Figure F-2** for both Granite and Deer Lodge Counties (the TPA lies primarily within Granite County but a small portion east of Georgetown Lake lies within Deer Lodge county). Note that a subdivision lot approval does not indicate occupancy of the lot, but rather that the lot was created for a residential or commercial use. Since 1990 the highest lot growth rate occurred in 2005 while the lowest lot growth rate occurred in 1993. Recently available information (Granite County Sanitarian, Lanes, Chad, personal communication 2013) indicates that approximately 390 septic permits were issued within the TPA between 1990 and 2010, which is much less than the number of lots created as shown in **Figure F-2**.

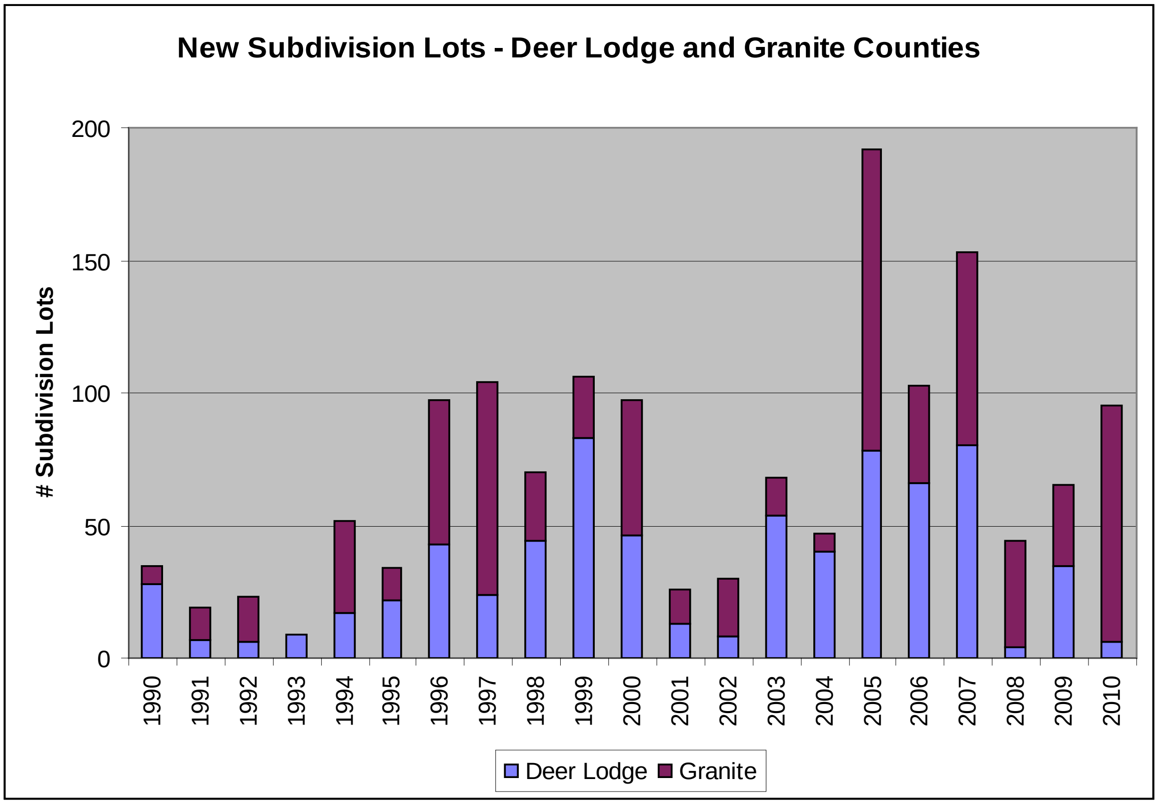

Figure F-2. Subdivision Lot Approvals in the Flint Creek Watershed by County

Using 2000 census data the distribution of septic systems within the TPA was estimated. Most of the medium (50-299 systems per square mile) and high density (300+ systems per square mile) areas are clustered around Georgetown Lake with some medium density areas near Philipsburg and Maxville. The remainder of the TPA has low septic density of less than 50 septic systems per square mile.

