Development and Application of a Spatially-Explicit Nitrogen Mass-Balance Model for the Wood River Valley Watershed, Idaho

by

Julianna B. Connolly

B.A. Chemistry
Williams College, 2001

and

Devin L. Shaffer

B.S. Civil Engineering
Oklahoma State University, 2002

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Signatures of Authors: ____________________

Department of Civil and Environmental Engineering
May 9, 2003

Certified by: ____________________

Peter Shanahan
Lecturer, Civil and Environmental Engineering
Thesis Supervisor

Accepted by: ____________________

Oral Buyukozturk
Chairman, Departmental Committee on Graduate Studies

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ABSTRACT

A spatially-explicit nitrogen mass-balance model for the Wood River Valley Watershed in south-central Idaho is developed (Blaine County Evaluation and Assessment of Nitrogen Sources (BEANS) model). The study is performed on behalf of the Blaine County Commissioners in response to concerns regarding increased nitrogen loading to the Big Wood and Little Wood Rivers as a result of continuing population growth in Blaine County. Nitrogen inputs incorporated in the BEANS model include atmospheric deposition, fertilizer applications, nitrogen fixation, livestock waste, and domestic wastewater from both on-site septic systems and municipal wastewater treatment plants. Nitrogen losses include ammonia volatilization, uptake by plants, retention by soils, aquifer denitrification, and instream denitrification. The magnitude of nitrogen inputs and losses are determined using basin-specific information when possible and from applicable literature when basin-specific values were not available. These values vary as a function of land use. Nitrogen loads are calculated for the entire Wood River Valley Watershed as well as for two sub-watersheds, referred to as the Upper Valley and the Northern Valley. The majority of future population growth in the watershed is expected to occur in these two sub-watersheds. The BEANS model calculates nitrogen loads for the entire watershed, the Upper Valley, and the Northern Valley of 664,500 kg N/yr, 165,000 kg N/year, and 55,600 kg N/year respectively. The nitrogen yields are 0.98 kg N/ha for the entire watershed, 0.74 kg N/ha for the Upper Valley, and 0.55 kg N/ha for the Northern Valley. Agricultural sources, primarily cattle waste and fertilizer applications, contribute 70% of the nitrogen to the entire watershed load. Wastewater sources contribute only 5% to the entire watershed load, but the relative magnitude of wastewater sources is greater in the Upper Valley (17%) and Northern Valley (33%). The BEANS model is used to analyze how future land use changes will affect the magnitude of the watershed nitrogen load. Reductions in agricultural nitrogen fertilizer application rates are identified as an option for reducing the watershed nitrogen load without losses in net agricultural production. Controlling the size of new residential lots and the nature of residential wastewater treatment could also provide reductions in the watershed nitrogen load without limiting the possibility of future economic development within the watershed.

Thesis Supervisor: Dr. Peter Shanahan
Title: Lecturer, Civil and Environmental Engineering

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1. INTRODUCTION

The Wood River Valley Watershed is composed of the land area that drains into both the Big Wood and Little Wood Rivers in south-central Idaho. The watershed boundary is defined as the combination of the Big Wood River and Little Wood River hydrologic unit boundaries (HUC # 17040219, HUC # 17040221) and encompasses approximately 680,000 ha (2,600 sq. mi.).

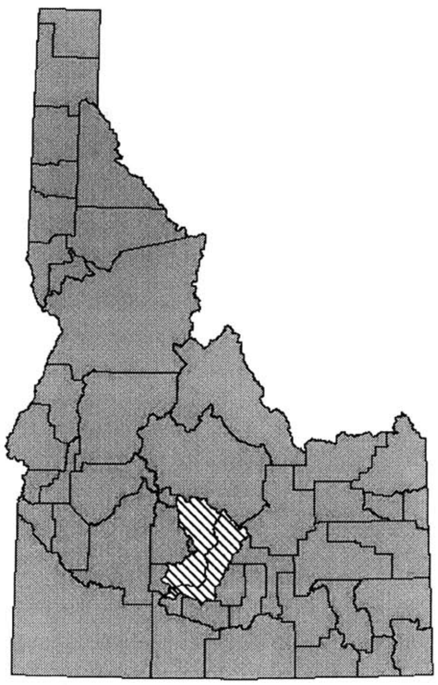


Figure 1-1: Location of the Wood River Valley Watershed in Idaho

The Little Wood River drains into the Big Wood River at the southern end of the watershed, and eventually the flow from both rivers drains into the Snake River. Fifteen cities are located within the watershed (Figure 1-2).

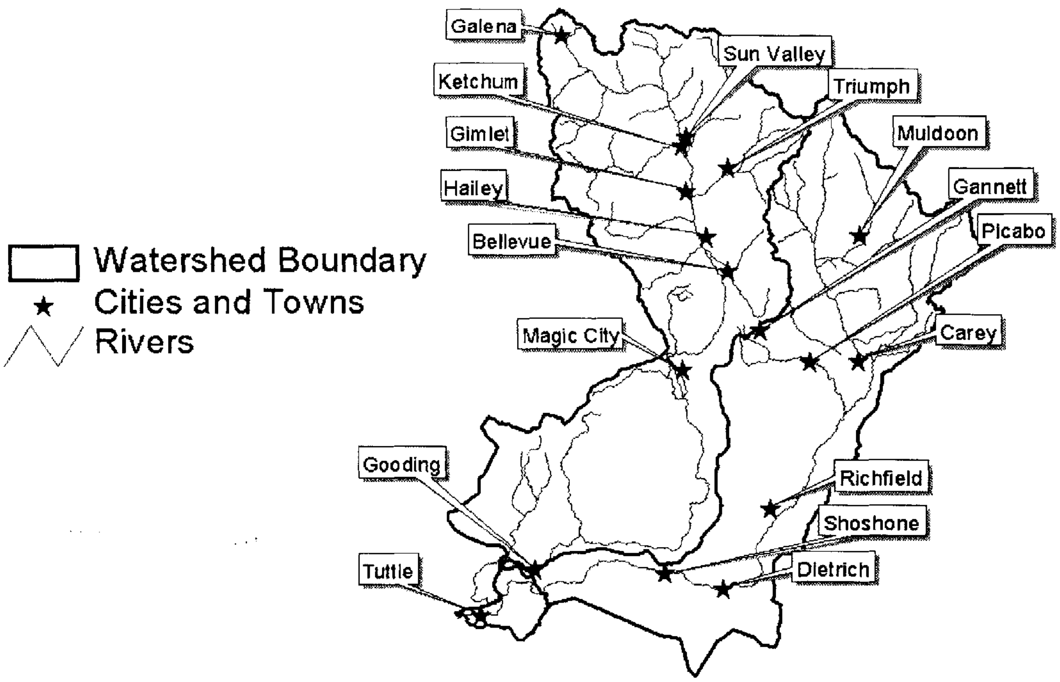


Figure 1-2: Cities within the Wood River Valley Watershed

Hailey, Gooding, Ketchum, and Bellevue have the largest populations of all the incorporated areas. Three of these larger cities (Hailey, Ketchum, Bellevue) are contained in the northern part of the watershed.

This thesis discusses the development and application of a nitrogen mass-balance model for the Wood River Valley Watershed. Concern about water quality has developed recently among local officials as a result of rapid population growth in the northern watershed. The mass-balance model enables us to identify and quantify nitrogen sources that are contributing to the nitrogen contamination of ground and surface water within the watershed. Using the model to analyze potential land use changes that may occur within the watershed in the future, we are able to predict how continued development will affect water quality with respect to expected future nitrogen loads.

1.1 CLEAN WATER ACT

The Federal Clean Water Act (CWA) requires that states and tribes restore and maintain the chemical, physical, and biological integrity of the nation's waters (33 USC § 1251.101). As directed by Section 303 of the CWA, states and tribes are to adopt necessary water quality standards to ensure the protection of fish, shellfish, and other wildlife while at the same time allowing recreation in and on the waters whenever possible. Section 303(d) of the CWA requires that states and tribes identify and prioritize water bodies that are water quality limited. For waters that are included on a state's 303(d) list because they do not meet water quality standards, the state must develop a watershed total maximum daily load (TMDL) for the pollutants responsible for the reduced water quality.

In 1998 the Idaho Department of Environmental Quality, acting pursuant to Section 303(d) of the CWA, designated sections of the Big Wood River and Little Wood River as having impaired water quality with respect to bacteria, dissolved oxygen, flow alteration, ammonia, nutrients, and sediment load. Table1-1 summarizes the reaches of the Big Wood and Little Wood Rivers that are currently listed on Idaho's 303(d) list. Figure 1-3 illustrates the location of these river reaches within the watershed.

Table 1-1: Idaho's 303(d) Listed Reaches within the Wood River Valley Watershed

Reach Name	***Boundaries***	***Year Listed***	***Year TMDL Due***
Big Wood River	Little Wood River to Interstate	1996	2001
Big Wood River	Highway 75 to Little Wood River	1996	2001
Big Wood River	Magic Reservoir to Highway 75	1996	2001
Big Wood River	Glendale Diversion to T1NR18ES35	1996	2001
Big Wood River	Trail Creek to Glendale Diversion	1996	2001
Rock Creek	Headwaters to Magic Reservoir	1996	2001
Croy Creek	Elk Creek to Big Wood River	1996	2001
Owl Creek	Headwaters to Big Wood River	1998	2006
Eagle Creek	Headwaters to Big Wood River	1998	2006
Baker Creek	Headwaters to Norton Creek	1998	2006
Placer Creek	Headwaters to Warm Springs Creek	1998	2006
Greenhorn Creek	Headwaters to Big Wood River	1998	2006
East Fork Wood River	Headwaters to Blind Canyon	1998	2006
Cove Creek	Headwaters to East Fork Wood River	1998	2006
Quigley Creek	Headwaters to mouth	1998	2006
Seamans Creek	Headwaters to Big Wood River	1998	2006
East Fork Rock Creek	Headwaters to Rock Creek	1998	2006
Thorn Creek	Thorn Reservoir to Schooler Creek	1998	2006
Horse Creek	Headwaters to Big Wood River	1998	2006
Lake Creek	Headwaters to Big Wood River	1998	2006
Little Wood River	Richfield to Big Wood River	1996	2003
Little Wood River	Silver Creek to Richfield	1996	2003
Little Wood River	East Canal Diversion to Silver Creek	1996	2003
Little Wood River Reservoir		1996	2003
Dry Creek	Headwaters to Little Wood River	1996	2003
Fish Creek	Fish Creek Reservoir to Carey Lake	1996	2003
Fish Creek Reservoir		1996	2003
Muldoon Creek	South Fork Muldoon to Little Wood R.	1998	2006
Loving Creek	Headwaters to Silver Creek	1998	2006
Fish Creek	Headwaters to Fish Creek Reservoir	1998	2006

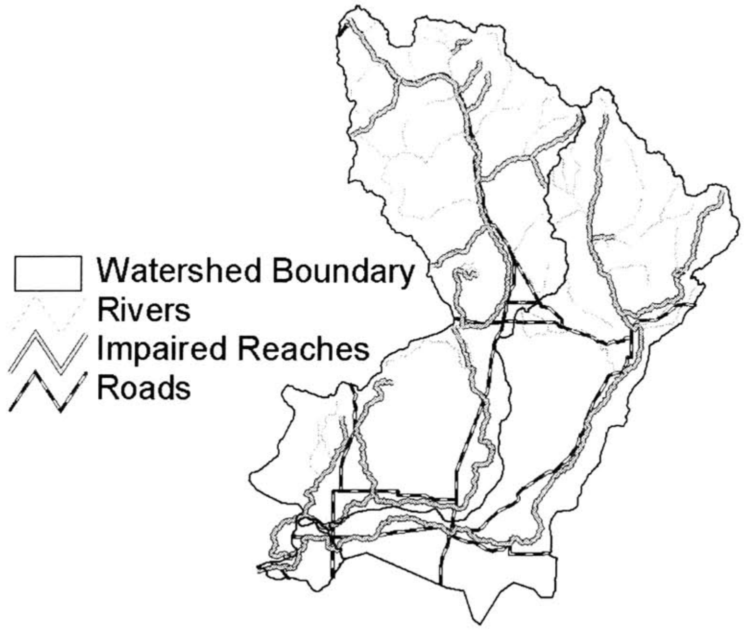


Figure 1-3: Location of 303(d) Listed River Reaches in the Wood River Valley Watershed

Studies are currently under way or already have been completed to develop TMDLs for some of these reaches with respect to the following pollutants: temperature, dissolved oxygen, bacteria, sediment load, and total phosphorus (IDEQ 2002). The Idaho Department of Environmental Quality has not yet developed nitrogen TMDLs for any reaches of the Big Wood or Little Wood Rivers.

1.2 Nitrogen in Aquatic Systems

The agricultural use of industrially-fixed nitrogen and the accelerated urbanization of society have significantly altered the historic nitrogen cycle. Industrially-fixed nitrogen fertilizers have altered the production-consumption food cycle, disrupted it from its prior steady state condition, and removed biologic control of the quantity of food that can be produced by agriculture. Unlimited food production for an increasingly urban population concentrates agricultural nitrogen in urban centers in the form of harvested agricultural products (Delwiche 1981). This disposition of nitrogen to urban centers results in a loss of nitrogen from agricultural systems and necessitates the use of more industrially-fixed nitrogen to sustain productivity. Consumed and excess agricultural nitrogen is released to ground and surface waters, where it can potentially harm human health; eventually excess nitrogen is transported to lakes and oceans and can alter nitrogen-limited ecosystems.

During recent decades, nitrogen contamination of ground and surface waters has become increasingly more common and more pervasive in the United States. Nitrate (NO_3^-) is the most commonly detected groundwater pollutant in the country (USEPA 1990), and it is also recognized as being the most widely spread of the common groundwater contaminants (Gillham and Cherry 1978). Throughout the 1990s in North America, increases in nitrate concentration in groundwater underlying agricultural areas ranged from 0.3 -2.2 mg N/L (Howarth 1998). The accumulation of nitrate in groundwater beneath cultivated land often reflects leaching of fertilizer from the surface at rates that exceed the nitrogen requirements of the underlying soil community.

A wide variety of point and distributed sources contribute to nitrate contamination. Nitrate is derived from point sources such as feedlots, waste lagoons, and wastewater treatment plants as well as from non-point sources such as agricultural runoff, septic fields, and the oxidation of organic nitrogen and ammonium in the unsaturated zone (Hendry et al. 1984). Of these sources, agricultural fertilizers and human and animal waste disposal are the most common and most responsible for contamination (Starr and Gillham 1993). Recently in many areas of the United States, escalating population growth combined with a continued reliance on individual on-site sewage disposal systems has resulted in dramatic increases in groundwater nitrate concentrations.

Nitrogen contamination of groundwater poses a threat to human health as well as to aquatic ecosystem health. Seepage of contaminated groundwater can promote the degradation of surface water quality in two ways: by contributing to eutrophication or by exceeding the recommended limit for human consumption. The maximum contaminant level (MCL) for nitrate in public drinking water supplies recommended by the U.S. Environmental Protection Agency is 10 mg N/L (USEPA 2002a). Concentrations exceeding this limit can cause methemoglobinemia in infants therefore making groundwater supplies dangerous for human consumption (Gillham and Cherry 1978). Additionally, it has been argued that eutrophication in coastal waters by increasing nitrogen loading from inland watersheds is the single most pervasive anthropogenic alteration to coastal ecosystems everywhere (GESAMP 1990).

2. BACKGROUND AND SETTING: PHYSICAL AND BIOLOGICAL CHARACTERISTICS

As identified by IDEQ, the Wood River Valley Watershed is made up of three elevation-ecological areas that reside in the counties of Blaine, Gooding, Lincoln, Camas, and Jerome in south-central Idaho (IDEQ 2002). These areas are defined as the Sawtooth National Forest (> 5,800 feet higher elevation), the Wood River Valley (4,000-5,800 feet middle elevation), and the agricultural area (< 4,000 feet lower elevation) (IDEQ 2002). The physical and biological characteristics of the watershed are related to the elevation-ecological areas (IDEQ 2002).

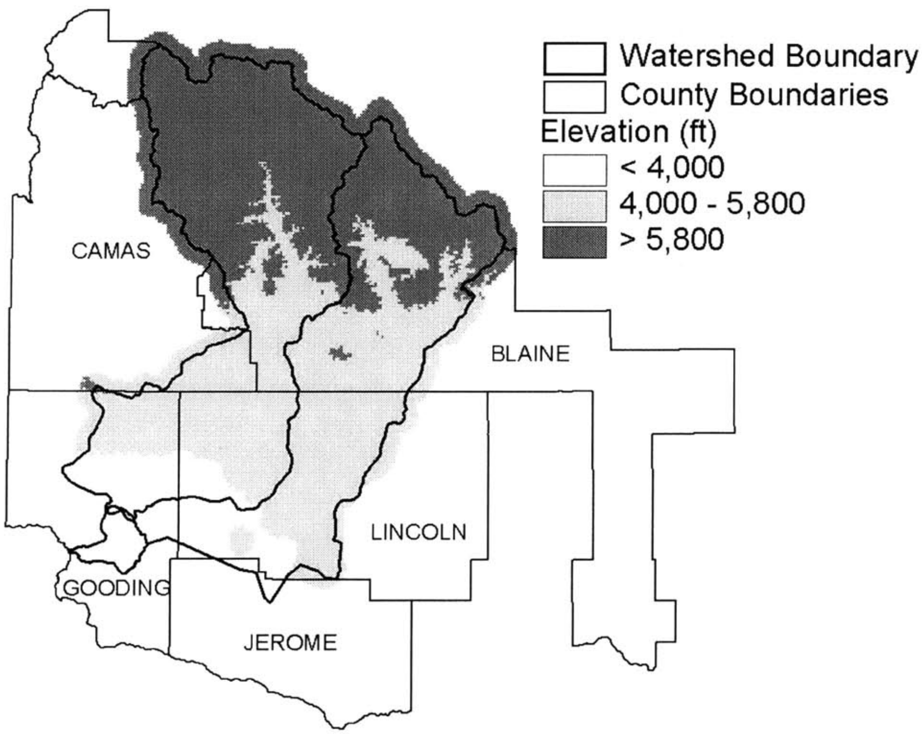


Figure 2-1: Elevation-Ecological Areas in the Wood River Valley Watershed

2.1 CLIMATE

Climate characteristics, such as precipitation, temperature, snowfall, and snow depth vary between elevation-ecological areas. The months of November through March receive the greatest precipitation, 58% of the total annual precipitation (IDEQ 2002).

Table 2-1: Climate Characteristics in the Wood River Valley Watershed

	Elevation-Ecological Area		
	Higher	***Middle***	***Lower***
Precipitation (cm)	52	34	26
Temperature Range (°C)	-6.2 - 12	-1.4 - 15	2.0 - 18
Snowfall (cm)	355	133	51
Snow Depth (cm)	327	94	17

Cloudiness and available sunlight vary as a function of season. Average available sunlight is 9.4 hours in winter, 13.3 hours in spring, 14.8 hours in summer and 11.1 hours in fall (IDEQ 2002).

2.2 HYDROGRAPHY

The Wood River Valley hydrology is dominated by the Big Wood and Little Wood Rivers running through either side of the watershed. In general, all streams and canals in the watershed discharge directly or indirectly to one of the rivers. Approximately 49% of the waterbodies in the watershed are perennial and 51% are intermittent. The rivers are predominantly perennial and are fed during periods of high runoff by numerous ephemeral, intermittent and perennial streams. Certain reaches are intermittent as a result of irrigation diversions. From the irrigation diversion at Glendale to Magic Reservoir, at least 10% of the Big Wood River is intermittent due to irrigation flow diversions. The section of the Big Wood River that is below Magic Reservoir has the potential to become intermittent during dry years due to the Richfield irrigation diversion (IDEQ 2002).