F3.4.2 Failure Rates

National septic failure rates may range from 10–20% (Environmental Protection Agency (EPA)/625/R-00/008). It is likely that there are lower failure rates in the Flint Creek watershed due to more recent population growth rates, and thus newer septic systems, when compared to national growth.

F4.0 Simple Septic Loading and Attenuation Estimates

A simple process for estimating household septic load estimates entering surface waters within the Flint Creek Watershed will be used to help determine if the parameterization and processes within SWAT produces reasonable septic load results. The input variables and results of this effort will be used to assist in parameterization of the septic related components of the SWAT model. Results of this simple assessment effort will also help determine if septic sources are represented reasonably within the Flint Creek SWAT model.

Methodology for the simple septic loading and attenuation assessment is provided in **Attachment A**. GIS was used to estimate how many septic systems are within certain distance to the stream network and to determine local soil types. The data derived from the GIS effort are then assessed via excel spreadsheet analysis. Simple loading and attenuation factors are applied to each septic system based upon the category it falls under, which are based upon soil type at the septic system, soil type at nearby stream and distance to the nearest gaining stream. The assessment also assumes all septic systems are conventional treatment, it does not account for consideration of level 2 systems, nor failing septic systems. The results of this effort provide an estimated load that enters any portion of the stream network in the watershed (**Table F-1**). The overall estimated treatment of nitrate and total phosphorus (TP) from septic systems to the point the load enters surface water is estimated at 64% and 85%, respectively. The loading values for nitrate and TP in **Table F-1** from single family homes are based on average drainfield loading rates of 30.5 lbs/year/home and 6.44 lbs/year/home, respectively.

Septic loading appears to be concentrated in two or three areas. Recently, the Georgetown Lake area has seen an increasing number of septic systems due to mostly recreational property development. The area around Philipsburg and the surrounding upper Flint Creek valley also contain higher septic density than surrounding parts of the watershed. Individual septic systems service the towns of Hall and Maxville which are located in the lower Flint Creek and Boulder sub-watersheds respectively.

Table F-1. Estimated Number of Systems, Nutrient Reduction, and Nutrient Loads from Septic Systems in the Flint Creek Watershed during 2009

Sub-Watershed	Number of Septic Systems	% Nitrate Reduction	% Phosphorus Reduction	Total Nitrate Loading at Drainfields (lbs/year)	Total Phos. Loading at Drainfields (lbs/year)	Total Nitrate Loading to Surface Water (lbs/year)	Total Phos. Loading to Surface Water (lbs/year)
Barnes Creek	24	66	88	732	155	249	19
Boulder Creek	106	57	74	3233	683	1390	177
Douglas Creek North	3	60	70	92	19	37	6
Georgetown Lake(1)	873	65	86	26627	5622	9319	787
Lower Flint Creek	167	67	91	5094	1075	1681	97
Lower Willow Creek	34	66	89	1037	219	353	24
Middle Flint Creek	92	65	86	2806	592	982	83
N. Fork Lower Willow Creek	6	67	90	183	39	60	4
Philipsburg	159	58	80	4850	1024	2037	205
Princeton Gulch	6	57	60	183	39	79	15
S. Fork Lower Willow Creek	3	67	90	92	19	30	2
Smart Creek	12	65	85	366	77	128	12
Trout Creek	55	67	91	1678	354	554	32
Upper Flint Creek	83	62	79	2532	535	962	112

Table F-1. Estimated Number of Systems, Nutrient Reduction, and Nutrient Loads from Septic Systems in the Flint Creek Watershed during 2009

Sub-Watershed	Number of Septic Systems	% Nitrate Reduction	% Phosphorus Reduction	Total Nitrate Loading at Drainfields (lbs/year)	Total Phos. Loading at Drainfields (lbs/year)	Total Nitrate Loading to Surface Water (lbs/year)	Total Phos. Loading to Surface Water (lbs/year)
WEIGHTED AVERAGE		**64**	**85**				
TOTALS	**1623**			**49502**	**10452**	**17860**	**1575**

(1) The loading values per home are based on typical occupancy rates for single-family homes. That occupancy rate may not apply in areas with a high percentage of vacation or second homes. In the Flint Creek watershed, the area surrounding Georgetown Lake likely falls into that category. Therefore, the nitrate and total phosphorus loading values in the Georgetown Lake sub-watershed may actually be 50%, or possibly even less, than what is listed depending on the actual home occupancy rate.

F5.0 Soil & Water Assessment Tool Model Groundwater Processing and Septic Assessment

F5.1 Soil & Water Assessment Tool Nutrient Processing in Soils and Groundwater

The SWAT depicts nitrogen and phosphorus transport in the groundwater profile in different manners. To understand this requires a basic understanding of SWAT's groundwater modeling approach. On a very basic level, SWAT splits the groundwater received from the soil layer in each sub-basin into two aquifers – a shallow, unconfined aquifer, and a deep, confined aquifer. Water entering the shallow aquifer can contribute back into the soil profile (revap), or back into the surface water system by recharging the main channel (groundwater/base flow). Water entering the deep aquifer is considered lost to the system (i.e., it flows back into the system outside of the modeled watershed).

Both nitrogen and phosphorus are modeled by SWAT in the soil profile, and also addressed within the shallow groundwater aquifer (any flows or nutrients entering the deep aquifer are considered lost to the system). Within the soil profile, nutrients are separated into different types based on solubility and reactivity. Nitrogen is divided into five separate types, including ammonium, nitrate, stable organic, reactive organic, and fresh organic. Phosphorus is divided into six separate types, including stable mineral, active mineral, soluble mineral, active organic, stable organic, and fresh organic. Each of these reacts within the soil profile via different mechanisms. Nutrients in the soil profile may move laterally within SWAT depending upon soil slope, hydraulic conductivity, distance to stream, and other factors. Lateral soil flow enters the stream at the sub-watershed node. However, only the soluble types are important to the groundwater function – the rest are important to overall SWAT modeling but do not influence the groundwater conditions.

F5.1.1 Nitrogen

After water containing soluble nitrate leaves the soil profile (but before it enters the aquifer), it enters the vadose zone. Although no chemical transformations or losses are simulated in the vadose zone, it does take time for the water to pass through this (depending on depth and other factors). To simulate this time lag, SWAT utilizes an exponential decay weighting function to account for the time spent in the vadose zone. This basically relates (on a daily basis) the amount of nitrate entering the shallow aquifer as a fraction of the nitrate in the water leaving the soil profile.

Once in the shallow aquifer, several things may happen to the nitrate. It may remain in the aquifer, move into the deep aquifer via recharge (where it is lost from the system), move into the main channel via groundwater/base flow, or move back into the soil profile (revap) as a response to low water content in the soil profile. Each of these functions is governed by a mass balance equation using a daily time step. Variables in these calculations include the inflows and incoming concentrations from other sources (shallow aquifer, deep aquifer, revap/soil profile, and groundwater flow between the main channel), which are based on additional SWAT algorithms and mass balances based upon field capacity, hydrologic conductivity, porosity and other factors.

Furthermore, nitrogen is reactive and can be removed from the groundwater via several mechanisms (e.g., uptake by bacteria present in the aquifer, undergo chemical transformations in the aquifer). To account for this, and all other biological or chemical transformations in the aquifer, SWAT uses a half-life function. This equation specifies the number of days that nitrate will remain in the shallow aquifer before the nitrate concentration is reduced by one-half. This is a standard half-life equation with the rate constant for nitrate removal equal to the natural log of two (2) divided by the half-life time. The user inputs the nitrate removal half-life. Based on the denitrification rate constant discussed in the nitrogen attenuation paper (**Attachment A**) (0.025 day-1), it would take about 11 years of travel time in the subsurface to remove almost all the nitrogen discharged from a septic system. So the half-life should be approximately 5.5 years in the SWAT model, with some adjustments that may also depend upon the vadose decay rate.