The watershed has several manmade reservoirs that are a part of the more complex network of natural and manmade waterbodies of the Wood River Valley river system. The Magic Reservoir is the largest of these reservoirs and is used for both irrigation and power generation. While the reservoir is located on the Big Wood half of the watershed, approximately 60% of the storage in Magic Reservoir is used in the middle Little Wood River area, and the remaining 40% is used on cropland in the middle Big Wood River Area. The Big Wood River Company (Shoshone, Idaho) operates the manmade canal system in the watershed. This canal system is a single management unit with storage space in American Falls Reservoir and behind the Magic Dam. The system also has natural flow rights on the Wood River system. In total, the Wood River System irrigates approximately 98,000 acres (IDEQ 2002).

2.3 Geology and Soils

The Wood River Valley watershed falls within two larger ecoregions: the Snake River Basin/High Desert and the Northern Rockies. A transitional zone exists at the middle-to-higher elevations between the two ecoregions. The Northern Rockies ecoregion is composed primarily of tertiary Challis Volcanic Rocks in the higher elevations. The Snake River Plain/High Desert ecoregion is composed of Miocene, Pliocene, and sedimentary rocks interbedded with older basalt flows in the lower elevations and valleys. Because of the geologic differences between these two ecoregions, rocks within the Wood River Valley watershed can be grouped into two general categories: 1) consolidated igneous and sedimentary rocks that make up the mountains that surround the valley floor and 2) unconsolidated fluvial and alluvial materials that make up the valley fill (IDEQ 2002).

More specifically, the Wood River Valley Watershed is underlain by three distinct water-bearing formations: the Snake River Plain Aquifer, the unconfined alluvial aquifer, and the confined alluvial aquifer (IDWA 1972). Figure 2-2 shows the distribution of the three aquifer systems within the watershed. An impermeable basement complex surrounds the aquifers.

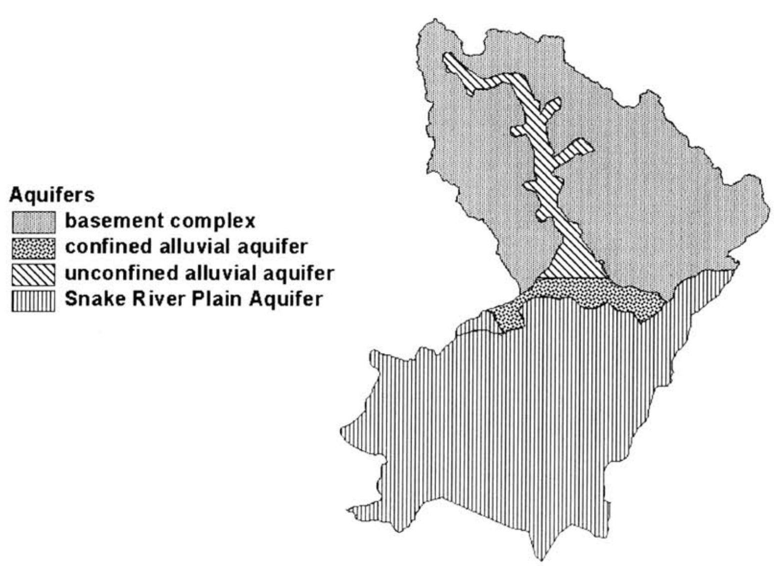


Figure 2-2: Wood River Valley Watershed Aquifer Systems

The Snake River Plain Aquifer consists of fractured basaltic rock. In the upper aquifer, the alluvial deposits were created as a result of repeated damming of the Big Wood River by volcanic flows. During periods when the river was dammed, a large lake formed in the upper Wood River Valley, and this quiescent water body allowed sediments carried by the Big Wood River to settle to the valley floor. Heavier coarse-grained sediments settled more quickly in the northern region of the lake, and fine-grained sediments settled more slowly and were carried to the southern part of the lake. Because of the differential settling of sediments that occurred in the ancient lakebeds, the northern alluvial aquifer consists of coarse-grained sediments and is an unconfined aquifer. The southern third of the alluvial aquifer contains many clay and silt lenses and is under confined conditions (Moreland 1977, IDWA 1972).

2.4 VEGETATION

In the hills, sagebrush and grasses dominate the valley vegetation, while the lowland areas consist of willows, cottonwoods, marshes, and other grasses. Vegetation on public lands can be divided into two categories: vegetation in the lower-to-middle elevations and vegetation in the middle-to-upper elevations (IDEQ 2002).

Table 2-2: Natural Vegetation in the Wood River Valley Watershed

Area	*Vegetation*
Lower-to-Middle	Sagebrush Riparian vegetation Grasslands
Middle-to-Upper	Forests Scrub-shrub vegetation Emergent (herbaceous) vegetation

2.5 FISHERIES

In general, fisheries productivity in the watershed is relatively low. In the upper watershed, the principal fish are wild rainbow trout, mountain whitefish, Wood River sculpin, and mottled sculpin. Occasionally, introduced brook trout and cutthroat trout are present in the rivers as they move out of mountain lakes that feed the Big Wood and Little Wood Rivers. In several heavily fished stream reaches, wild trout populations are supplemented with hatchery rainbow trout stocks (IDEQ 2002).

Certain threatened and endangered species are affected by the water quality in the Wood River Valley Watershed: the bald eagle, which relies on the availability of fish in streams; and several mollusk species, which are dependent on water quality. The Wood River sculpin is protected under federal regulations because it is listed as a sensitive non-salmonid species in Idaho (IDEQ 2002).

2.6 LAND USE

Land uses within the watershed include forest, agricultural lands, rangelands, and urban and suburban uses including residential and commercial development (Figure 2-3). Table 2-3 lists the different land uses within the watershed and the area of land attributed to each use. Agricultural lands include irrigated and non-irrigated cropland, pastureland, and feed-lots. Agricultural lands are rapidly being converted into development areas for larger cities. As the population in Blaine and neighboring counties grows, expansion of developed areas into forests and rangelands is also increasing (IDEQ 2002).

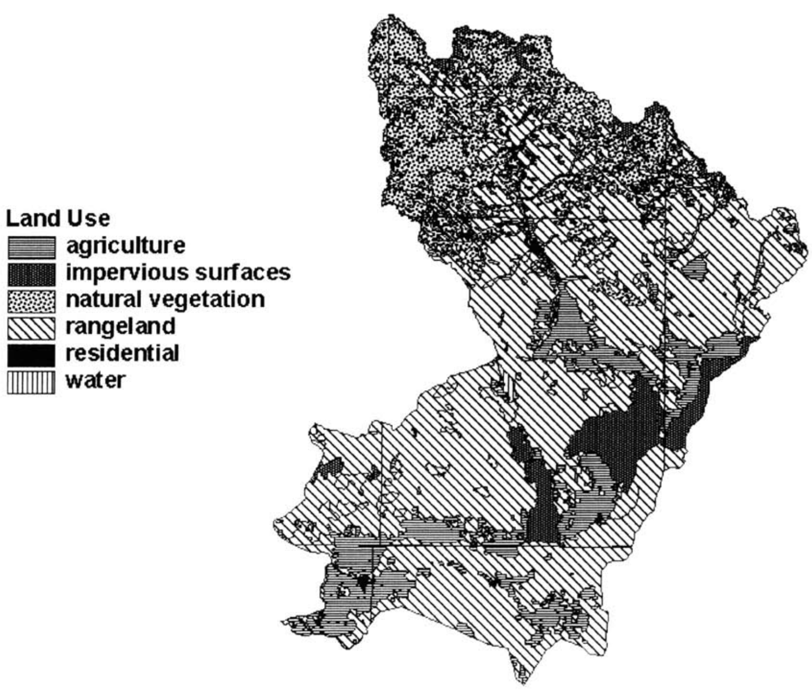


Figure 2-3: Land Uses in the Wood River Valley Watershed

Table 2-3: Land Use Areas within the Wood River Valley Watershed

Land Use	***Area (ha)***
Agriculture	84,800
Impervious surfaces	48,200
Natural vegetation	115,000
Rangeland	427,900
Residential	2,200
Water	2,200

2.7 LAND OWNERSHIP

Much of the land in the watershed, including the Sawtooth National Recreation Area, is held in the public trust and is controlled by the U.S. Bureau of Land Management (BLM) and the U.S. Forest Service. These lands are not open to residential development or farming, but many ranchers in the area have grazing rights for sheep and cattle in the BLM lands. Other land in the watershed is owned by the State of Idaho. Figure 2-4 illustrates the ownership of the watershed land, and the areas attributed to each owner are listed in Table 2-4.

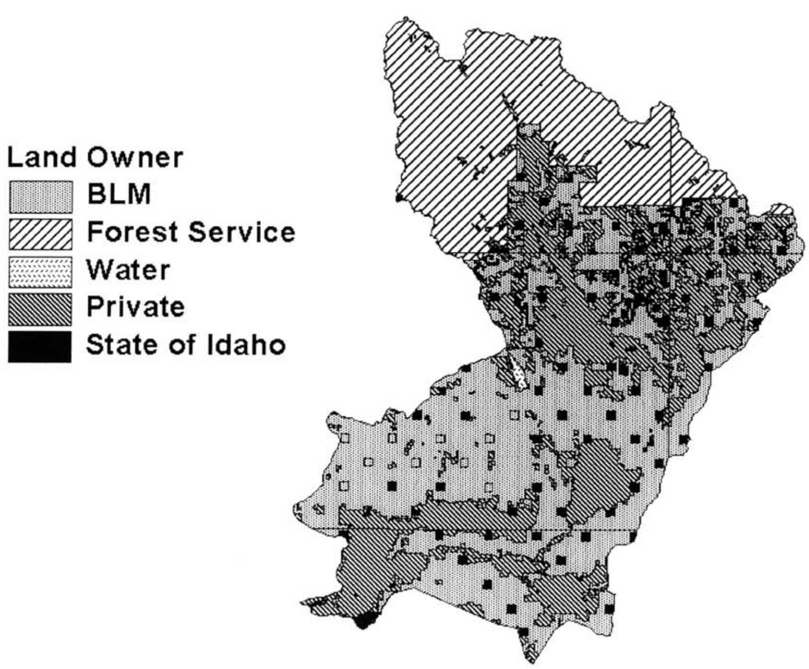


Figure 2-4: Land Ownership in the Wood River Valley Watershed

Table 2-4: Land Ownership Areas in the Wood River Valley Watershed

Land Owner	Area (ha)
U.S. Bureau of Land Management	289,300
U.S. Forest Service	165,500
Open Water	2,100
Private	195,000
State of Idaho	28,200

2.8 DEMOGRAPHICS

Despite its rural history, the region has undergone substantial population growth in recent decades (Figure 2-5). In Figure 2-5, the watershed population from 1960 to 2000 was estimated by multiplying the annual population of each county by the percentage land area of that county that lies within the watershed boundary and then summing the adjusted population values for each county.

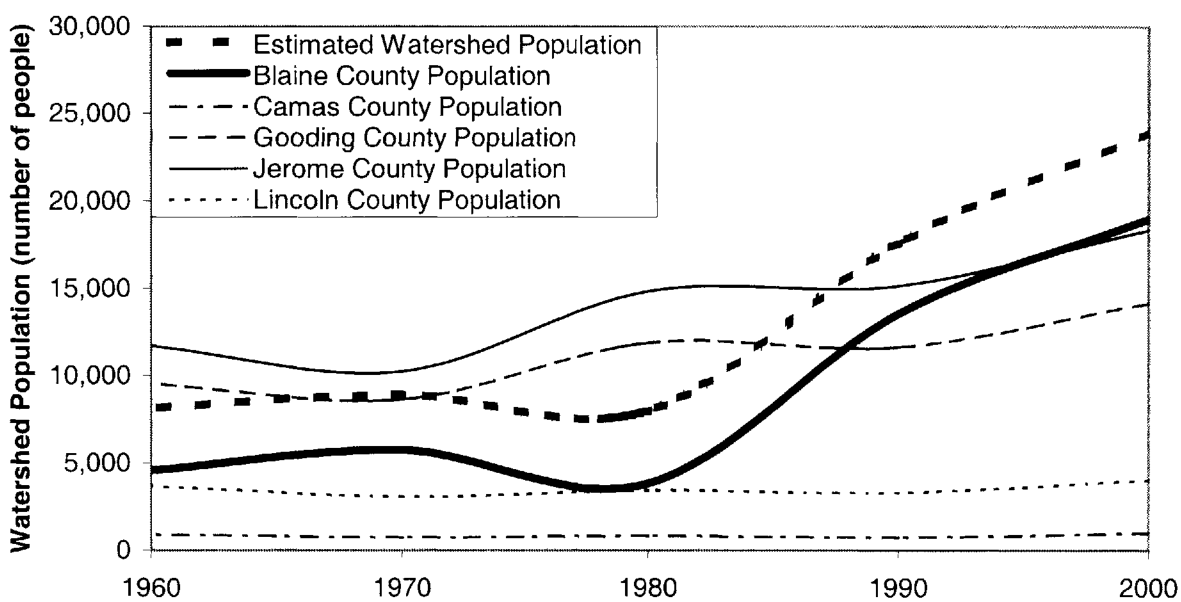


Figure 2-5: Population Growth in the Wood River Valley Area (U.S. Census Bureau 2000)

Blaine County has undergone the most rapid population growth due to growth in the towns of Ketchum, Hailey, and Bellevue. The population in Blaine County is increasing faster than populations in surrounding counties, and the majority of the watershed falls within Blaine County (60% of land area). Thus, population growth within Blaine County has the largest impact on population increases within the watershed.

3. MODELING APPROACH

We have developed a nitrogen mass-balance model, the Blaine County Evaluation and Assessment of Nitrogen Sources (BEANS) model, which identifies the nature and location of nitrogen sources within a watershed and also quantifies the relative magnitude of those sources. The model is developed for the Big Wood and Little Wood Rivers in south-central Idaho. This section describes the general structure of the model. The following section explains in detail specific parameters and calculations that are incorporated in the application of the BEANS model to the Wood River Valley Watershed.

In the BEANS model, nitrogen inputs to a watershed are distributed spatially over different land uses in a geographic information system (GIS). Use of the GIS allows the model to be spatially-explicit in its identification of nitrogen sources within the watershed. Instead of simply identifying which type of nitrogen sources (i.e. atmospheric deposition, wastewater, agricultural sources) are significant contributors to the overall watershed nitrogen load, this spatially-explicit model identifies the location of specific land areas that are responsible for introducing significant masses of nitrogen into the watershed.

A spatially-explicit nitrogen mass-balance model is a useful tool in evaluating the relative magnitude of different nitrogen sources within a watershed because the model defines a precise geographic location for each source. A source's location is important because it can influence how and when nitrogen inputs from that source will impact a receiving water body like a river. Both the distance of a source from a receiving water body and the hydrologic properties of the intervening groundwater aquifer determine how long it takes for nitrogen inputs to a given land area to travel through the groundwater and reach a surface water body. Figure 3-1 illustrates the idea of groundwater travel time.

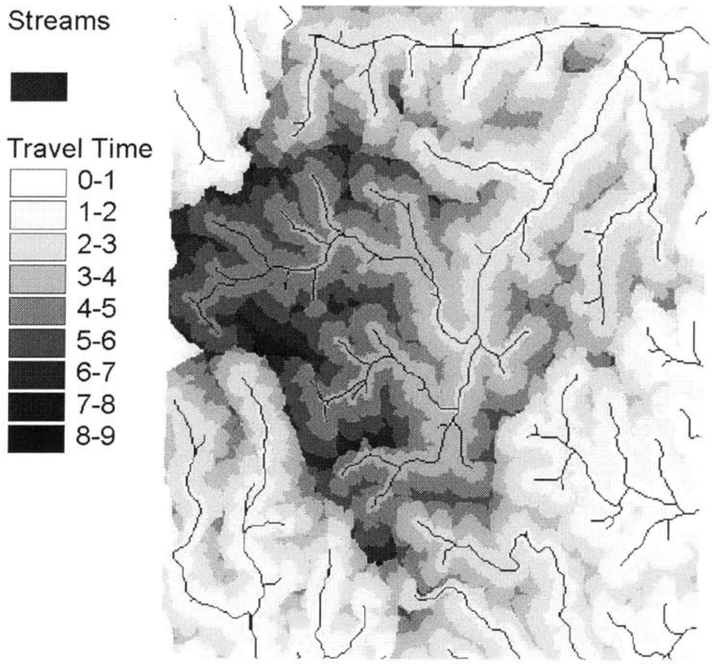


Figure 3-1: Example of Groundwater Travel Time. This figure illustrates travel time to stream channels in Ashfield, MA

In Figure 3-1, darker colors illustrate longer travel time to rivers and lighter colors illustrate shorter travel times. Because groundwater travel time from a given land area to the receiving water body can vary within the watershed, nitrogen loading to some land areas will affect water quality in the receiving water body sooner than nitrogen loads from more distant land areas or from regions with slower groundwater velocities. In addition, nitrogen losses during groundwater transport (such as denitrification) can depend on groundwater travel time.

To capture the variability in watershed nitrogen loading across both spatial and temporal scales, the BEANS model combines a geographically referenced watershed nitrogen budget with delineated groundwater travel time bands for the defined watershed land area. Using these two data sets in combination, we can identify specific land areas within a watershed that may pose a threat to receiving water quality both because of the amount of nitrogen they introduce into the watershed and because of the speed with which that nitrogen is able to impact the receiving water body.

3.1 CONCEPTUAL MODEL

3.1.1 Groundwater Travel Time

We used a method similar to that described by Brawley et al. (2000) to incorporate the effects of groundwater travel through subsurface aquifers into our nitrogen mass-balance calculations. This method, first, defines how long it will take for groundwater to travel from any given point in the watershed to the closest river channel or contributing tributary stream and, then, to divide the watershed into zones of similar groundwater travel time. In order to calculate groundwater travel time, we must know 1) the distance from any point in the watershed to the closest receiving water body and 2) the velocity with which groundwater will travel from the land surface to the nearest receiving water body.

Darcy's Law allows us to calculate groundwater velocities within the watershed.

$$Q = -KA\frac{dh}{dL} \qquad \text{(eqn. 3-1)}$$

Q is the flow through a porous medium and has units of $\left(\frac{length^3}{time}\right)$. K is the hydraulic conductivity $\left(\frac{length}{time}\right)$. A is the cross-sectional area through which the groundwater flows $\left(length^2\right)$. The hydraulic gradient, $\frac{dh}{dL}$, is the ratio of the change in hydraulic head $\left(dh = h_2 - h_1\right)$ to the distance the groundwater has traveled $\left(dL = L_2 - L_1\right)$. Hydraulic gradient is a unitless value $\left(\frac{length}{length}\right)$.

Darcy's law can be rewritten to express the specific discharge or flow per unit area:

$$q = \frac{Q}{A} = -K\frac{dh}{dL} \qquad \text{(eqn. 3-2)}$$

The specific discharge, q, has units of velocity $\left(\frac{length}{time}\right)$ but is not the true velocity of groundwater moving through the aquifer. Groundwater actually moves at a faster velocity than the specific discharge suggests because groundwater is only able to travel through the volume of the aquifer that is open space (pores). Much of the cross-sectional area of an aquifer is filled with the solid grains of the aquifer material. The fraction of the aquifer volume that is made up of pores is referred to as the aquifer porosity, n:

$$n = \text{volume of pores / total aquifer volume} \qquad \text{(eqn. 3-3)}$$

We can calculate the actual velocity of groundwater with the following expression:

$$v = -\frac{K}{n}\frac{dh}{dL} \qquad \text{(eqn. 3-4)}$$

$$v = \frac{q}{n} \qquad \text{(eqn. 3-5)}$$

Knowing groundwater velocities within the watershed, groundwater travel time can be calculated from the following expression:

$$t = \frac{d}{v} \qquad \text{(eqn. 3-6)}$$

This method requires information about the following watershed parameters in order to calculate groundwater travel time: hydraulic conductivity (K), hydraulic head $\left(\frac{dh}{dL}\right)$, porosity ($n$), and distance to the receiving water body (d).