F5.1.2 Phosphorus

Phosphorus is much less soluble than nitrate, and therefore does not transport via groundwater as readily. It readily binds into the soil in many situations. Therefore, SWAT does not utilize any modeling for phosphorus determinations in groundwater. The user simply inputs a concentration of soluble phosphorus in the shallow aquifer, and groundwater flow into the main channel is then calculated based on hydraulic loading. The phosphorus concentration in the shallow groundwater remains constant throughout the simulation period. There is no phosphorus-related interaction between the soil horizon and the shallow aquifer; phosphorus is assumed to bind into the soil and remain there. Alternatively, the groundwater can transport a phosphorus load based upon a constant concentration of phosphorus that remains throughout the simulation. The groundwater phosphorus concentration is set by the user and is completely separate from any soil phosphorus functions. Therefore any phosphorus load artificially introduced into the soil layers to simulate a septic system is not transferred to the groundwater.

F5.1.3 Proposed Septic Nutrient Load Assessment for Flint Creek Soil & Water Assessment Tool Model

SWAT 2009 may provide a useful septic loading tool (Biozone module) that could be used simulate the 1,623 septic systems estimated to exist in the Flint Creek drainage. Otherwise, an alternative process will be incorporated into the SWAT model as was described in **Section F4.0**.

ATTACHMENT FA – NITROGEN AND PHOSPHORUS MIGRATION AND ATTENUATION ASSESSMENT FROM SUBSURFACE WASTEWATER TREATMENT SYSTEMS

FA1.0 INTRODUCTION

This document presents a summary of the factors affecting migration and attenuation of nitrogen and phosphorus after disposal from subsurface wastewater treatment systems (i.e., septic systems). This summary is used to support methods proposed for determining nitrogen and phosphorus reduction as these nutrients migrate towards surface waters.

The methods described in the document should not be used to determine nutrient attenuation on a small scale (e.g., single development/municipality discharge) due to the potentially wide variation in nutrient attenuation between sources in similar settings. These methods are designed for use on a larger basin-wide scale that effectively allows averaging of the processes that occur in the subsurface.

While the processes of nutrient attenuation described in this document are well documented, the attenuation percentages proposed are estimates. Where possible, the results of the methods described should be verified with site-specific data.

FA2.0 NITROGEN

Nitrogen in partially treated domestic wastewater (in the septic tank) is primarily in the form of ammonia. Disposal of 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 (Montana Department of Environmental Quality, 2009; Morgan et al., 2007; Idaho Department of Environmental Quality, 2002; National Decentralized Water Resources Capacity Development Project, 2005a; Heatwole and McCray, 2006; Toor et al., 2011). Unless an advanced wastewater system is used (referred to as a level 2 system in Montana), conventional treatment removes between 10 and 30 percent of the nitrogen in the wastewater (Costa et al., 2002; Lowe et al., 2007; Gold and Sims, 2000; Laak, 1981; Pell and Nyberg, 1989; Rosen et al., 2006; Seabloom et al., 2004). That treatment level is accounted for in the nitrogen concentration (50 mg/L) that Montana estimates is discharged from the typical septic system serving a single-family home. Septic systems are not designed to complete the final step of the nitrogen cycle, conversion of nitrate to nitrogen gas (denitrification), which then dissipates into the atmosphere and does not have any further impacts to groundwater or surface water. Denitrification generally occurs after drainfield treatment, and is difficult to predict.

In Montana, the estimated nitrate loading rate for a single-family home septic system is based on an average concentration of 50 mg/L and an average effluent rate of 200 gallons per day (Montana Department of Environmental Quality, 2009). Those concentration and effluent rates are within the range of published values (U.S. Environmental Protection Agency, 2002). Those values provide a nitrogen loading rate of 30.5 lbs/year for a conventional wastewater system. For comparison purposes, the nitrogen loading rate for a level 2 system is 14.6 lbs/year.

Denitrification requires the correct environment to occur, the key factors are adequate temperature (typically above 10 °C), a food source for the bacteria (typically carbon), an anoxic environment (generally an oxygen range of less than 1-2 mg/L), and the correct bacteria. A riparian zone with shallow groundwater is the most common environment that has those conditions (Tri-State Water Quality Council, 2005; Gilliam, 1994; Gold and Sims, 2000; Rosenblatt et al., 2001; Harden and Spruill, 2008). A carbon source is cited as the most common limiting factor for denitrification (Gold and Sims, 2000; Rivett et al., 2008; Starr and Gillham, 1989). Studies have identified "micro-sites" of low oxygen in shallow groundwaters, which are typically assumed to be rich in oxygen, to provide the necessary anoxic environment (Gold and Sims, 2000; Jacinthe et al., 1998; Parkin, 1987). The required bacteria are generally ubiquitous in the environment, and will naturally thrive when the conditions are correct and there is a nitrogen source. However, it should be noted that the U.S. Environmental Protection Agency (2002) stated that "Denitrification has been found to be significant in the saturated zone only in rare instances where carbon or sulfur deposits are present". This conclusion is contrary to the numerous studies that have found high denitrification rates in common environments; the same EPA document recognizes some of those studies.

Because fine-grained soils are more likely to contain two of the conditions necessary for denitrification, anoxic conditions and carbon, fine-grained soils typically provide better conditions for denitrification than coarse-grained soils (Mueller et al., 1995; Tesoriero and Voss, 1997; Umari et al., 1995; Briar and Dutton, 2000). Anderson (1998) used results from several studies to show a correlation (r=0.91) between denitrification rates and soil organic content. One study (Ricker et al., 1994) estimated the amount of denitrification beneath drainfields as 15% for sandy soils and 25% for finer soils.

Denitrification rates are site-specific and the rates can vary considerably in similar environments (Starr and Gillham, 1993; Robertson et al., 1991). Some studies have provided measurable chemical characteristics to determine where denitrification is more likely to occur (Trojan et al., 2002; Minnesota Pollution Control Agency, 1999), but the studies typically only provide relative denitrification rates (e.g., high or low). However, several studies (National Decentralized Water Resources Capacity Development Project, 2005c; Kirkland, 2001; McCray et al., 2005), have published a specific denitrification rate based on the median of cumulative frequency distributions of field measured denitrification rates (0.025 day-1). At that rate, it takes over 10 years to denitrify all of the nitrate from a source. At typical groundwater velocity rates of 0.1 to 10 ft/day wastewater could travel between 400 and 40,000 feet in that time. Using a single denitrification rate for all situations may be unrealistic as one study indicated it would take a denitrification rate that ranges over 3 orders of magnitude to provide a 95% confidence interval (Heatwole and McCray, 2006). McCray et al. (2005) could not correlate soil type to denitrification rate due to variability in the existing data; therefore, the median denitrification rate was not used for the proposed method of estimating nitrate reduction.

Another factor that has been correlated with denitrification is travel time in the environment: the longer the nitrate is in the environment the more time it has to encounter the correct conditions for denitrification (Kroeger et al., 2006). Distance is used in the proposed methods instead of travel time because it is easier to measure distances than groundwater travel time which requires three parameters that are difficult and/or expensive to measure for large areas: hydraulic gradient, hydraulic conductivity and effective porosity.

Based on the existing information, the following method has been developed to estimate the nitrogen reduction as wastewater migrates from a drainfield to a receiving surface. This method uses a matrix (see **Table FA-1**) combining three factors that impact the amount of denitrification: soil type beneath the drainfield; soil type in the riparian area; and distance to surface water. In **Table FA-1**, each drainfield is assigned a percent nitrate reduction for each of the three criteria. The percent reductions for each column are then added to provide the total percent nitrate removal for that septic system. The nitrate loading rate (30.5 lbs/year for a conventional system) to the surface water is then reduced accordingly. Any system with a 100% or higher reduction contributes no nitrate to the surface water.