After calculating groundwater travel times throughout the watershed, we divide the land areas within the watershed into bands of increasing groundwater travel time. The first travel-time band includes all the overlying land areas that will discharge into the closest river or stream channel within 1 yr. The next band includes all areas that will take between 1 and 2 yr to discharge into the river, and so on. In the nitrogen mass-balance model, land use patterns within each travel time band determine the mass of nitrogen that is added to the watershed within that band. The discharge of this nitrogen into the receiving water body is then delayed in accordance with its band's calculated travel time.

3.1.2 Nitrogen Mass-Balance Calculations

The BEANS model uses an algorithm similar to that used in the Waquiot Bay Land Margin Ecosystems Research project (WBLMER) to calculate total nitrogen (TN) additions to the watershed (Valiela et al. 1997). The WBLMER model "provides a description of how nitrogen transport through adjoining landscape units in the coastal zone, and nitrogen transformations within the units result in marked changes in mass balances of externally delivered nitrogen" (Valiela et al. 1997). Important differences between the BEANS model and the WBLMER model are 1) the BEANS model is designed for a mountainous, inland watershed rather than a coastal watershed and 2) the BEANS model is designed for a watershed with significant agricultural activity while the WBLMER algorithm includes only limited nitrogen inputs from agricultural sources.

Nitrogen inputs in the BEANS model include atmospheric deposition, agriculture and lawn fertilizer, fixation of atmospheric nitrogen (N_2) by plants, animal waste, and domestic (septic) and municipal (sewer) wastewater. The BEANS model considers several processes through which nitrogen can be removed from groundwater including uptake into plants, adsorption to soils, volatilization of ammonia, and denitrification. In the BEANS model, nitrogen inputs and losses are applied to different land areas in a watershed in accordance with land use types; consequently, the types of land uses that exist within the watershed determine the mass of nitrogen entering the watershed and the extent to which that nitrogen enters the aquifer. The BEANS model considers the following land use categories: cropland and pasture, feed-lots, natural vegetation, rangeland, residential areas, fresh water ponds and reservoirs, and impervious surfaces.

3.2 NITROGEN INPUTS AND LOSSES

Because of the variation in land use types within a watershed, all areas of the watershed do not receive nitrogen inputs from all the potential nitrogen sources. For example, a residential area would not receive nitrogen inputs from beef or dairy cattle waste, and similarly, a forested area of land would not receive nitrogen inputs from domestic lawn fertilizer. Table 3-1 catalogues which nitrogen inputs are applied to which land uses in the BEANS model.

Table 3-1: Nitrogen Inputs Applied to Various Land Use Types

	Non-Point Sources							**Point Sources**	
	Atmospheric Deposition	***Fertilizer***	***Fixation***	***Animal Waste***				***Wastewater***	
				Cattle	*Hogs*	*Sheep*	*Chickens*	*Septic*	*Sewage*
Cropland and Pasture	x	x	x	x		x			
Feed-Lots	x			x	x		x		
Natural Vegetation	x								
Rangeland	x			x		x			
Residential Areas	x	x						x	x
Ponds and Reservoirs	x								
Impervious Surfaces	x								

3.2.1 Atmospheric Deposition

Atmospheric deposition can contribute significant amounts of nitrogen to watersheds in North America. As shown in Figure 3-2, the magnitude of this contribution can vary dramatically between different regions of the United States; therefore, calculations of nitrogen input to a watershed from atmospheric deposition must be based on local deposition data.

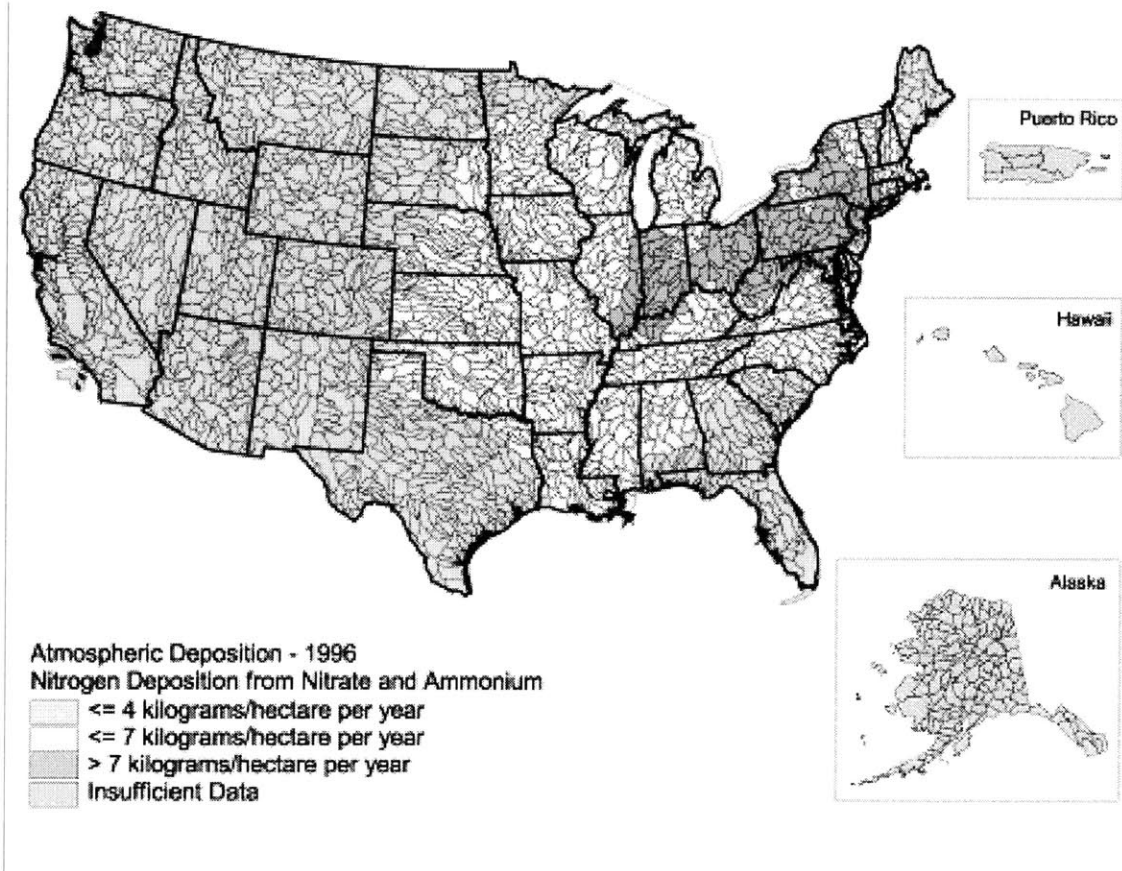


Figure 3-2: Regional Variation in Atmospheric Deposition of Nitrogen. (Source: U.S. Environmental Protection Agency. Index of Watershed Indicators. Retrieved: March 2, 2003 from http://www.epa.gov/iwi/1999april/iii17_usmap.html).

Calculations of atmospheric deposition to a watershed should include input from both wet and dry deposition. Wet deposition accounts for nitrogen that enters a watershed through precipitation (NO_x dissolved in rain and snow). Dry deposition is more difficult to measure because it is nitrogen introduced to watersheds through accumulation of atmospheric nitrogen particles that settle onto the land surface and through adsorption of NO_x gas and ammonia by plant leaves. The National Atmospheric Deposition Program (NADP) collects wet deposition data at sampling locations around the United States.

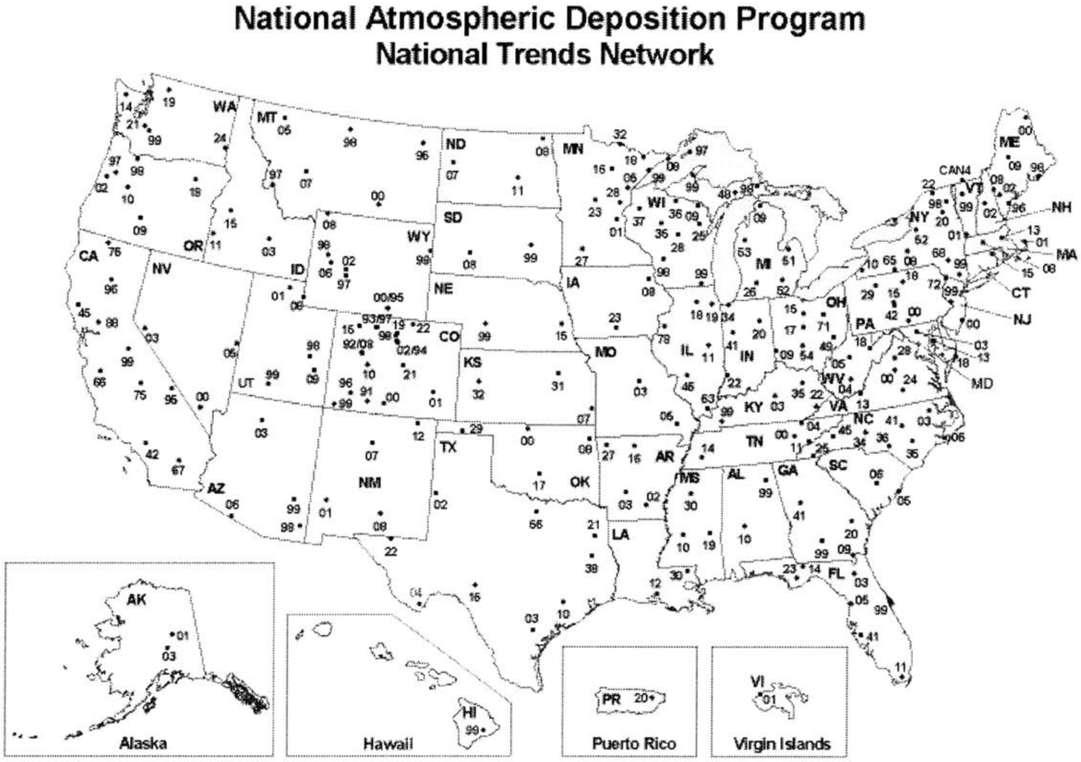


Figure 3-3: NADP Monitoring Locations. (Source: National Atmospheric Deposition Program. Retreived: March 2, 2003 from http://nadp.sws.uiuc.edu/networks.html#adp)

Valiela et al. (1997) reviewed literature data that were available for both wet and dry deposition and found that the ratio of dry deposition to wet deposition was indistinguishable from 1; thus, in the BEANS model we calculate total atmospheric deposition as two times the wet deposition values measured by NADP.

Because variation in the magnitude of atmospheric deposition occurs on a regional rather than local scale, the magnitude of atmospherically deposited nitrogen will be the same across all areas of a watershed; however, the fate of that deposited nitrogen once it enters a land surface depends on whether that land parcel is covered by cropland, pasture, feed lots, natural vegetation, rangeland, residential land, fresh water bodies, or impervious surfaces.

Deposition to cropland and pasture: Nitrogen deposited onto cropland and pasture has the potential to be taken up and stored in crops or adsorbed to soil particles. According to

Valiela et al. (1997), 62% of atmospherically derived nitrogen is retained in agricultural land by these mechanisms, allowing only 38% of this nitrogen to travel through the groundwater towards a receiving water body.

Deposition to feed-lots: The BEANS model assumes that nitrogen deposited onto feed lots does not have the potential to be retained by plants because the density of animals in these facilities does not allow for the persistence of a significant plant community; consequently, the 62% retention of nitrogen by plants and soils that is applied to agricultural land is an overestimate of retention in feed lots. Not knowing the exact contribution of plant uptake versus soil adsorption, the BEANS model assumes that each mechanism is responsible for half (31%) of the observed nitrogen retention, allowing 69% of nitrogen deposited to feed lots to travel through the groundwater towards a receiving water body.

Deposition to natural vegetation: Natural vegetation includes land areas that are covered in forests and tundra. Valiela et al. (1997) report that 65% of atmospherically deposited nitrogen is retained in naturally vegetated parcels; consequently, 35% of nitrogen deposited to these land areas is able to travel through the groundwater toward a receiving water body.

Deposition to rangeland: The BEANS model assumes that the vegetation on rangeland is more similar to that on naturally vegetated parcels than it is to that on agricultural parcels. The 65% plant retention is applied to rangeland parcels, allowing 35% of nitrogen deposited to rangeland to travel through the groundwater toward a receiving water body.

Deposition to residential land: Deposition to residential land is different than deposition to the land uses discussed previously because the issue of the fate of nitrogen that is deposited onto roofs and driveways is introduced in residential areas. While the roofs and driveways themselves are impervious and would have 0% retention, the BEANS model assumes that this deposition runs off onto adjacent lawns and then is subject to retention by uptake into and storage by grasses. The BEANS model assumes grasses are more similar to agricultural vegetation than they are to forest and rangeland vegetation, so the 62% plant retention is applied to residential areas, and 38% of this atmospherically deposited nitrogen is able to travel through the groundwater toward a receiving water body.

Deposition to lakes and reservoirs: As discussed by Valiela et al. (1997), nitrogen retention in ponds and lakes is often approximately 56%; consequently, the BEANS model assumes that 44% of atmospheric deposition that falls onto ponds, lakes, and reservoirs contributes to the net nitrogen loading to the watershed surface water bodies.

Deposition to impervious surfaces: The category of impervious surfaces includes all land uses that are covered with a material that does not allow for direct transfer of water from the land surface into the subsurface aquifer. This is a broad category including exposed rock, commercial land, and industrial land. These land uses are similar to the roofs and driveways in the residential category, but differ in that they are not adjacent to vegetated areas like lawns. Atmospheric deposition to impervious surfaces likely runs off the surface into storm water drains that either empty into the subsurface or are transported to wastewater treatment plants. In either case, 0% of the nitrogen deposited on an impervious land area is retained within that land area, and 100% of the nitrogen is available for transport into a receiving water body.

3.2.2 Fertilizer Applications

Unlike atmospheric deposition, nitrogen from fertilizer applies only to agricultural and residential land uses.

Fertilizer inputs to agricultural lands: Because fertilizer application rates can vary substantially depending upon the type of crop being grown, understanding the types of crops and where they are grown within a watershed is critical to calculating the agricultural nitrogen load across the watershed. Fertilizer application rates can also vary between regions and between growers, so obtaining the most locally-specific data possible is important.

The mass of nitrogen applied to an agricultural land parcel depends on the fertilizer application rate for the specific crop and the area of land that is contained within the parcel. In the process of applying fertilizers to the land surface, 39% of the nitrogen is lost as gas (Boyer et al. 2002), allowing 61% of fertilizer nitrogen to percolate into the soil. This remaining 61% of the fertilizer nitrogen is then subject to the 62% plant retention factor that was mentioned in the atmospheric deposition discussion; consequently only 23% of the initial fertilizer nitrogen that was applied is available for transport through the groundwater to a receiving water body.

Fertilizer inputs to residential lands: Fertilizers are routinely used on residential lawns. The mass of nitrogen entering the watershed from this source is calculated as the percentage of residential land used as lawn multiplied by the lawn fertilizer application rate that is common to the watershed area. This fertilizer is again subject to 39% loss as gas followed by a 62% retention in plants, so 23% of this nitrogen is allowed to travel through the groundwater toward a receiving water body.

3.2.3 Nitrogen Fixation

Nitrogen fixation generally occurs exclusively on agricultural land areas where nitrogen-fixing crops like legumes are grown. On some occasions, certain grassy areas can also fix nitrogen if the grass contains nitrogen-fixing species like clover. To calculate the mass of nitrogen introduced to the watershed through fixation, the nitrogen fixation rate and the area of each nitrogen-fixing crop (or grass) present in the watershed must be determined. Applying these fixation rates to the area of land dedicated to growing each respective nitrogen-fixing crop yields the mass of nitrogen that is fixed into plants each year. This nitrogen is held in the plants until the crop is rotated out and the plant material is tilled into the soil. Nitrogen from fixation does not have the potential to percolate into groundwater and travel to a receiving water body until this rotation occurs, but once this nitrogen is tilled into the soil, it has the same potential to be taken up into plants as atmospherically deposited or fertilizer nitrogen. On agricultural lands, 62% of nitrogen is retained in plants and soils, so 38% of fixed nitrogen that is rotated out of the crop rotation will enter the groundwater in a given year.

3.2.4 Animal Waste

Nitrogen from animal waste is applied to pasture, rangeland, and feed-lots. To calculate the mass of nitrogen entering a watershed from animal waste, the number of animals within the watershed and the number of animals on each type of land use must first be determined. This can be done through direct counts, through estimations from Agricultural Census statistics, or through estimations based on animal densities in pastures, rangeland, and feed-lots. A direct count would be the most accurate method, though this method is often not feasible. A combination of the three methods mentioned above allows for the use of detailed counts where they are available and uses estimation techniques for the remaining areas.

Animal waste inputs to pasture: Once the number of animals on pasture is determined, the mass of nitrogen they introduce into the watershed is calculated by applying an animal-specific nitrogen excretion rate to each type of animal. Initially, some of the nitrogen in this waste will be lost to the atmosphere as gas. These gas loss percentages are also animal specific. The nitrogen that is not lost as gas percolates into the soil where it has the potential to be taken up into plants. In pasture, 62% of nitrogen is retained in plants and soils, so 38% of animal waste nitrogen that is not lost as gas is able to travel in groundwater toward a receiving water body.

Animal waste inputs to rangeland: The mass of nitrogen introduced to a watershed from animals on rangeland is calculated by the method described above for pasture using the number of animals, animal-specific nitrogen excretion rates, and animal waste gas loss percentages; however, in the case of rangeland animal waste, 65% of nitrogen that percolates into the soil has the potential to be taken up in to plants, leaving 35% of this nitrogen to be transported by groundwater.

Animal waste inputs to feed-lots: The feed-lot calculation is similar to those for pasture and rangeland except for the percentage of nitrogen that is retained on the site. In the case of feed-lot animal waste, nitrogen inputs are not subject to uptake into plants but only to retention by soils (31%), so 69% of this nitrogen is able to travel in the groundwater toward a receiving water body.

3.2.5 Losses in Vadose Zone and Aquifer: Non-Point Sources

Nitrogen from atmospheric deposition, fertilizer applications, nitrogen fixation, and animal waste enter watersheds in a non-point fashion. These sources are applied to the land surface, percolate through surface soils, and enter the vadose zone diffusely. Because these four sources all enter the subsurface in a dispersed manner, they are transported through the vadose zone and aquifer similarly and undergo indistinguishable transformation and loss processes (such as nitrification and denitrification) during transport. This section describes these additional losses that occur once nitrogen from non-point sources enters the vadose zone and aquifer.

The fate of nitrogen during travel through vadose zones and aquifers is perhaps the least well-understood parameter in the BEANS model. Valiela et al. (1997) report that in the Waquoit Bay watershed 61% of nitrogen that is able to percolate into the unsaturated vadose zone from forested and cultivated land areas is lost during transport through this zone. While it is likely that vadose zone losses vary significantly between watersheds as well as within watersheds between land uses, the 61% used in the WBLMER model is our best estimate for calculating losses in the vadose zone. The BEANS model allows 39% of non-point source nitrogen that has percolated into the vadose zone to continue to travel through into the saturated aquifer.