This method assumes steady-state conditions; it does not account for the time needed for the nitrogen load from a new discharge source to migrate towards the receiving surface water. That lag time is dependent on the travel rate through both the vadose and saturated zones.

This method (and the phosphorus method described below) does not account for failing septic systems because the number of hydraulically failing systems where wastewater is flowing at the surface (and likely to bypass natural treatment in soils) is typically a small percentage of the total number of septic systems on a basin wide scale and is not a significant nutrient load for TMDL purposes. A surfacing, failing system is also likely to be repaired quickly, further minimizing any impacts to surface waters. However, there may be site-specific situations where failing septic systems are a significant source and need to be accounted for using a different method.

Table FA-1. Nitrogen Attenuation Factors for Septic System Discharges to Groundwater

Percent Nitrogen Load Reduction(1)	Soil Type @ Drainfield(2)	Soil Type within 100' of Surface Water(2)	Distance to Surface Water (ft)
0	A	A	0 – 100
10	B		101 – 500
20	C	B	501 – 5,000
30	D	C	5,001 – 20,000
50		D	20,001+

(1) The total nitrogen reduction is the sum of the individual reductions for each column of the table. For example a drainfield that is in a type C soil (20%) that drains to a surface water with type B soil (20%) and is 200 feet from the surface water (10%) would reduce their nitrogen load to the surface water by 50% from what is discharged from the drainfield.

(2) Soil descriptions are available via the Natural Resources Conservation Service (NRCS) web soil survey 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 Qualities and Features" – "Drainage Class". The NRCS soil survey has seven soil drainage classes that are correlated to the A, B, C and D designation in the table as follows:

A = excessively drained or somewhat excessively drained
B = well drained or moderately well drained
C = somewhat poorly drained
D = poorly drained or very poorly drained

Within the defined area of interest, the soil survey application provides the percent of soil types with these attributes. That feature provides a quick way to determine the percent of each soil type and therefore the percent reduction for each area of interest defined.

FA3.0 PHOSPHORUS

Phosphorus, which has much lower mobility than nitrogen, is removed in soils below drainfields by two primary processes, adsorption and precipitation. Precipitation is a slower process compared to adsorption but may be the more important process for retarding the migration of phosphorus. Soils may have a limited amount of adsorption capacity which could allow migration of phosphorus after reaching equilibrium (Gold and Sims, 2000). However, precipitation reactions may occur indefinitely with the correct conditions thereby limiting phosphorus migration indefinitely (Lombardo, 2006; Robertson et al., 1998). Lombardo (2006) estimated that phosphorus travel times to nearby surface waters could range from tens of years to hundreds of years depending on the types of soils between the source and waterbody. The vadose zone is considered the primary location for phosphorus retardation, once it reaches groundwater phosphorus migration is generally faster than in the vadose zone.

In Montana, the estimated phosphorus loading rate for a single-family home septic system is based on an average concentration of 10.6 mg/L and an average effluent rate of 200 gallons per day (Montana Department of Environmental Quality, 2009). Those concentration and effluent rates are within the range of published values (U.S. Environmental Protection Agency, 2002). Those values provide a loading rate of 6.44 lbs/year for a conventional wastewater system.

Non-calcareous soils retard the movement of phosphorus more than calcareous soils due to the calcareous soils ability to maintain pH levels where phosphorus precipitation does not readily occur (Lombardo, 2006; 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% calcium carbonate and non-calcareous soils as those containing less than 1% calcium carbonate.

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.

Easily measurable wastewater phosphorus plumes 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; Gold and Sims, 2000; Lombardo, 2006; Reneau et al., 1989; Robertson et al., 1998). This indicates that a significant portion of the phosphorus is quickly bound up shortly after being discharged. However, in many cases low level phosphorus detection limits are not used in groundwater analyses, and the existence of long, low concentration phosphorus plumes may have been overlooked (Houston, 2001).