Several studies suggest that denitrification or other losses also occur during transport through aquifers. Nitrate and dissolved organic matter decrease downgradient in watersheds in Maryland (McFarland 1989), Ontario (Trudell et al. 1986, Gillham 1991), and Wisconsin (Cherkauer et al. 1992). Losses of 20-35% (Valiela and Costa 1988) and 62% (Cherkauer et al. 1992) were calculated in groundwater travel to Buttermilk Bay, Massachusetts, and Lake Michigan respectively. In the WBLMER model, Valiela et al. (1997) use an aquifer denitrification loss percentage of 35%.

Additionally, denitrification rates have been shown to vary depending on nitrate concentration. Denitrification losses are greater in areas with elevated nitrate concentration (Pabich (submitted), Howarth et al. 1996), so denitrification losses from nitrogen sources should increase as the magnitude of the source increases. Pabich et al. (submitted) have developed an empirical model to predict denitrification rates based on groundwater nitrate concentrations. Denitrification rates are modeled using a Michaelis-Menten type substrate-utilization expression (Pabich (submitted)). Because denitrification loss rates vary between watersheds and even within watersheds, local data on denitrification losses should be used whenever possible and the magnitude of these losses should represent differences in the initial magnitude of nitrogen sources. Depending on the nature of available data, local denitrification losses can be introduced to the BEANS model as direct, local denitrification rate measurements or as estimated rate measurements calculated using local nitrate concentration data and the model described by Pabich et al. (submitted).

3.2.6 Septic System Inputs and Losses

Nitrogen from septic systems does not percolate diffusely through the land surface but rather is input directly into the vadose zone where it is transported through the aquifer in distinct plumes. Because of this plume transport, septic system nitrogen behaves more like a point source than a non-point source and is subject to different loss rates than non-point inputs.

Like the WBLMER model, the BEANS model considers inputs from on-site wastewater disposal systems of conventional design (Kaplan 1991). These systems consist of a septic holding tank designed to accomplish sedimentation and microbial degradation of organic matter. Wastewater effluent overflows out of this holding tank into a leaching field that allows for effluent dispersal into surrounding unsaturated soils.

Nitrogen inputs to septic systems: To calculate the contribution of septic system nitrogen to receiving waters, we first must estimate how many people are living in the watershed and using septic systems. The easiest way to estimate this number is to determine the number of people not on septic (that is, on sewers instead) and then subtract this value from the total watershed population. A per capita nitrogen release rate is then applied to the population on septic to obtain the mass of nitrogen that is input into septic systems.

Nitrogen losses in septic systems: The assumption is often made that approximately 50% of nitrogen is retained in septic systems themselves. Valiela et al. (1997) conducted a comprehensive review of the literature concerning septic tank nitrogen retention and estimated that 40% of septic tank nitrogen is retained in the septic tank itself. The BEANS model uses this 40% retention rate and allows 60% of nitrogen entering septic tanks to overflow into leaching fields.

Nitrogen losses in plumes: Nitrogen leaving septic system leach fields is subject to losses both in plumes and in aquifers. Losses in plumes are distinguished from losses in the aquifer because of higher plume concentrations of nitrogen, which relates to the differences between point and non-point sources. In plumes, septic nitrogen still behaves as a point source with elevated concentrations and direct flow paths. Because of these elevated concentrations, there exists the possibility for greater denitrification rates than those that are observed for diffusely introduced non-point sources. Valiela et al. (1997) define a septic system plume as persisting for a distance approximately 200 m downgradient of a septic tank and estimate that 34% of leaching field nitrogen is lost in plumes.

Nitrogen losses in the aquifer: After the septic system plume travels 200 m from its source, it has undergone enough dispersion to render it indistinguishable from a non-point source (Valiela et al. 1997). Because of this transition from point-like to non-point-like source, an appropriate denitrification loss is applied to septic nitrogen after it has traveled a distance of 200 m from its source. The dependence of septic nitrogen losses on distance from the septic tank itself means that septic tanks that are closer than 200 m to a receiving water body will have a greater impact on surface water quality than will more distant septic systems. The Blaine County Zoning Code requires that three hundred feet (300') be "the minimum separation between any drain field site and a natural stream, spring or lake" (Blaine County 2002). Because of this set back requirement, the majority of septic system leach fields in the watershed are greater than 200m from the nearest surface water body.

3.2.7 Sewage Treatment Plant Loading

Sewage treatment plant effluents are often discharged directly into receiving water bodies and, therefore, are not subject to losses in the vadose zone and aquifer. Because we do not need to consider transport through the aquifer, masses of nitrogen introduced to the watershed from sewage treatment plants can be calculated directly from measurements of nitrogen concentrations in the sewage treatment plant effluent and flow rates of effluent out of the plant. These data can be obtained from the wastewater treatment plant operators. The important distinction is made between total nitrogen and just nitrate or ammonia values. Total nitrogen is used for sewage treatment plant load calculations. Generally, total nitrogen will be the sum of nitrate, ammonia, and organic nitrogen concentrations or the sum of nitrate and Total Kjeldahl Nitrogen (TKN) concentrations depending on the types of measurements routinely taken by the sewage treatment plants.

3.2.8 In-stream Denitrification

Denitrification losses can occur during both aquifer transport and in-stream transport. Howarth et al. (1996) report that average in-stream losses range from 10% to 20% but can be as great as 45% in rivers with very high nitrogen loads that promote the existence of anoxic conditions. In rivers that fall within the 10%-20% loss range, greater losses are observed in rivers with flows less than 28.3 m^3/s (Smith et al. 1997).

4. APPLICATION OF BEANS MODEL TO WOOD RIVER VALLEY WATERSHED

4.1 DELINEATION OF GROUNDWATER TRAVEL TIME BANDS

The BEANS model uses Darcy's Law as the primary method for determining groundwater travel time within an aquifer. In the Wood River Valley Watershed, there are three distinct aquifer types: an unconfined alluvial aquifer in the northern valley, a confined alluvial aquifer in the center of the watershed, and the Snake River Plain aquifer over the southern half of the watershed. Darcy's Law is used to determine groundwater velocities in these three aquifers; however, the mountain area that surrounds the northern unconfined aquifer is composed of an essentially impermeable basement complex.

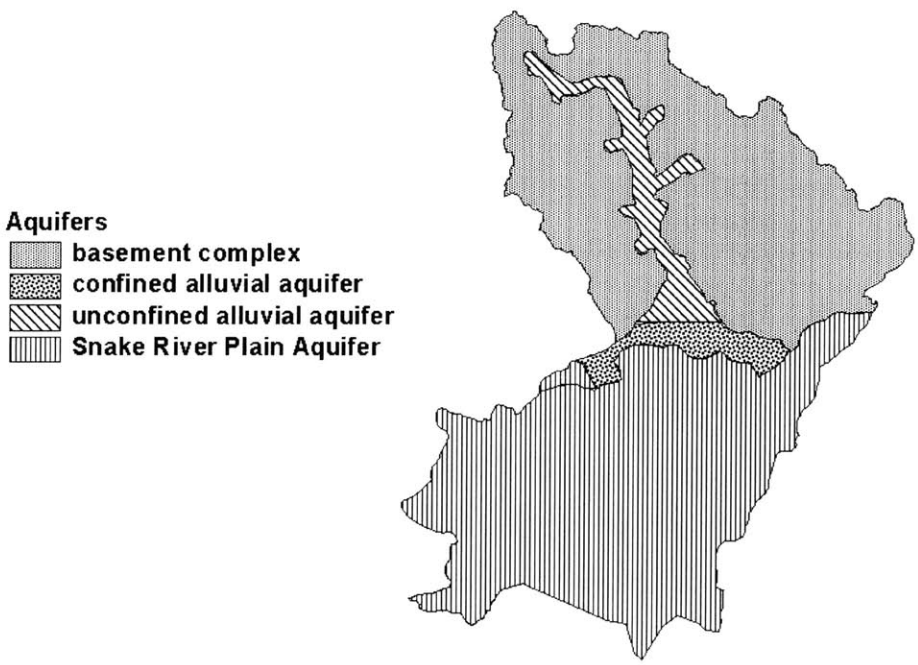


Figure 4-1: Wood River Valley Watershed Aquifer Systems

The hydrogeology of the valley suggests that this mountainous area does not have a substantial subsurface aquifer, so groundwater transport out of this area is extremely limited or nonexistent (IDWA 1972). Additionally, there is no connection between the mountainous area and the alluvial aquifers at the center of the valley (IDWA 1972). Because of the hydrogeologic characteristics of the basement complex area, the BEANS model assumes that transport of water and nitrogen from the mountains occurs through surface water run-off rather than through groundwater transport.

4.1.1 Mountain Run-off Velocities

We used the Upland Method of Estimating Time of Concentration (UMETC) to estimate run off velocities from the impervious basement area (Kent 1972). This method determines velocity as a function of land slope and land cover. Land slope was calculated from a digital elevation model (DEM) in the GIS. The DEM was originally obtained from the USEPA BASINS (Better Assessment Science Integrating Point and Nonpoint Sources) spatial data database (USEPA/BASINS 2002).

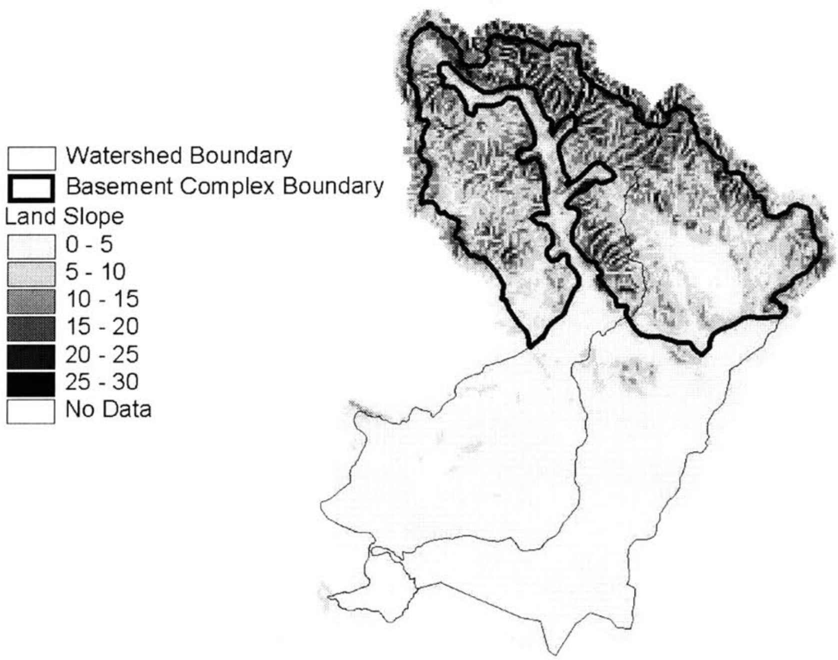


Figure 4-2: Land Slope within the Wood River Valley Watershed

The land cover categories that the UMETC method considers are: forest with heavy ground litter and meadow, fallow or minimum tillage cultivation, short grass pasture and lawns, nearly bare ground, and paved areas. The land uses in the mountain area are evergreen forest, rangeland, and tundra. To incorporate this land use information, the BEANS model uses the average of the velocities for forested and nearly bare ground land cover types (Table 4-1).

Table 4-1: Surface Water Run-off Velocities Used in the BEANS Model

Slope	***Forest Velocity (m/day)***	***Bare Ground Velocity (m/day)***	***Average Velocity (m/day)***
5%	14,500	60,600	**37,500**
10%	21,100	92,200	**56,600**
15%	26,300	113,000	**69,800**
20%	29,000	119,000	**73,700**
25%	36,900	137,000	**86,900**

After assigning these velocities to different land areas based on land slope, we calculate run-off travel time by dividing distance to the closest river or stream by run-off velocity.

$$\text{run - off travel time} = \frac{\text{distance}}{\text{run - off velocity}} \qquad \text{(eqn. 4-1)}$$

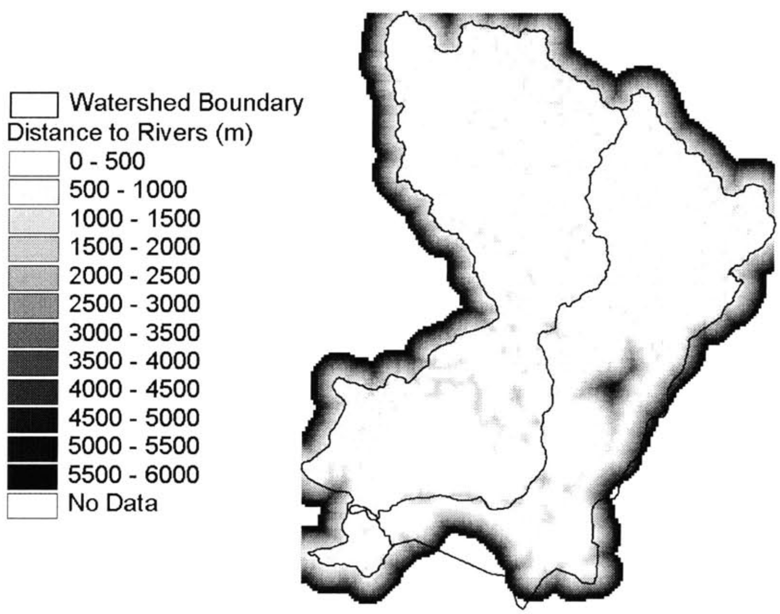


Figure 4-3: Distance from Any Area in the Wood River Valley Watershed to the Closest River or Stream Channel

Distances to the river in the mountain area are small relative to the run-off velocities: distances of 0 to 1,500 m compared to velocities of 37,000 to 87,000 m/d. As a result of fast velocities over short distances, the entire mountain area falls within the 1-year travel time band.

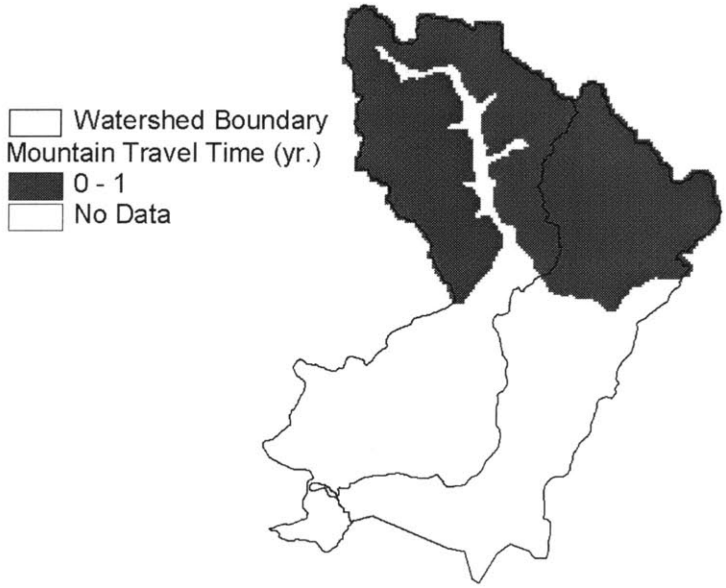


Figure 4-4: Run-off Travel Time for the Mountain Area of the Wood River Valley Watershed

4.1.2 Valley Groundwater Velocities

Travel time for the rest of the watershed is defined using Darcy's Law. In order to use Darcy's Law to calculate groundwater velocities, we must know aquifer hydraulic conductivity, K, hydraulic gradient, $\frac{dh}{dL}$, and aquifer porosity, n.

$$v = -\frac{K}{n}\frac{dh}{dL} \qquad \text{(eqn. 4-2)}$$

Hydraulic Conductivity: The Idaho Department of Water Resources (IDWR) has measured aquifer transmissivity in the unconfined and confined alluvial aquifers. These transmissivity values range from 1,000 to 30,000 m^2/day (IDWA 1972, IDWR 1977). For the Snake River Plane aquifer, we used transmissivity values that were estimated in a groundwater model (AWRA 1987). These values ranged from 100 to 95,000 m^2/day (AWRA 1987). Transmissivity in the Snake River Plane aquifer exhibit greater variability than those in the alluvial aquifers because groundwater transport in this aquifer occurs by means of fractured flow through Snake River basalts. Spatial variation in the density of these fractures creates variability in aquifer transmissivity.

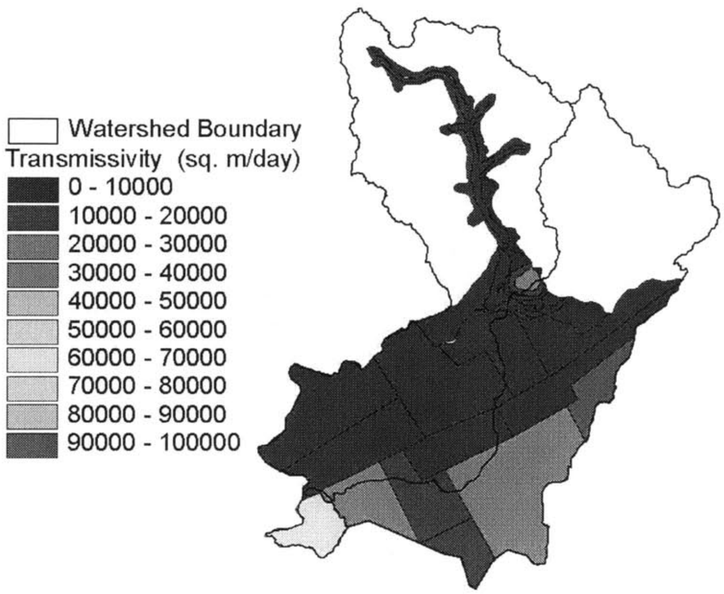


Figure 4-5: Distribution of Aquifer Transmissivity in the Wood River Valley Watershed

Aquifer transmissivity is hydraulic conductivity multiplied by aquifer saturated thickness, *b*.

$$T = Kb \qquad \text{(eqn. 4-3)}$$

Average aquifer thicknesses for the unconfined and confined alluvial aquifers are 45 and 30 m respectively (IDWA 1972). The Snake River Plain aquifer is significantly thicker with an average depth of about 100 m (McLeana and Johnson eds. 1987).

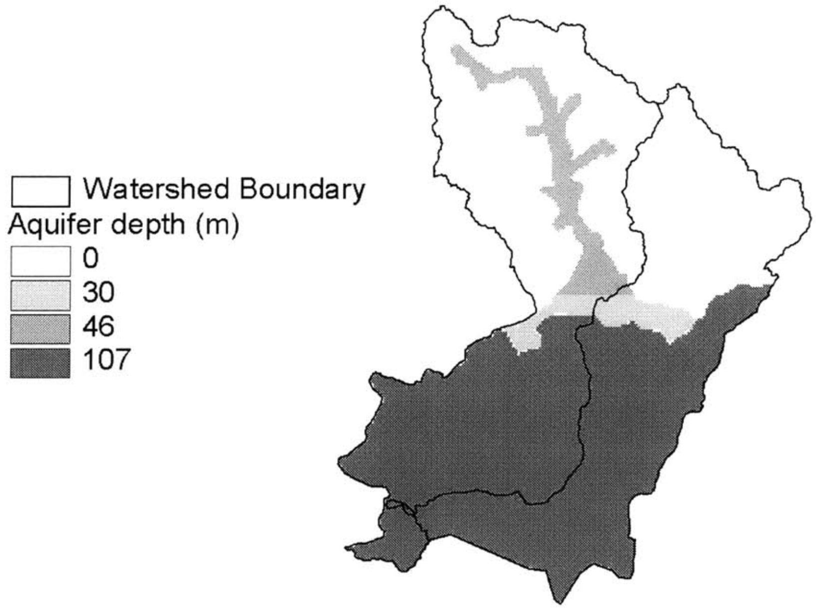


Figure 4-6: Aquifer Thicknesses in the Wood River Valley Watershed

By dividing aquifer transmissivity by depth, we obtain spatially distributed aquifer conductivity, K, across all three aquifers. In Figure 4-7, darker areas show regions with low hydraulic conductivity and lighter areas show regions with high hydraulic conductivity.