Due to the small amount of phosphorus that migrates significant distances, some methods assume that only failing systems contribute phosphorus to surface water. For example, the MANAGE (Method for Assessment, Nutrient-Loading, and Geographic Evaluation of Nonpoint Pollution) nutrient migration model (Kellogg et al., 2006) only accounts for phosphorus discharges from failing drainfields. Other information (Tri-State Water Quality Council, 2005; Gold and Sims, 2000; National Decentralized Water Resources Capacity Development Project, 2005b) also implicates failing or improperly sited drainfields (e.g., drainfields located over shallow groundwater, in coarse soils, or too close to surface water) as a greater threat to surface water than properly constructed and sited systems.

Lombardo (2006) suggested that phosphorus migration to surface waters is only a problem in areas with high groundwater tables and higher groundwater velocities (the report provided a lower end for the high velocities of approximately 0.2 to 3 feet/day). Below those velocities soils typically contain higher amounts of clay and/or silt, thus increasing the soils adsorption capacity.

Except for failing or poorly sited septic systems, existing evidence indicates that only small amounts of phosphorus migrate to surface waters, but that in some cases even small amounts can have noticeable impacts to surface water quality. To be consistent with existing information on phosphorus migration the proposed method to estimate phosphorus reduction was designed to estimate relatively high percentages of phosphorus removal.

Based on the existing information, the following method has been developed to estimate the phosphorus reduction as wastewater migrates from a drainfield to a receiving surface. This method uses (**Table FA-2**) a matrix, similar to the one used for nitrogen , combining three factors that impact the amount of phosphorus reduction: soil type beneath the drainfield; calcium carbonate percent in the soil beneath the drainfield; and distance to surface water. In **Table FA-2**, each drainfield is assigned a percent phosphorus reduction for only one of the three soil type columns (which combines the soil and calcium carbonate type), and then an additional percent phosphorus reduction for the last column (distance to surface water). The percent reductions for each column are then added to provide the total percent phosphorus removal for that septic system. The phosphorus loading rate (6.44 lbs/year for a conventional or level 2 system) to the surface water is then reduced accordingly. Any system with a 100% or higher reduction contributes no phosphorus to the surface water.

This method assumes steady-state conditions; it does not account for the time needed for the phosphorus load from a new discharge source to migrate towards the receiving surface water. That lag time is dependent on the travel rate through both the vadose and saturated zones.

Table FA-2. Phosphorus Attenuation Factors for Septic System Discharges to Groundwater

Percent Phosphorus Load Reduction[1]	Soil Type @ Drainfield[2, 3] ($CaCO_3$ <= 1%)	Soil Type @ Drainfield[2, 3] ($CaCO_3$ >1% and <15%)	Soil Type @ Drainfield[2, 3] ($CaCO_3$ >=15%)	Distance to Surface Water (ft)
0	A	A	A	0 – 100
10			B	
20		B	C	
30	B		D	101 - 500
40		C		
60	C	D		501 - 5,000
90	D			
100				5,001 +

(1) The total phosphorus reduction is the sum of the individual reductions for the soil type (only use one of the three soil columns) and the distance to surface water. For example a drainfield that is in a type B soil with less than 1% CaCO3 (30%) and is 200 feet from the surface water (30%) would reduce their nitrogen load to the surface water by 60% from what is discharged from the drainfield.

(2) Soil descriptions are available via the Natural Resources Conservation Service (NRCS) web soil survey 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 Qualities and Features” – “Drainage Class”. The NRCS soil survey has seven soil drainage classes that are correlated to the A, B, C and D designation in the table as follows:

A = excessively drained or somewhat excessively drained
B = well drained or moderately well drained
C = somewhat poorly drained
D = poorly drained or very poorly drained

Within the defined area of interest, the soil survey application provides the percent of soil types with these attributes. That feature provides a quick way to determine the percent of each soil type and therefore the percent reduction for each area of interest defined.