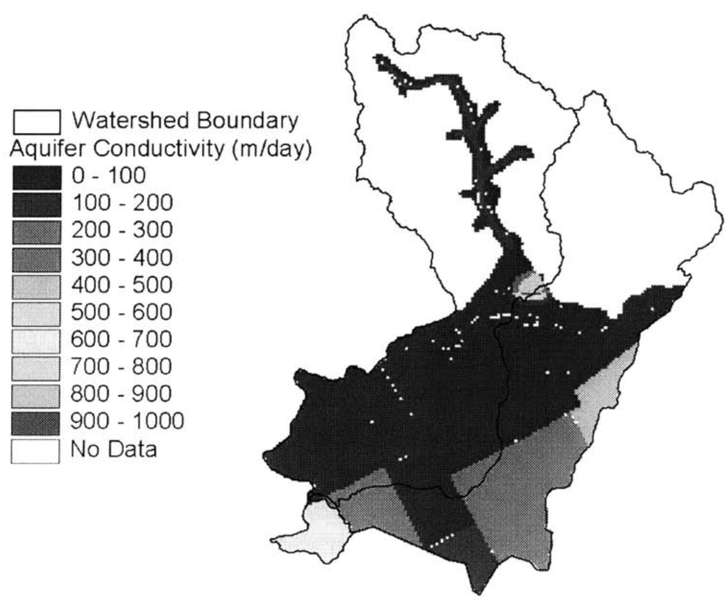


Figure 4-7: Aquifer Conductivities in the Wood River Valley Watershed

Hydraulic Gradient $\left(\frac{dh}{dL}\right)$: Hydraulic gradient is effectively the slope of the water table. We obtained a GIS layer from Blaine County that shows the hydraulic head (water table elevation) for a series of wells within the watershed.

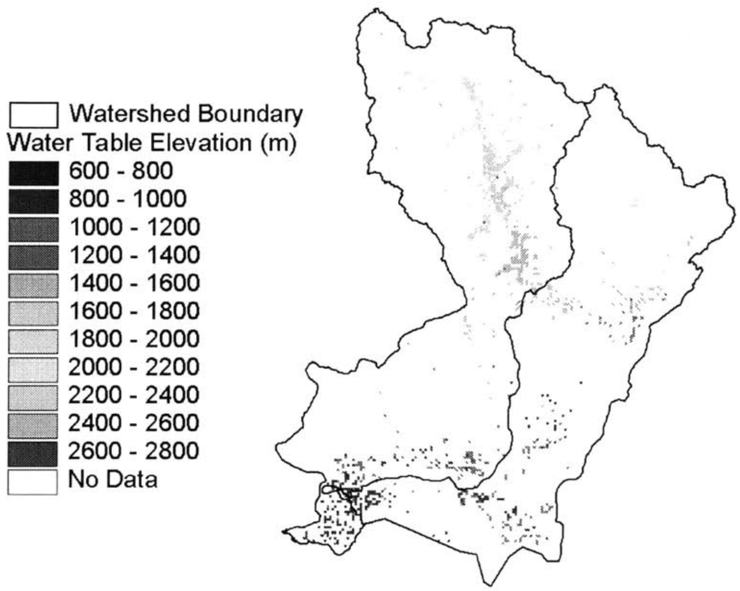


Figure 4-8: Hydraulic Head (m) for Wells within the Wood River Valley Watershed

While this data set contains a large number of wells, there are large areas of the watershed that have no representative water table elevation value. In order to define an approximate water table elevation for all areas of the watershed, ground water table contour lines were drawn by hand based on the original well data set.

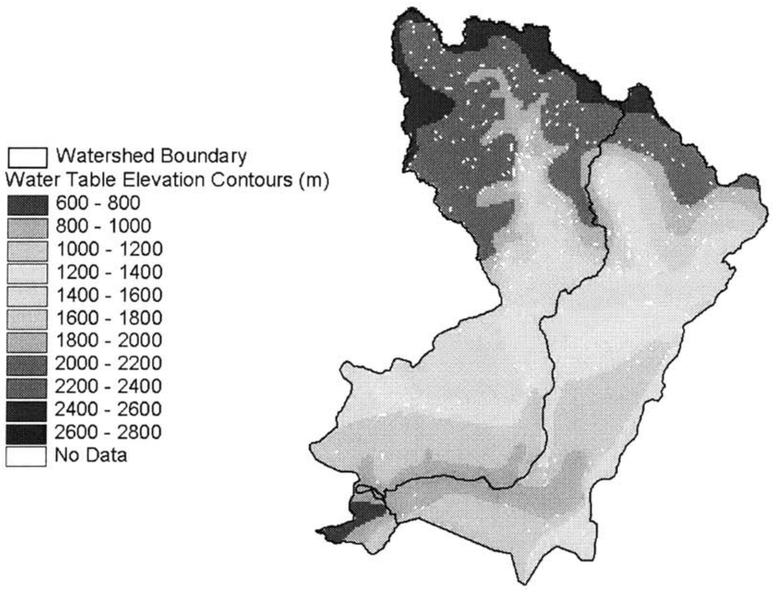


Figure 4-9: Water Table Elevation Contours in the Wood River Valley Watershed

Having defined a water table elevation for each location in the watershed, we are able to estimate the change in that elevation from any given area in the watershed to the closest stream or river channel. Like the water table elevation contours, zones of equal head distance above the river were also estimated by hand and subsequently incorporated into the GIS.

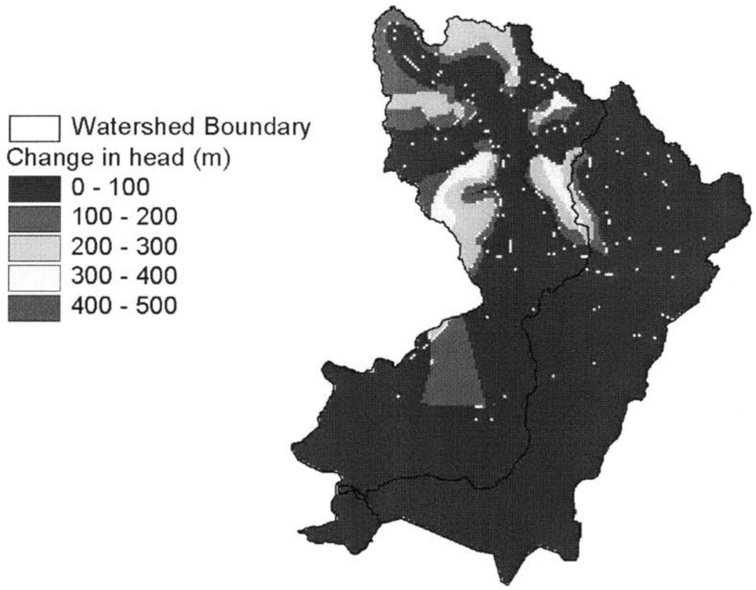


Figure 4-10: Change in Hydraulic Head from Any Location within the Wood River Valley Watershed to the Nearest Stream or River Channel

By dividing the change in hydraulic head by the distance from any location to the nearest river, we determine the hydraulic gradient, $\frac{dh}{dL}$. It is important to recognize that we have defined *dh/dL* as the total change in head from a given location to the nearest river divided by the distance from that location to the nearest river. Essentially, we have defined the hydraulic gradient to remain constant as water travels from any given location in the watershed to the nearest river.

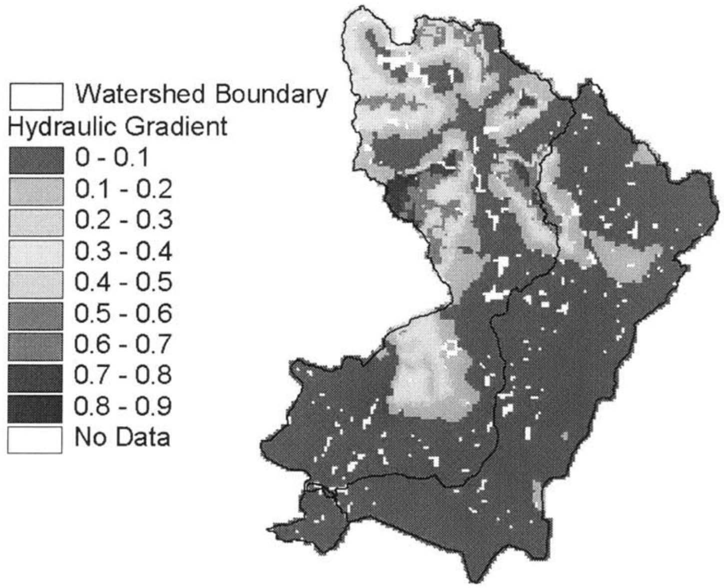


Figure 4-11: Variation in Hydraulic Gradient $\left(\frac{dh}{dL}\right)$ within the Wood River Valley Watershed

Porosity (n): Aquifer porosity, *n*, is a difficult value to identify. Based on discussions with a local hydrogeologist, we estimated the aquifer porosity to be 0.25 across the entire watershed (Brown 2003). While this estimation is probably a gross oversimplification of the aquifers, it is the best estimate that we can make based on available data. Additionally, this estimate is consistent with reported porosity ranges for sedimentary aquifers and crystalline rocks

Table 4-2: Range in Porosity Values (adapted from Domenico and Schwartz 1998)

Material	***Porosity***
SEDIMENTARY	
Gravel, coarse	0.24-0.36
Gravel, fine	0.25-0.38
Sand, coarse	0.31-0.46
CRYSTALLINE ROCKS	
Fractured crystalline rocks	0-0.10
Dense crystalline rocks	0-0.05
Basalt	0.03-0.35
Weathered granite	0.34-0.57

Groundwater Velocity: Combining spatially distributed information on hydraulic conductivity, hydraulic gradient, and porosity, we are able to calculate groundwater velocities for the entire valley region of the watershed.

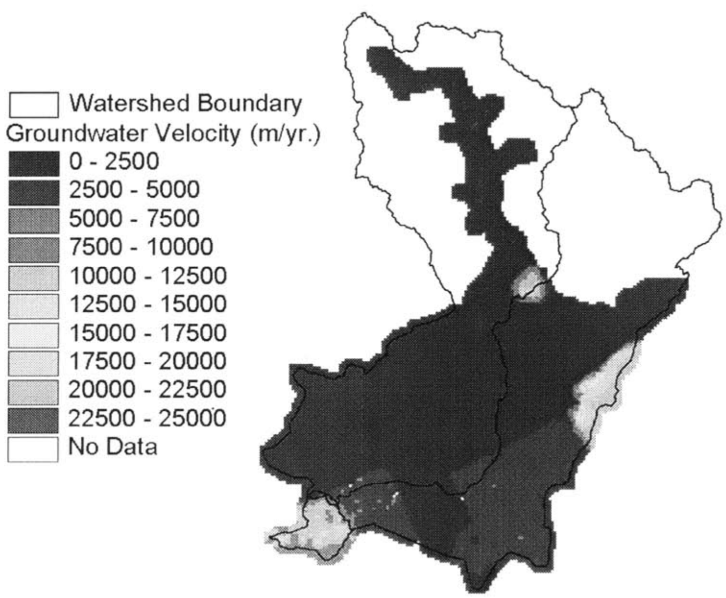


Figure 4-12: Groundwater Velocity within the Wood River Valley Watershed

Because porosity is taken to be constant over the entire watershed and $\frac{dh}{dL}$ is relatively constant throughout the valley, groundwater velocity is primarily a function of hydraulic conductivity, K.

Groundwater Travel Time: By dividing distance from any location to the nearest river by groundwater velocity, we obtain groundwater travel time for the valley region of the watershed. Combining these times with the run-off travel times calculated for the mountain region, we can define the amount of time it will take water to travel from any location in the watershed to the nearest stream or river channel.

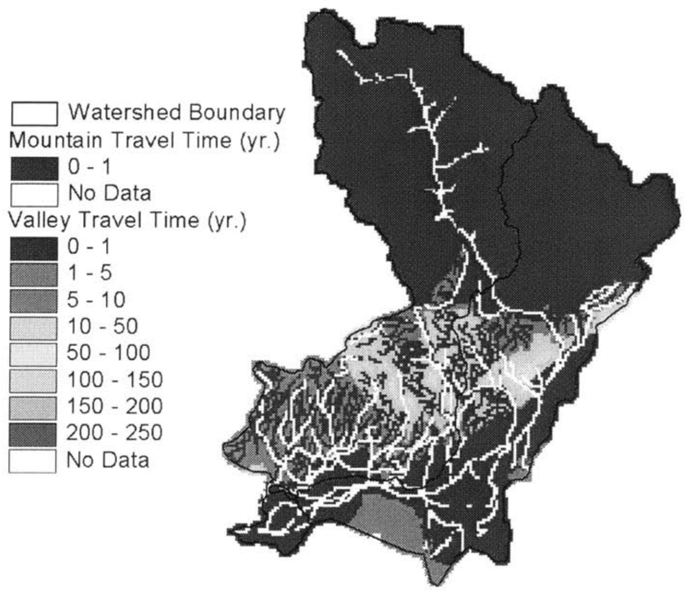


Figure 4-13: Variation in Run-off and Groundwater Travel Time within the Wood River Valley Watershed

The range of groundwater travel times is broad, ranging from less than 1 year to greater than 250 years (the scale in Figure 4-13 represents varying intervals of travel time); however, 85% of the watershed, or approximately 5,800 ha, falls within the 1-year travel time band.

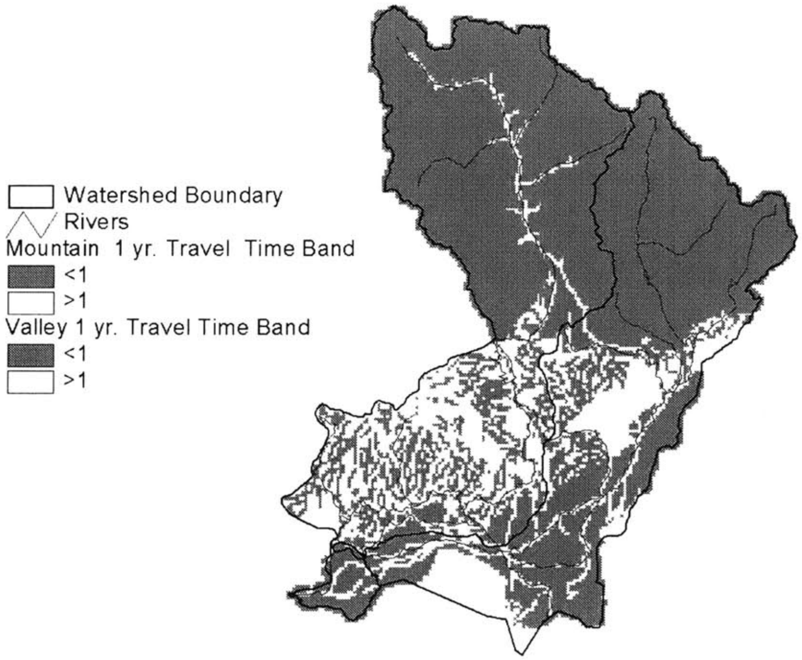


Figure 4-14: One-year Travel Time Band for the Wood River Valley Watershed

As discussed in the previous section, nitrogen inputs to land areas in different travel time bands are delayed in the BEANS Model according to that band's characteristic travel time; thus, in calculating the 2002 nitrogen load to the Big Wood and Little Wood Rivers, the BEANS model should consider nitrogen inputs from 2002 for the 1-year travel time band, nitrogen inputs from 2001 for the 2-year travel time band, and nitrogen inputs from 1998 for the 5-year travel time band, etc.

Much of the land area that falls outside the 1-year travel time band in the Wood River Valley Watershed is covered with exposed rock, Snake River Basalts. The presence of these rocks limits the types of land uses that can occur in these areas and also prevents future agricultural, residential, or commercial development of these areas. Current land use in this area is dominated by impervious surfaces and rangeland.

Table 4-3: Land Use Outside of the 1-year Travel Time Band

Land Use	***Area (ha)***	***Percentage***
Agricultural Land	5,000	5 %
Impervious Surfaces	19,000	18 %
Rangeland	82,000	77 %

Because land use outside the 1-year travel time band is relatively uniform and is unlikely to change with time, the BEANS model makes the assumption that nitrogen inputs to these land areas have been and will continue to be relatively constant. This assumption simplifies the mass balance calculations within the BEANS model such that in calculating the nitrogen load to the rivers, the model considers land use information from only one year.

4.2 NITROGEN MASS BALANCE CALCULATIONS

Before nitrogen inputs and losses are calculated, the Wood River Valley Watershed is divided into areas of different land use. We obtained the land use GIS data layer in Figure 4-15 from the USEPA BASINS spatial data database.

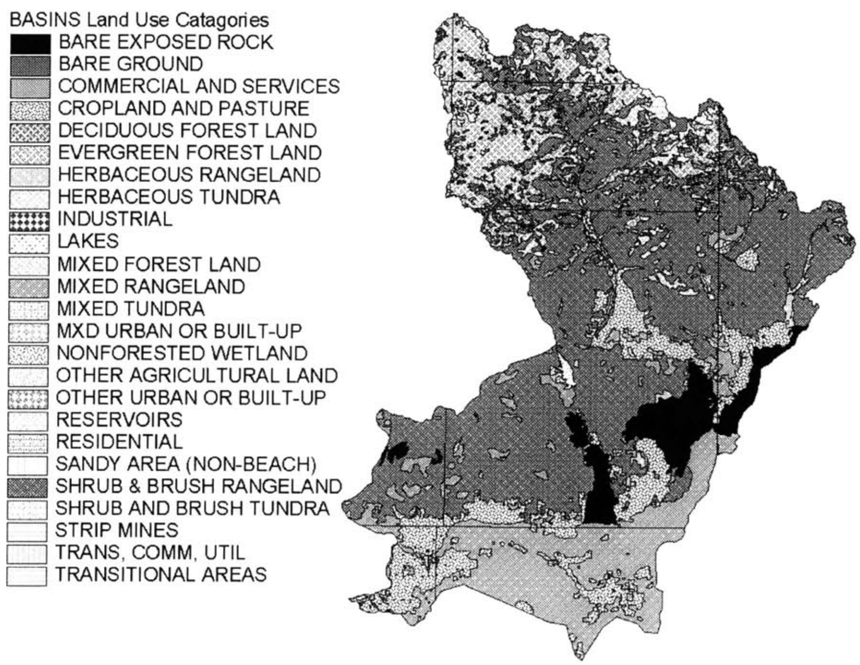


Figure 4-15: Initial Land Use Categories from BASINS Data Layer

All of the land use categories in the BASINS data set are reassigned to one of the seven land use types considered in the BEANS model: cropland and pasture, feed-lots, natural vegetation, rangeland, residential areas, water, and impervious surfaces (Figure 4-16).

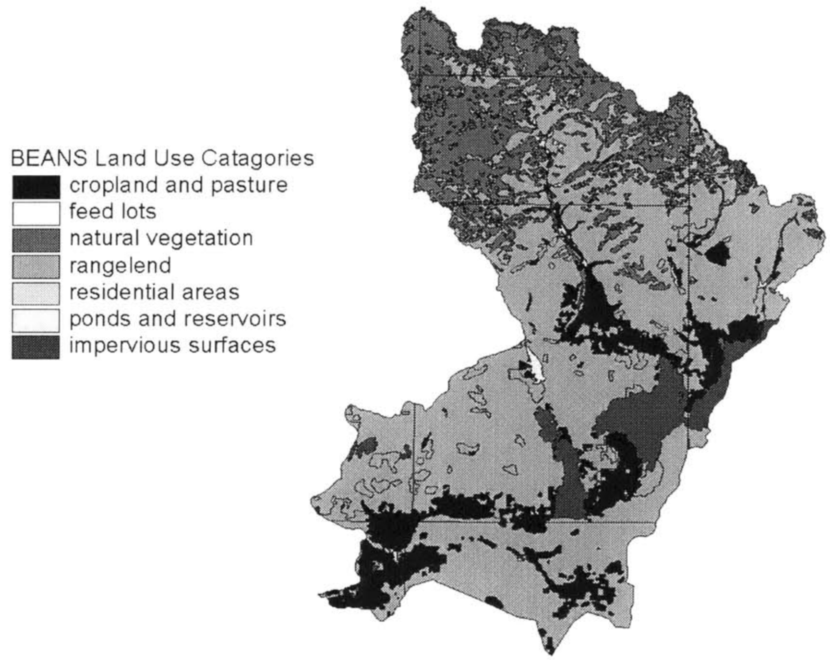


Figure 4-16: BEANS Land Use Category Areas in the Wood River Valley Watershed

The Wood River Valley Watershed covers an area of approximately 680,000 ha. Rangeland is the most common land use within the watershed and feed-lots are the least common.

Table 4-4: Land Use Areas within the Wood River Valley Watershed

Land Use	***Area (ha)***
Rangeland	427,900
Natural vegetation	115,000
Cropland and pasture	84,500
Impervious surfaces	48,200
Residential areas	2,200
Water	2,000
Feed-lots	215

4.2.1. Cropland and Pasture

Agricultural lands receive nitrogen inputs from atmospheric deposition, fertilizer applications, nitrogen fixation, and animal waste.

Atmospheric Deposition: Average annual wet deposition for this area of Idaho is approximately 1 kg N/ha. This value is obtained from the National Atmospheric Deposition Program Craters of the Moon National Monument Monitoring Station (NADP 2002), which is located in Butte County, Idaho. The BEANS model assumes that dry deposition is approximately equal to wet deposition (Valiela et al. 1997, Boyer et al. 2002), so the total annual deposition rate is 2 kg N/ha. This deposition rate is applied uniformly to all the cropland and pasture areas within the watershed, and then 62% of this nitrogen is lost to retention by plants and soils (Valiela et al. 1997, Brawley et al. 2000). Overall, we estimate that approximately 59,900 kg of nitrogen from atmospheric deposition percolates into the subsurface beneath cropland and pasture each year.

Fertilizer Applications: Fertilizer application rates vary between crops, so the cropland and pasture area within the watershed is separated into areas of different crops. The area of land within the watershed that is dedicated to an individual crop is estimated using data from the United States Census of Agriculture (NASS 1997). This census is conducted every five years, so the most recent data available are from 1997. The census data are reported at a county-wide level. Crop data for the watershed are estimated from data for the five counties within which the watershed falls: Blaine, Camas, Gooding, Jerome, and Lincoln. The crops that are grown within these five counties are: barley, wheat, sugarbeets, corn, oats, alfalfa hay, dry beans, and potatoes. The area of each crop within the watershed is estimated based on the percentage of each county that falls within the watershed boundary. For example, the area of potatoes in the watershed is calculated by multiplying the area of potatoes in Blaine County by the percent of Blaine County that falls within the watershed, then adding that area to the area of potatoes in Camas County multiplied by the percent of Camas County that falls within the watershed, etc. This estimation is carried out for each of the crops mentioned above.

Table 4-5: Crop Areas within the Wood River Valley Watershed

	Blaine County	***Camas County***	***Gooding County***	***Jerome County***	***Lincoln County***	***Watershed***
% in watershed	*59%*	*7%*	*41%*	*3%*	*55%*	
Barley (ha)	8,200	5,100	1,500	7,300	4,400	**8,400**
Wheat (ha)	900	1,300	3,600	7,600	5,000	**5,000**
Sugarbeets (ha)	0	0	1,900	5,600	3,000	**2,600**
Corn (ha)	0	0	8,100	10,400	2,400	**4,900**
Oats (ha)	1,300	400	400	1,900	2,000	**2,000**
Alfalfa Hay (ha)	7,900	17,800	14,000	19,600	7,700	**16,400**
Dry Beans (ha)	0	0	400	3,400	0	**200**
Potatoes (ha)	800	0	3,300	6,100	2,400	**3,300**
					TOTAL:	**42,800**

The total crop area within the watershed is approximately 43,000 ha. The BEANS model assumes that the remaining 41,000 ha classified as cropland and pasture is pasture. It is important to recognize that this method of determining crop areas assumes that agricultural areas are distributed uniformly across each county. This assumption may not be true, but given that crop data are available at the county level, this estimation is the most accurate approximation we could make.

Different crops can have extremely different fertilizer application rates, so in the BEANS model, an understanding of where each crop is being grown (and fertilized) within the watershed is important. We gained an understanding of the spatial distribution of crops in the watershed through conversations with local growers (Purdy 2003, Gardner 2003). Through these conversations, we learned that crops are grown in different areas of the watershed as a result of temperature, soil type, and precipitation variations. In general, agricultural land in the northern half of the watershed is devoted to alfalfa hay, barley, and pasture. Crops such as potatoes, wheat, corn, oats, and sugarbeets are grown in the southern half of the watershed. Based on these controlling factors, we define seven distinct agricultural areas within the watershed (Purdy 2003, Gardner 2003). These seven agricultural areas are referred to as area 1 through area 7 and are illustrated in Figure 4-17.

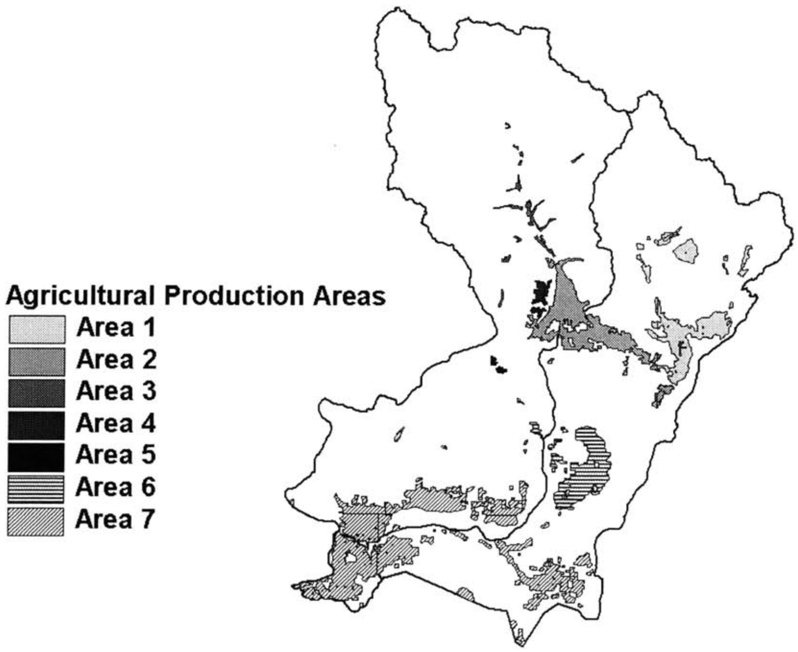


Figure 4-17: Agricultural Production Areas in Wood River Valley Watershed

The nitrogen fertilization rate for each of the seven agricultural production areas is a function of the fertilization rates for each individual crop, the amount of land in each agricultural production area devoted to each crop, and the typical crop rotation pattern for that area. The area of each crop and pasture in these agricultural sections is shown in Table 4-6.

Table 4-6: Crops within the Seven Agricultural Areas

(ha)	*Area 1*	*Area 2*	*Area 3*	*Area 4*	*Area 5*	*Area 6*	*Area 7*
Barley	0	2,400	2,300	0	0	2,400	1,400
Wheat	0	0	0	0	0	1,300	3,800
Sugarbeets	0	0	0	0	0	0	2,600
Corn	0	0	0	0	0	0	4,900
Oats	0	0	0	0	0	0	2,100
Alfalfa Hay	2,200	3,300	0	0	500	7,100	3,300
Dry Beans	0	0	0	0	0	0	200
Potatoes	0	1,800	0	0	0	0	1,500
Pasture	10,200	7,200	0	1,500	0	0	22,100
TOTAL	**12,400**	**14,700**	**2,300**	**1,500**	**500**	**10,800**	**42,200**

The agricultural areas are differentiated based on the types of crops grown. Some areas contain only one type of crop, while area 7 contains all possible crops as well as pasture. The areas also substantially vary in size. Table 4-7 lists the nitrogen fertilizer application rates used in the BEANS model. We used local fertilizer application rates obtained from communication with local farmers whenever possible (Beck 2003, Gardner 2003, Hansen 2003). When local rates are not available, the BEANS model uses the average U.S. fertilizer application rate as reported by the International Fertilizer Industry Association (IFA 1998).

Table 4-7: Crop Nitrogen Fertilization Rates

Crop	***N Fertilization Rate (kg/ha)***	***Source***
Barley	112	Gardner (2003)
Wheat	112	Gardner (2003)
Sugarbeets	120	IFA (1998)
Corn	151	Beck (2003)
Oats	151	Beck (2003)
Alfalfa hay	34	Mahler (1999)
Dry beans	80	IFA (1998)
Potatoes	190	Hansen (2003)
Pasture	0	Beck (2003)

Potatoes, corn, and oats have the highest fertilizer application rates, while alfalfa hay and dry beans have the lowest fertilizer application rates. Crops are rotated in an agricultural production area to maximize production yield (Purdy 2003). Adding nitrogen-fixing plants, such as alfalfa hay, to the crop rotation sustains the productivity of the soil (Purdy, 2003). Alfalfa hay is fertilized much less than other crops grown in the watershed (Table 4-7), and introduction of alfalfa into a crop rotation pattern subsequently lowers the time-averaged nitrogen fertilization rate. Using crop areas, crop fertilizer application rates, and crop rotation patterns for the seven agricultural areas, we defined average fertilizer application rates for each of these areas.

Table 4-8: Crop Rotations and Annual Nitrogen Fertilization Rates for Agricultural Production Areas in Wood River Valley Watershed

Agricultural Area	***Crop Rotation Pattern***	***Annual Nitrogen Fertilization Rate (kg/ha)***
Area 1	no rotation	6
Area 2	2 years barley, 5 years alfalfa hay, 1 year potatoes, 5 years alfalfa hay (Gardner)	49
Area 3	2 years barley, 5 years alfalfa hay (Gardner)	112
Area 4	no rotation	0
Area 5	no rotation	34
Area 6	2 years barley, 5 years alfalfa hay, 3 years wheat, 5 years alfalfa hay (Gardner, Purdy)	60
Area 7	no rotation	56

These area-specific fertilizer application rates are applied to the total land area within each agricultural section. Thirty-nine percent of the original fertilizer applied is lost as gasses, and then 62% of this remaining nitrogen is lost to retention by plants and soils (Valiela et al. 1997, Brawley et al. 2000). This method of an average, area-specific fertilizer application rate is used because while we know generally where different crops are located within the watershed (enough to separate them into areas), we do not know the exact location of different crops within a given area and those crops change over time in most cases. Overall, we estimate that 945,000 kg of nitrogen from fertilizers percolates into the subsurface below cropland and pasture each year.

Nitrogen Fixation: Nitrogen fixation occurs in the agricultural areas that contain either dry beans or alfalfa hay. Areas 1, 2, 5, 6, and 7 all contain alfalfa hay, but only area 7 contains dry beans. The nitrogen fixation rates for dry beans and alfalfa hay are 93 kg N/ha and 224 kg N/ha respectively (Baldwin et al. 2000). Nitrogen from fixation does not enter the soil until the plants are tilled under, so the total fixation rates are reduced by the percentage of the bean or alfalfa crop that is rotated out each year (Baldwin et al. 2000). We assume that 62% of nitrogen introduced from nitrogen fixing plants is lost to retention by plants and soils (Valiela et al. 1997, Brawley et al. 2000). Overall, we estimate that 350,000 kg of nitrogen from fixation percolates into the subsurface below cropland and pasture each year.

Animal Waste: Nitrogen from animal waste is introduced to agricultural areas that contain pasture. Agricultural areas 1, 2, 4, and 7 all contain pasture. Pastures receive nitrogen from both sheep and cattle wastes but not from chickens and hogs, which are usually confined to feed-lots. The magnitude of the nitrogen inputs from sheep and cattle is calculated by multiplying a pasture animal density by the area of land in pasture to determine the number of animals on a given area of pasture. Because the winter season in the Wood River Valley can be extremely harsh, some livestock do not spend the entire year in the watershed but instead are moved south for the colder months of the year (Purdy 2003), so the number of animals calculated based on animal densities is reduced by the percentage of the year that the animals spend outside the watershed. This effective population is multiplied by an animal-specific nitrogen excretion rate to obtain the mass of nitrogen that is introduced to the land surface from animals.

Sheep Waste: Sheep are assumed to graze on both pasture and rangeland. The BEANS model assumes that the density of sheep is the same on both pasture and rangeland. The density of sheep in the Wood River Valley Watershed is determined by dividing the total number of sheep in the watershed by the total area of pasture and rangeland in the watershed. It should be noted that at any given time, the density of sheep is probably higher than the density we calculate. The sheep are likely kept in well-defined herds that are moved around to different grazing areas. The density of any individual herd will be greater than the overall density used in the BEANS model; however, the BEANS model calculates the annual nitrogen load to the watershed. On an annual scale, the effect of denser herds moving around the total rangeland and pasture area will be similar to the effect of a less dense population of sheep that remains stationary, as the model assumes.

The total number of sheep in the watershed is calculated in a method similar to how the total area of each crop is calculated. County-wide animal statistics are combined based on the percent of each county that falls within the watershed to obtain watershed animal populations. County-wide animal populations are obtained from Environmental Defense's Scorecard online database (Environmental Defense 2002). These values are collected from the United States Census of Agriculture, so the most recent data available are from 1997.

Table 4-9: Number of Sheep in the Wood River Valley Watershed

	Blaine County	***Camas County***	***Gooding County***	***Jerome County***	***Lincoln County***	***Watershed***
% in watershed	*59%*	*7%*	*41%*	*3%*	*55%*	
Sheep	31,300	0	26,700	0	800	**29,700**

Dividing the 29,700 sheep in the watershed by 470,000 ha of pasture and rangeland yields a sheep density of 0.06 sheep/ha.

The total area of pasture in the watershed is 41,000 ha, so we infer the total number of sheep on pasture to be 2,600 animals. Boyer et al. (2002) conducted a comprehensive survey of livestock waste production rates, and the values they identified for sheep are summarized in Table 4-10.

Table 4-10: Annual Sheep Nitrogen Excretion Rates, kg N/animal (adapted from Boyer et al. 2002)

	Thomas &Gilliam 1997	*Blekan & Bakken 1997*	*Van Horn 1998*	*Van der Hoak & Bouwman 1999*	*Smil 1999*	*SCS 1992*
Sheep	---	12.3	**5.00**	19.90	---	---

The nitrogen excretion rate used for sheep in the BEANS model is 5.00 kg N/animal because it is the value that best represents current agricultural management practices in the U.S. (Boyer et al. 2002).

From speaking with local ranchers, we learned that sheep spend about two months in the summer in the watershed and then are moved to warmer locations, so the sheep spend only about 17% of the year within the watershed (Purdy 2003). Fifty-five percent of nitrogen introduced to pasture as sheep waste is lost to volatilization as ammonia (Boyer et al. 2002), and 62% of the remaining nitrogen retained in plants and soils (Valiela et al. 1997, Brawley et al. 2000). Overall, we estimate that approximately 360 kg of nitrogen from sheep waste percolates into the subsurface below pastures. This is the smallest nitrogen input to agricultural lands.

Cattle Waste: Like sheep, cattle graze on both pasture and rangeland, but unlike sheep some cattle are also kept in feed-lots; therefore, the cattle density number is calculated as the total number of cattle in the watershed less the number of cattle in feed-lots. The total number of cattle in the watershed is estimated in the same way as the total number of sheep is estimated.

Table 4-11: Number of Cattle in the Wood River Valley Watershed

	Blaine County	***Camas County***	***Gooding County***	***Jerome County***	***Lincoln County***	***Watershed***
% in watershed	*59%*	*7%*	*41%*	*3%*	*55%*	
Cattle	26,800	11,200	141,000	133,600	36,400	**97,200**

The number of cattle in feed-lots is calculated by multiplying the density of cattle in feed-lots by the area of feed-lots used for cattle. A cattle feed-lot density of 96 head/ha is estimated from information on the number of head in a local feed-lot (Purdy, 2003). The area of feed-lots used for cattle is calculated as the area of feed-lot not used for hogs or chickens. Hogs and chickens are kept in feed-lots exclusively, while cattle are split between feed lots and grazing areas. The specifics of this calculation are discussed in greater detail in the feed-lot section. The area of feed-lots used for cattle is about 26 ha, which is 12% of the total feed-lot area in the watershed. Subtracting the 2,500 feed-lot cattle from the total cattle population means that about 91,750 cattle are left on pasture and rangeland. Distributing these cattle across the 470,000 ha of pasture and rangeland yields a cattle density of 0.2 head/ha.

The total area of pasture in the watershed is 41,000 ha, so the total number of cattle on pasture is 8,200 animals. The nitrogen excretion rates identified by Boyer et al. (2002) for beef cattle are summarized in Table 4-12.

Table 4-12: Annual Cattle Nitrogen Excretion Rates, kg N/animal (adapted from Boyer et al. 2002)

	Thomas &Gilliam 1997	*Blekan & Bakken 1997*	*Van Horn 1998*	*Van der Hoak & Bouwman 1999*	*Smil 1999*	*SCS 1992*
Cattle	44.00	66.60	**58.51**	40.70	50.00	41.72

The nitrogen excretion rate used for cattle in the BEANS model is 58.51 kg N/animal because it is the value that best represents current agricultural management practices in the U.S. (Boyer et al. 2002).

Local farmers informed us that cattle, unlike sheep, spend the entire year in the watershed (Purdy 2003), and the total mass of nitrogen introduced by cattle is not reduced by time spent out of the watershed. Thirty-two percent of the nitrogen in cattle waste is lost to volatilization as ammonia (Boyer et al. 2002), and 62% of the remaining mass is retained in plants and soils (Valiela et al. 1997, Brawley et al. 2000). Overall, we estimate that 125,000 kg of nitrogen from cattle waste percolates into the subsurface below pastures.

Losses in the Vadose Zone and Aquifer: In total, we estimate that approximately 1,480,000 kg of nitrogen percolates through to the subsurface from cropland and pasture land areas. On average, this is about 18 kg N/ha. The BEANS model assumes the 61% of nitrogen is lost in the vadose zone, leaving about 578,000 kg to travel through the aquifer (Valiela et al. 1997, Brawley et al. 2000). Denitrification activity in aquifers around the Wood River Valley Watershed has been estimated to remove between 0% and 40% of incoming nitrogen (Rupert 1996). Because denitrification rate depends on nitrate concentration, denitrification losses are greater in areas with higher nitrate concentrations (Pabich (submitted)). Denitrification losses are calculated for the seven agricultural areas using the Pabich et al. model that estimates denitrification rate as a function of initial nitrate concentration. Nitrate concentration in groundwater below the seven agricultural areas are estimated based on annual nitrogen inputs and average annual precipitation for each area (Table 4-13).

Table 4-13: Modeled Denitrification Losses for Agricultural Areas in the Wood River Valley Watershed

Agricultural Area	***Nitrate Concentration (mg N/L)***	***Denitrification Loss***
Area 1	0.9	34%
Area 2	1.9	34%
Area 3	2.8	35%
Area 4	0.4	33%
Area 5	3.1	35%
Area 6	3.0	35%
Area 7	1.8	34%

Because some water introduced as precipitation is lost to evapotranspiration, the nitrate concentrations estimated based on average precipitation are probably lower than true concentrations because the volume of water available to dissolve the input nitrogen is actually smaller than the total precipitation volume. Because losses are greater for areas with higher initial concentrations, the true loss percentages are likely greater than those presented in Table 4-13; consequently, the BEANS model assumes that denitrification losses from agricultural lands are approximately 40%.