(3) $CaCO_3$ percent is available via the NRCS web soil survey 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 ($CaCO_3$)”. Within the defined area of interest, the soil survey application provides the percent of land with the percent of $CaCO_3$. That feature provides a quick way to determine the percent of area of different $CaCO_3$ percentages and therefore the percent reduction for each area of interest defined.

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.

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.

Costa, Joseph E., George Heufelder, Sean Foss, Newton P. Milham, and Brian Howes. 2002. Nitrogen Removal Efficiencies of Three Alternative Septic System Technologies and a Conventional Septic System. *Environment Cape Cod.* 5(1): 15-24.

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

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

Houston, A. J. 2001. Estimation of the Contribution of Phosphorus From On-Site Sewage Disposal Systems to Lakes: Canada Mortgage and Housing Corporation.

Idaho Department of Environmental Quality. 2002. Nutrient-Pathogen Evaluation Program for On-Site Wastewater Treatment Systems. http://www.deq.idaho.gov/media/493564-nutrient_pathogen_eval_guide.pdf. Accessed 6/14/2013.

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.

Kellogg, Dorothy Q., Marie Evans Esten, Lorraine Joubert, and Arthur J. Gold. 2006. Database Development, Hydrologic Budget and Nutrient Loading Assumptions for the "Method for Assessment, Nutrient-Loading, and Geographic Evaluation of Nonpoint Pollution" (MANAGE) Including the GIS-Based Pollution Risk Assessment Method.

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.

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.

Laak, Rein. 1981. Denitrification of Blackwater With Greywater. *ASCE Journal of Environ.Eng.Div.* 58: 581-590.

Lanes, Chad. 2013. Personal Communication. Regensburger, Eric.

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

Lowe, Kathryn S., Nathan K. Rothe, Jill M. B. Tomaras, Kathleen DeJong, Maria B. Tucholke, Jorg Drewes, John E. McCray, and Junko Munakata-Marr. 2007. Influent Constituent Characteristics of the MOdern Waste Stream From Single Sources: Literature Review. Water Environment and Research Foundation.

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.

Minnesota Pollution Control Agency. 1999. Estimating Ground Water Sensitivity to Nitrate Contamination.

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., 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.

National Decentralized Water Resources Capacity Development Project. 2005a. Application of Simulation-Optimization Methods for Management of Nitrate Loading to Groundwater From Decentralized Wastewater Treatment Systems Near La Pine, Oregon. U.s. Geological Survey.

-----. 2005b. Micro-Scale Evaluation of Phosphorus Managment: Alternative Wastewater Systems Evaluation.

-----. 2005c. Quantifying Site-Scale Processes and Watershed-Scale Cumulative Effects of Decentralized Wastewater Systems. Colorado School of Mines.

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

Pell, Mikael and Fred Nyberg. 1989. Infiltration of Wastewater in a Newly Started Pilot Sand-Filter System: III. Transofrmation of Nitrogen. *Journal of Environmental Quality.* 18: 463-467.

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.

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.

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.

Seabloom, Robert W., Terry Bounds, and Ted Loudon. 2004. University Curriculum Development for Decentralized Wastewater Management - Septic Tanks. http://www.onsiteconsortium.org/ed_curriculum/University/IV.%20B.%20Septic%20Tanks/University_Septic_Tanks_Text.pdf:

Starr, Robert C. and Robert W. Gillham. 1989. Controls on Denitrification in Shallow Unconfined Aquifers. *Contaminant Transport in Groundwater.*: 51-56.

-----. 1993. Denitrification and Organic Carbon Availability in Two Aquifers. *Ground Water.* 31(6): 934-47.

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.

Tri-State Water Quality Council. 2005. Septic System Impact on Surface Waters: A Review for the Inland Northwest. Sandpoint, ID: Tri-State Water Quality Council.

Trojan, Michael D., Moira E. Campion, Jennifer S. Maloney, James M. Stockinger, and Erin P. Eid. 2002. Estimating Aquifer Sensitivity to Nitrate Contamination Using Geochemical Information. *Ground Water Monitoring and Remediation.* 22(4): 100-108.

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.

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.