The average agricultural nitrogen input of 18 kg N/ha is in the high range of nitrogen inputs in the watershed, so the 40% loss of nitrogen from agricultural areas during travel through aquifers is consistent with the observation of elevated loss percentages from areas with elevated nitrate concentrations. The magnitude of this loss also is similar to the 35% loss reported by Valiela et al. (1997) for a residential watershed on Cape Cod Massachusetts. The BEANS model estimates 346,000 kg of nitrogen from agricultural areas is able to enter the surface waters of the watershed. Figure 4-18 depicts the relative proportion of this mass that originates from the different nitrogen sources considered.

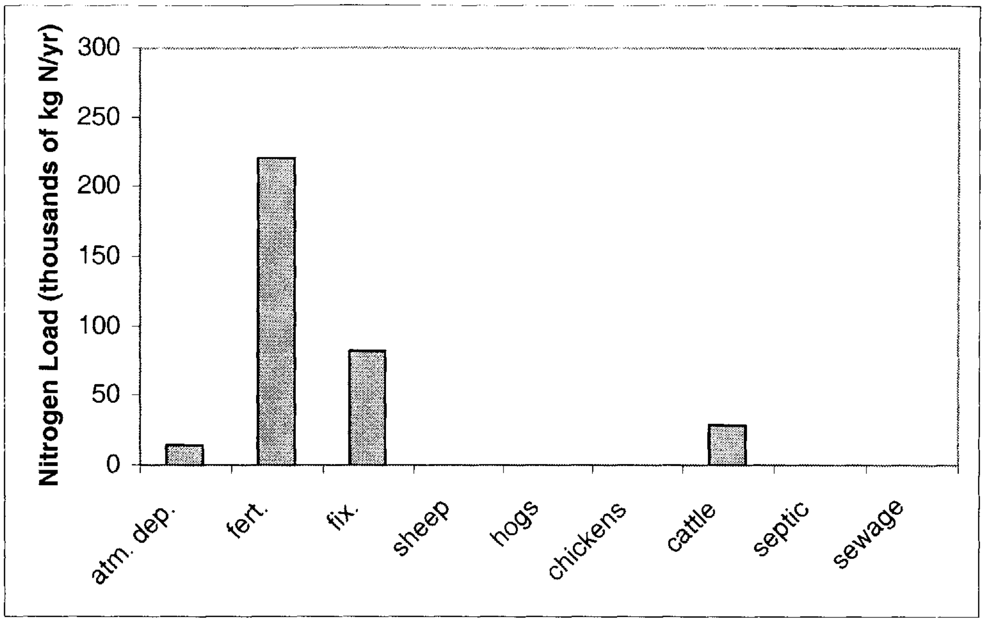


Figure 4-18: Relative Magnitude of Source Nitrogen Loads from Agricultural Land Areas

4.2.2. Feed-lots

Feed-lots receive nitrogen inputs from atmospheric deposition and animal waste from hogs, chickens, and cattle.

Atmospheric Deposition: Atmospheric deposition is assumed to be constant throughout the watershed, so the annual deposition rate of 2 kg N/ha is applied to all areas designated as feed-lots (NADP 2002). Sixty-two percent of nitrogen deposited onto agricultural land is retained by plants and soils (Valiela et al. 1997, Brawley et al. 2000). The density of animals in feed-lots limits the ability of these land areas to maintain substantial vegetation, so we assume in the BEANS model that nitrogen retention occurs only through soil mechanisms. The soil nitrogen retention rate is assumed to be 31%, half of the combined soil and plant retention rate. Overall, we estimate that 280 kg of nitrogen from atmospheric deposition percolates into the subsurface beneath feed-lots each year.

Animal Waste: Feed-lots receive animal waste from hogs, chickens, and cattle. The magnitude of the nitrogen inputs from hogs, chickens, and cattle is calculated by multiplying a feed-lot animal density by the area of feed-lots dedicated to each type of animal to determine the number of animals in each feed-lot. These animal population values are multiplied by animal-specific nitrogen excretion rates to obtain the mass of nitrogen introduced to the land surface from each type of animal.

Hog Waste: The BEANS model assumes that all hogs within the watershed are contained in feed-lots. The total number of hogs in the watershed is calculated as an area-weighted average of the number of hogs in each of the five counties that fall within the watershed boundary.

Table 4-14: Number of Hogs in the Wood River Valley Watershed

	Blaine County	***Camas County***	***Gooding County***	***Jerome County***	***Lincoln County***	***Watershed***
% in watershed	*59%*	*7%*	*41%*	*3%*	*55%*	
Hogs	30	0	360	1300	860	**670**

John Patience of the Prairie Swine Centre (PSC) in Saskatchewan, Canada has calculated hog densities around the world (Prairie Swine Center 2003). In the United States, hog densities range from 2.1 head/ha in Iowa to 4.8 head/ha in North Carolina, which is the highest density in the United States (Hursch 2002, PSC 2003). The BEANS model assumes an average hog density of 3.5 head/ha for the Wood River Valley watershed. A total feed-lot area within the watershed dedicated to hogs (190 ha) is calculated by dividing the 670 hogs kept in feed-lots by the density of 3.5 hogs/ha. The total feed-lot area within the watershed is 215 acres, and 88% of this area is used for hogs.

The nitrogen excretion rates identified by Boyer et al. (2002) for hogs are summarized in Table 4-15.

Table 4-15: Annual Hog Nitrogen Excretion Rates, kg N/animal (adapted from Boyer et al. 2002)

	Thomas &Gilliam 1997	*Blekan & Bakken 1997*	*Van Horn 1998*	*Van der Hoak & Bouwman 1999*	*Smil 1999*	*SCS 1992*
Hogs	6.10	4.34	**5.84**	10.46	10.00	19.70

The nitrogen excretion rate used for hogs in the BEANS model is 5.84 kg N/animal because it is the value that best represents current agricultural management practices in the U.S. (Boyer et al. 2002). Seventy-two percent of the nitrogen in hog waste is lost to volatilization as ammonia, and 31% of the remaining nitrogen is retained in soils below feed-lots. Overall, we estimate that 750 kg of nitrogen from hog waste percolates into the subsurface below feed-lots.

Chicken Waste: The BEANS model assumes that all chickens within the watershed are contained in feed-lots. The total number of chickens in the watershed is calculated as an area-weighted average of the number of chickens in each of the five counties that fall within the watershed boundary.

Table 4-16: Number of Chickens in the Wood River Valley Watershed

	Blaine County	***Camas County***	***Gooding County***	***Jerome County***	***Lincoln County***	***Watershed***
% in watershed	*59%*	*7%*	*41%*	*3%*	*55%*	
Chickens	0	0	730	690	0	**310**

Ralph Ernst at the University of California at Davis has calculated average animal densities for chickens in the U.S. (Ernst 1995). Chicken densities range from 107,000 birds/ha to 150,000 birds/ha (Ernst 1995). Because the number of chickens in the watershed is so small, we assume that the density of chickens is relatively low; therefore, a density of chickens in the Wood River Valley Watershed of 110,000 birds/ha is used in the BEANS model. A watershed feed-lot area of 0.003 ha dedicated to chickens is determined by dividing the 310 chickens kept in feed-lots by a density of 110,000 birds/ha. This area is much less than 1% of the total feed-lot area within the watershed.

The nitrogen excretion rates identified by Boyer et al. (2002) for chickens are summarized in Table 4-17.

Table 4-17: Annual Chicken Nitrogen Excretion Rates, kg N/animal (adapted from Boyer et al. 2002)

	Thomas &Gilliam 1997	*Blekan & Bakken 1997*	*Van Horn 1998*	*Van der Hoak & Bouwman 1999*	*Smil 1999*	*SCS 1992*
Chickens	0.83	0.61	**0.55**	0.81	0.30	0.21

The nitrogen excretion rate used for chickens in the BEANS model is 0.55 kg N/animal because it is the value that best represents current agricultural management practices in the U.S. (Boyer et al. 2002). Thirty-six percent of the nitrogen in chicken waste is lost to volatilization as ammonia, and 31% of the remaining nitrogen is retained in soils below feed-lots. Overall, we estimate that 54 kg of nitrogen from chicken waste percolates into the subsurface below feed-lots each year.

Cattle Waste: Cattle within the watershed are kept primarily on rangeland and pasture, though a small number of cattle are also kept in feed-lots. Of the 215 ha of feed-lot area in the watershed, 190 ha is reserved for hogs and chickens, leaving 25 ha of feed-lots for cattle. A local feed lot contains approximately 1,000 head and has an area of about 10.5 ha (Purdy 2003). This density of 95.5 head/ha is low relative to estimates ranging from 250 to 2,000 head/ha made by the Texas Water Resources Institute at Texas A&M University (Reddell et al. 1973). From conversations with local farmers, we learned that cattle feed-lots within the watershed are used mostly for pre-conditioning calves for rangeland grazing, so it seems possible that the cattle density in this watershed could be lower than densities observed on cattle ranches.

In total there are 2,500 cattle in feed-lots within the watershed. As described in the cropland and pasture section, the annual nitrogen excretion rate for cattle is 58.51 kg N/animal (Boyer et al. 2002). Thirty-two percent of the nitrogen in cattle waste is lost through volatilization of ammonia (Boyer et al. 2002), and then 31% of the remaining nitrogen is retained in soils below feed-lots. Overall, we estimate that 68,000 kg of nitrogen from cattle waste percolates into the subsurface below feed-lots each year. This is by far the largest nitrogen input to feed-lots.

Losses in the Vadose Zone and Aquifer: In total, approximately 69,000 kg of nitrogen is estimated to percolate through to the subsurface from feed-lots. On average, this is about 320 kg N/ha. The BEANS model assumes the 61% of nitrogen is lost in the vadose zone, leaving about 27,000 kg to travel through the aquifer (Valiela et al. 1997, Brawley et al. 2000). Because the nitrogen input to feed-lots is relatively high compared to other land uses in the watershed, the BEANS model assumes that 40% of the remaining nitrogen is removed by denitrification as groundwater moves through the aquifer (Rupert 1996, Pabich (submitted)); consequently, approximately 16,000 kg of nitrogen from feed-lots is able to enter the surface waters of the watershed. Figure 4-19 depicts the relative proportion of this mass that originates from the different nitrogen sources considered.

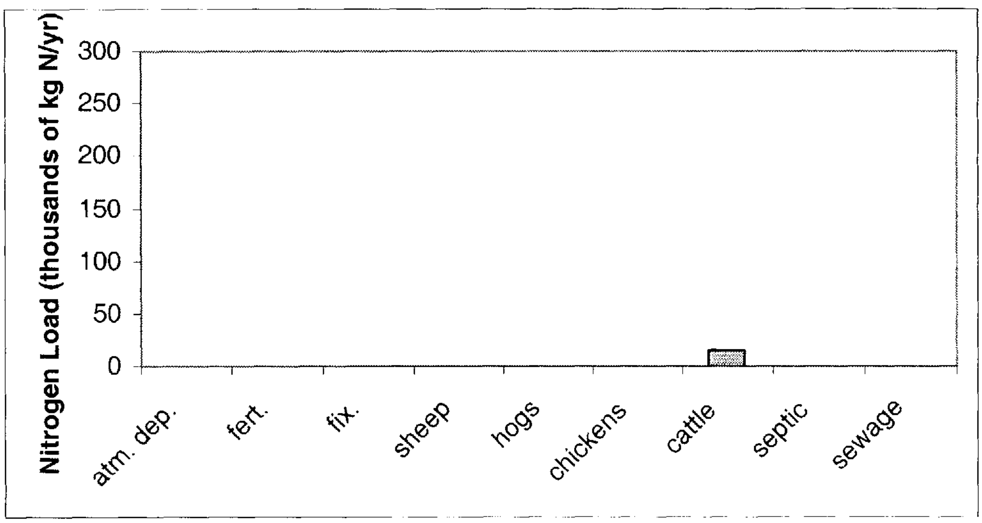


Figure 4-19: Relative Magnitude of Source Nitrogen Loads from Feed-Lots

4.2.3. Rangeland

Rangelands receive nitrogen inputs from atmospheric deposition and animal waste from sheep and cattle.

Atmospheric Deposition: The annual atmospheric deposition rate of 2 kg N/ha is applied to the 430,000 ha of rangeland within the Wood River Valley Watershed. Sixty-five percent of this nitrogen is retained in plants and soils; consequently, 280,000 kg of nitrogen from atmospheric deposition percolates into the subsurface below rangeland.

Animal Waste: The majority of both sheep and cattle in the watershed graze on rangeland. The magnitude of the nitrogen inputs from these animals is calculated by multiplying the pasture/rangeland animal density by the area of rangeland to determine the number of animals on rangeland. Because sheep are removed from the watershed during the winter months (Purdy 2003), the sheep population is reduced by the percentage of the year that the sheep spend out of the watershed. The cattle population and the effective sheep population values are multiplied by animal-specific nitrogen excretion rates to obtain the mass of nitrogen introduced to the lands surface from each type of animal.

Sheep Waste: Using the rangeland sheep density within the watershed of 0.06 sheep/ha, we calculate a total of 27,000 sheep grazing on rangeland. Because these sheep only spend about two months per year in the watershed, the effective sheep population is about 4,500 sheep, 17% of the total population. The annual nitrogen excretion rate for sheep is 5.00 kg N/animal (Boyer et al. 2002). Fifty-five percent of the nitrogen introduced as sheep waste is volatilized as ammonia (Boyer et al. 2002), and then 65% of that remaining nitrogen is retained by plants and soils (Valiela et al. 1997, Brawley et al. 2000). Overall, we estimate that 3,500 kg of nitrogen from sheep waste percolates into the subsurface below rangeland.

Cattle Waste: Using the rangeland cattle density within the watershed of 0.2 head/ha, we calculate a cattle rangeland population of 87,000 head of cattle. The annual nitrogen excretion rate for cattle is 58.51 kg N/animal (Boyer et al. 2002). Thirty-two percent of cattle waste nitrogen is lost through ammonia volatilization (Boyer et al. 2002), and 65% of the remaining nitrogen is retained by plants and soils (Valiela et al. 1997, Brawley et al. 2000). Overall, we estimate that 1,200,000 kg of nitrogen from cattle waste is able to percolate into the subsurface below rangeland.

Losses in the Vadose Zone and Aquifer: In total, approximately 1,490,000 kg of nitrogen percolates through to the subsurface from rangeland. This is an average nitrogen load of 4 kg N/ha. The BEANS model assumes the 61% of nitrogen is lost in the vadose zone, leaving about 580,000 kg to travel through the aquifer (Valiela et al. 1997, Brawley et al.

2000). Because the input to rangeland is only a moderate input compared to other land uses in the watershed, the BEANS model assumes 35% of the remaining nitrogen is removed by denitrification as groundwater moves through the aquifer (Rupert 1996, Pabich (submitted)); consequently, approximately 380,000 kg of nitrogen from rangeland is able to enter the surface waters of the watershed. Figure 4-20 depicts the relative proportion of this mass that originates from the different nitrogen sources considered.

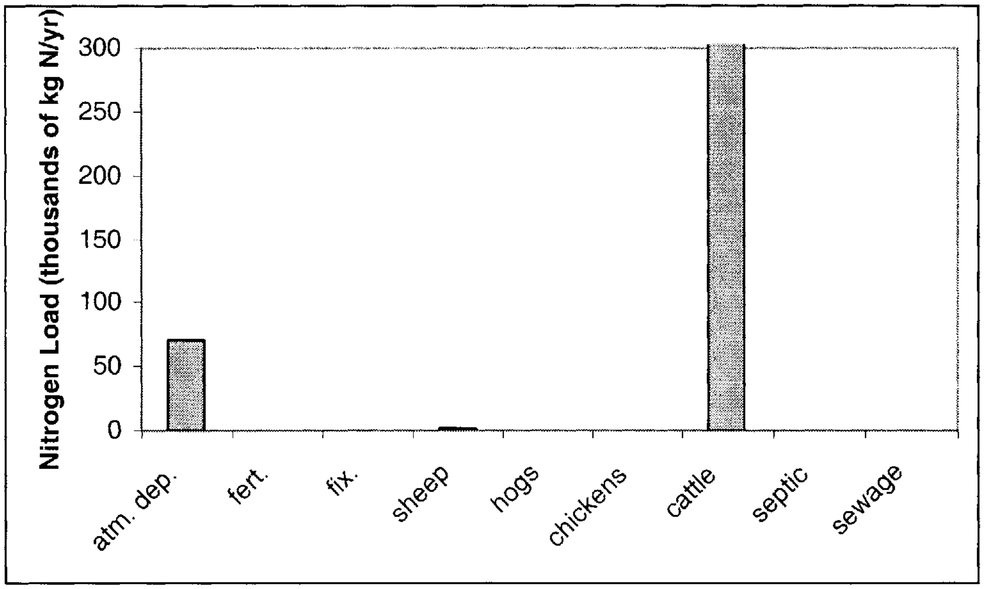


Figure 4-20: Relative Magnitude of Source Nitrogen Loads from Rangeland

4.2.4. Residential

Residential land areas within the watershed receive nitrogen inputs from atmospheric deposition, fertilizer applications, septic systems, and sewage. The first two sources, atmospheric deposition and fertilizer applications, are introduced to the land surface, then percolate into the subsurface where they are transported by groundwater to receiving surface water bodies. Septic system nitrogen is also transported by groundwater but is introduced directly to the subsurface. Nitrogen losses that occur during groundwater transport differ between sources that are introduced to the land surface and those that are introduced to the subsurface, so septic system nitrogen inputs are discussed separately from atmospheric deposition and fertilizer inputs. Sewage nitrogen never enters the groundwater beneath residential areas because it is transported directly to sewage treatment plants by sewers; however, for the purpose of comparing the relative magnitude of nitrogen inputs from different land uses, sewage nitrogen is attributed to the residential areas where it originates rather than to the treatment plants where it is ultimately discharged.

Atmospheric Deposition: The annual atmospheric deposition rate of 2 kg N/ha is applied to the 2,200 ha of residential land within the Wood River Valley watershed. Sixty-five percent of this nitrogen is retained in plants and soils (Valiela et al. 1997, Brawley et al. 2000), resulting in 1,600 kg of nitrogen from atmospheric deposition percolating into the subsurface below residential areas.

Fertilizer Applications: From speaking with local landscapers, we learned that an average of 25% of each residential lot is covered in lawn (Webb 2003). Twenty-five percent of the total residential area in the watershed is 550 ha. Locally, lawn fertilizers are applied at an annual rate of 150 kg N/ha (Webb 2003, Jones 2003). Thirty-nine percent of the original fertilizer applied is lost to the atmosphere as gases, and then 62% of the remaining nitrogen is retained in plants and soils (Valiela et al. 1997, Brawley et al. 2000). Overall, we estimate that 19,300 kg of nitrogen from fertilizer applications percolates into the subsurface below residential areas.

Losses in the Vadose Zone and Aquifer: In total, we estimate that approximately 20,900 kg of nitrogen percolates through to the subsurface from atmospheric deposition and fertilizer applications to residential land areas. The BEANS model assumes the 61% of nitrogen is lost in the vadose zone, leaving about 8,200 kg to travel through the aquifer (Valiela et al. 1997, Brawley et al. 2000). On average, total inputs (including wastewater) to residential land are approximately 50 kg N/ha, which is a large load compared to other land uses in the watershed. Because of these high nitrogen inputs, the BEANS model assumes that 40% of the remaining nitrogen from residential land is removed by denitrification as groundwater moves through the aquifer (Rupert 1996, Pabich (submitted)); consequently, approximately 4,900 kg of nitrogen from surface inputs to residential areas is able to enter the surface waters of the watershed.

Septic System Inputs and Losses: The mass of nitrogen introduced to septic systems in the watershed is calculated by multiplying the number of people using septic systems by an annual per capita nitrogen input. The total population of the watershed is obtained from USGS Water Use Hydrologic Unit Estimates (USGS 2003). The most recent estimates available from the USGS are from 1995. In total, there are 23,950 people living in the watershed (USGS 2003). The population living within the cities of Ketchum, Sun Valley, Hailey, and Bellevue is 13,200; therefore, their homes are connected to one of three municipal wastewater treatment plants (Hyde 2003, Swindell 2003, Wright 2003). Within the watershed, 10,750 people rely on on-site septic systems for treatment of their domestic wastewater.

The annual per capita nitrogen input is 4.8 kg N (Valiela et al. 1997, Ericson 2003), so 51,600 kg of nitrogen are input into septic systems within the watershed each year. On average, 40% of this nitrogen is denitrified within the septic tank and leaching field of the system itself (Valiela et al. 1997, Brawley et al. 2000). The remaining 31,000 kg of nitrogen travels in distinct plumes away from septic leaching fields where an additional 34% of the nitrogen is lost (Valiela et al. 1997, Brawley et al. 2000). After traveling 200m from a leaching field, septic system plumes are diluted enough that they behave similarly to the non-point sources that percolated into the subsurface, so septic system nitrogen inputs are then subject to the 40% loss that is applied to surface-derived nitrogen sources from residential land areas (Rupert 1996, Pabich (submitted)). Ultimately, 12,300 kg of nitrogen from septic systems remains to be transported into receiving surface water bodies. On a per capita basis, septic systems in the watershed discharge about 1.14 kg N/person each year into receiving water bodies.

Sewage Treatment Plant Loading: Nitrogen loading from three wastewater treatment plants in the watershed is considered in the BEANS model: Bellevue, which serves 1,100 people (Wright 2003); Hailey, which serves 6,200 people (Hyde 2003); and Ketchum/Sun Valley, which serves 5,200 people (Swindell 2003). The level of treatment accomplished by each of these plants varies substantially because of differences in the process design of each plant. The nitrogen load for all three plants is calculated directly from plant flow data and concentrations of total nitrogen in the plant effluent.

The Bellevue wastewater treatment facility consists of a series of wastewater aeration and settling lagoons that discharge into a percolation pond. At certain times of year, some of the effluent is applied to an adjacent agricultural field (Turner 2003, Wright 2003). Table 4-18 summarizes the important effluent concentrations from the Bellevue treatment plant.

Table 4-18: Concentrations of Nitrogen in Bellevue Sewage Treatment Plant Effluent (Turner 2003)

Nitrogen Species	***Effluent Concentration (mg N/L)***
TKN [NH_3 + org. N]	19.9
Nitrate [NO_3^-]	6.6
Total N	26.5

The average daily flow rate of the Bellevue wastewater treatment plant is approximately 1.1 million L/day (Turner 2003). The average annual nitrogen load produced from this flow rate and concentration data is 10,500 kg N/yr. Because the Bellevue treatment plant discharges into percolation ponds, this nitrogen is likely subject to the same losses in its plume and the aquifer as septic system nitrogen inputs. After considering plume (34%) and aquifer (40%) losses (Valiela et al. 1997, Rupert 1996, Pabich (submitted)), 4,200 kg of nitrogen from the Bellevue wastewater treatment plant remains to be transported into receiving surface water bodies. On a per capita basis, the Bellevue treatment plant discharges approximately 2.3 kg N/person each year into receiving water bodies.

The current City of Hailey wastewater treatment plant came on-line in 2000 and has the capacity for tertiary treatment of domestic and commercial wastewater (Hyde 2003). Table 4-19 summarizes the important effluent concentrations from the Hailey treatment plant.

Table 4-19: Concentrations of Nitrogen in Hailey Sewage Treatment Plant Effluent (Hyde 2003)

Nitrogen Species	*Effluent Concentration (mg N/L)*
TKN [NH_3 + org. N]	1.0
Nitrate [NO_3^-]	1.5
Total N	2.5

On average, the Hailey sewage treatment plant discharges approximately 2.9 million L/day to the Big Wood River (Hyde 2003). These flow rate and concentration data produce an average annual nitrogen load of 2,666 kg N/yr. On a per capita basis, the Hailey treatment plant discharges approximately 0.4 kg N/person each year into the Big Wood River.

The Ketchum/Sun Valley wastewater treatment plant provides secondary treatment of domestic and commercial wastewater (Swindell 2003). Table 4-20 summarizes the important effluent concentrations from the Ketchum/Sun Valley treatment plant.

Table 4-20: Concentrations of Nitrogen in Ketchum Sewage Treatment Plant Effluent (Swindell 2003)

Nitrogen Species	*Effluent Concentration (mg N/L)*
TKN [NH_3 + org. N]	0.8
Nitrate [NO_3^-]	9.6
Total N	10.4

The average daily effluent flow rate at this treatment plant is 5.9 million L/day, which discharges directly into the Big Wood River (Swindell 2003). These flow rate and concentration data produce an average annual nitrogen load of 22,400 kg N/yr. On a per capita basis, the Ketchum/Sun Valley treatment plant discharges approximately 4.3 kg N/person each year into receiving water bodies.

In total, sewage treatment plants discharge 29,300 kg of nitrogen to the surface water bodies of the watershed each year. This is the largest nitrogen input from residential sources. It is important to remember the differences in the level of wastewater treatment that are embedded in wastewater nitrogen load values.

Table 4-21: Summary of Per Capita Nitrogen Inputs to Receiving Water Bodies from Treated Wastewater

Type of Treatment	*Per Capita N Input (kg N/yr)*	*Total Nitrogen Load (kg N/yr)*
Septic (secondary)	1.2	13,300
Bellevue WWTP (secondary)	2.3	4,500
Ketchum/Sun Valley WWTP (secondary)	4.3	22,400
Hailey WWTP (tertiary)	0.4	2,666

As illustrated by nitrogen concentrations in the effluent from the Hailey wastewater treatment plant, the implementation of tertiary treatment drastically reduces both per capita and total nitrogen loads from wastewater to receiving water bodies. Figure 4-21 depicts the relative proportion of the total residential nitrogen load that originates from the different residential nitrogen sources.

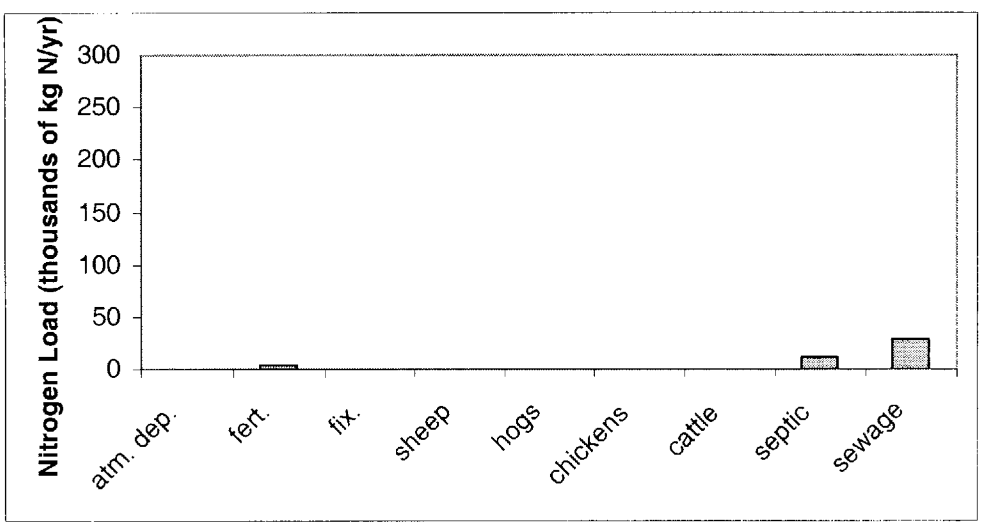


Figure 4-21: Relative Magnitude of Source Nitrogen Loads from Residential Areas

4.2.5. Impervious Surfaces

The impervious surfaces land use category includes mainly urban, commercial, and industrial land areas within the watershed. The only nitrogen input to these land areas is from atmospheric deposition. Because these areas are defined as impervious, the BEANS model assumes the nitrogen inputs enter the groundwater by running off into stormwater drains that discharge directly into the subsurface through dry wells.

Atmospheric Deposition: The annual atmospheric deposition rate of 2 kg N/ha is applied to the 48,000 ha of impervious surfaces within the Wood River Valley watershed. Because this nitrogen is transferred directly to the subsurface without percolating through soils, 100% of the nitrogen introduced from atmospheric deposition is transported into the subsurface. Below impervious surfaces, 91,000 kg of nitrogen from atmospheric deposition enters the subsurface.

Losses in the Vadose Zone and Aquifer: Sixty-one percent of the 91,000 kg of nitrogen entering the subsurface below impervious surfaces is lost in the vadose zone, leaving approximately 35,000 kg to travel through the aquifer (Valiela et al. 1997, Brawley et al. 2000). On average, nitrogen inputs to impervious surfaces are approximately 2 kg N/ha. This is a moderate input considering other land uses in the watershed, so the BEANS model assumes 35% of the remaining nitrogen is removed by denitrification as groundwater moves through the aquifer (Rupert 1996, Pabich (submitted)); consequently, we estimate that approximately 23,000 kg of nitrogen from impervious surfaces is able to enter the surface waters of the watershed. Figure 4-22 depicts the relative proportion of this mass that originates from the nitrogen sources from impervious surfaces.

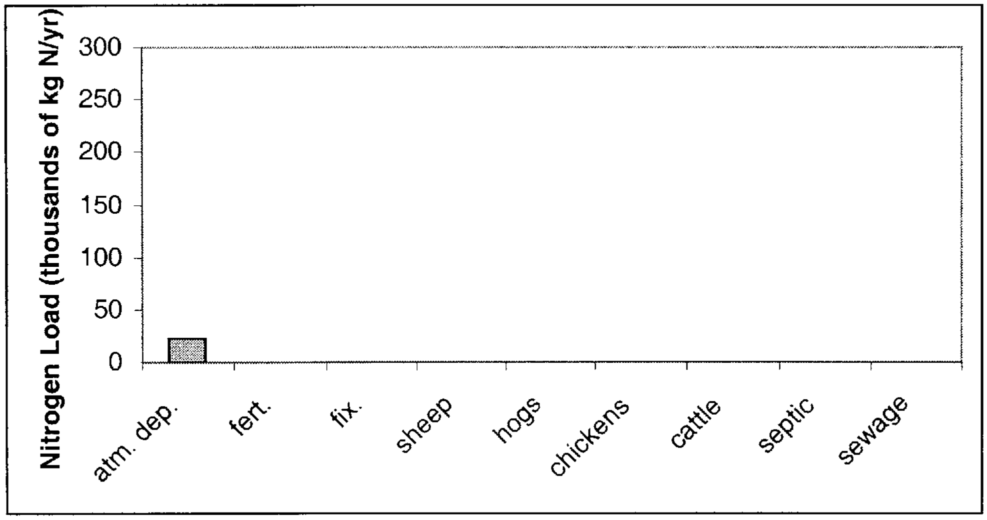


Figure 4-22: Relative Magnitude of Source Nitrogen Loads from Impervious Surfaces

4.2.6. Natural Vegetation

The natural vegetation land use category includes forest and tundra areas within the watershed. The only nitrogen input to these areas is from atmospheric deposition.

Atmospheric Deposition: The annual atmospheric deposition rate of 2 kg N/ha is applied to the 115,000 ha of natural vegetation within the watershed. Sixty-five percent of this nitrogen is retained in plants and soils, allowing 76,000 kg of nitrogen from atmospheric deposition to enter subsurface below natural vegetation.

Losses in the Vadose Zone and Aquifer: Sixty-one percent of the 76,000 kg of nitrogen entering the subsurface below natural vegetation is lost in the vadose zone, leaving 29,500 kg to travel through the aquifer (Valiela et al. 1997, Brawley et al. 2000). On average, less than 1 kg N/ha is input to natural vegetation land areas, which is the lowest nitrogen input to any land use in the watershed. Because of the small nitrogen input to these land areas, the BEANS model assumes that only 30% of the remaining nitrogen is removed by denitrification as groundwater moves through the aquifer (Rupert 1996, Pabich (submitted)). The BEANS model estimates that 20,700 kg of nitrogen from natural vegetation is able to enter the surface waters of the watershed. Figure 4-23 depicts the relative proportion of this nitrogen mass that originates from the different nitrogen sources considered.

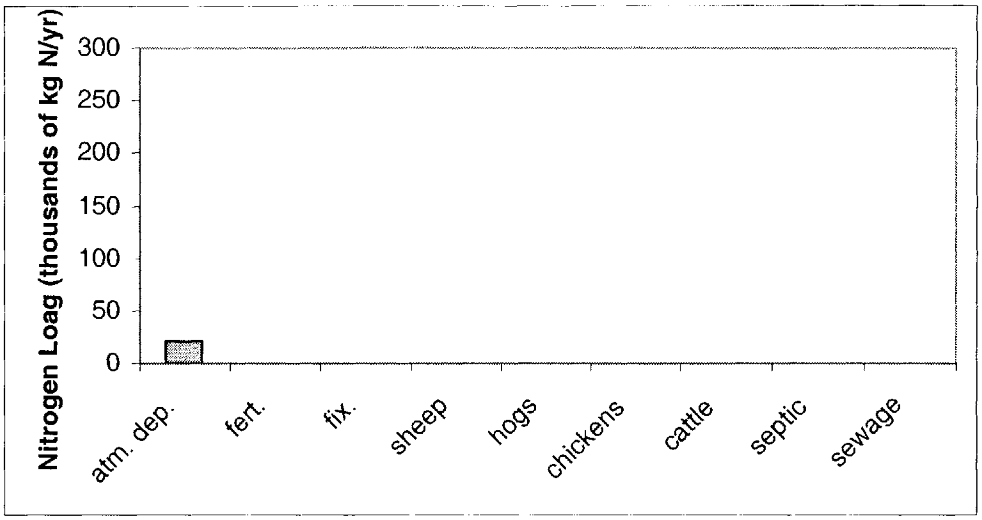


Figure 4-23: Relative Magnitude of Source Nitrogen Loads from Natural Vegetation

4.2.7. Water

The only direct input of nitrogen to ponds and reservoirs within the watershed is from atmospheric deposition.

Atmospheric Deposition: The annual atmospheric deposition rate of 2 kg N/ha is applied to the surface areas of the 18 ponds, lakes, and reservoirs within the watershed. These water bodies have a combined surface area of 2,000 ha, so the total atmospheric deposition input is about 4,000 kg of nitrogen. Lakes and ponds have been shown to retain about 55% of nitrogen inputs (Valiela et al. 1997, Brawley et al. 2000, Brawley et al. 2000). Overall, we estimate that 1,700 kg of nitrogen from atmospheric deposition is transported through the surface water body system within the watershed. Figure 4-24 depicts the relative proportion of this mass that originates from the different nitrogen sources considered.

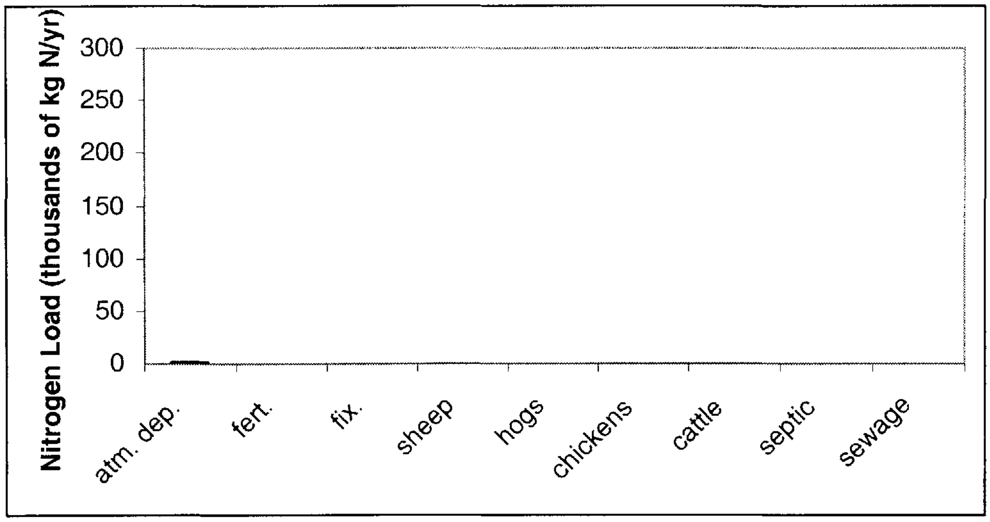


Figure 4-24: Relative Magnitude of Source Nitrogen Loads from Lakes and Reservoirs

5. BEANS Model Results

The BEANS model predicts an annual total nitrogen load to the Wood River Valley watershed of 664,500 kg. This predicted mass incorporates a 20% loss due to instream denitrification. This mass is 30% of the total nitrogen inputs to the watershed, which is consistent with the relationship between riverine nitrogen exports and watershed nitrogen inputs observed by Howarth et al. (1996) and Boyer et al. (2002). Howarth et al. (1996) report that riverine nitrogen exports are approximately 20% of net anthropogenic nitrogen inputs, and Boyer et al. (2002) report that riverine N exports are approximately 25% of total nitrogen inputs.

The BEANS annual nitrogen load is equivalent to an average nitrogen yield of 0.98 kg N/ha. As discussed in Chapter 3 of this thesis, nitrogen loss from aquifer denitrification is the least well-understood parameter in the model and varies both between and within watersheds as a function of nitrate concentration, the availability of dissolved organic carbon (DOC), and the presence of anaerobic conditions (Starr and Gillham 1993). The BEANS model assumes denitrification losses ranging from 30% to 40% depending on land use and the magnitude of nitrogen inputs (Rupert 1996, Valiela et al. 1997). Considering the area of land to which different denitrification losses are applied, the average watershed denitrification rate is approximately 37%, which is consistent both with denitrification losses measured in Idaho by the USGS (0-40%, Rupert 1996) and with the 35% loss rate that is used in the WBLMER model (Valiela et al. 1997, Brawley et al. 2000). While an average 37% loss is probably a good estimate of average watershed denitrification losses, it is possible that denitrification losses within certain areas of the watershed are as low as 0% and as high as 40%. To gain an understanding of the sensitivity of the total nitrogen load to the value of denitrification losses, we recalculate the total nitrogen load using a range of average watershed denitrification loss values.

Table 5-1: Sensitivity of Watershed Nitrogen Load and Yield to Denitrification Loss Values

Average Denitrification Loss	*Total Nitrogen Load (kg N)*	*Average Nitrogen Yield (kg N/ha)*
0%	1,050,000	1.54
5%	997,000	1.47
10%	946,000	1.39
15%	894,500	1.32
20%	843,000	1.24
25%	792,000	1.16
30%	740,000	1.09
35%	689,000	1.01
37%	**664,500**	**0.98**
40%	637,000	0.94

Increasing the denitrification loss value by 5% causes a corresponding decrease in the total nitrogen load of about 51,000 kg of nitrogen, which is 7.7% of the watershed nitrogen load estimated by the BEANS model. Since the total watershed area is about 680,000 ha, the average nitrogen yield increases by 0.08 kg N/ha for each 5% decrease in average denitrification loss.

To put the watershed nitrogen load estimated by the BEANS model into context, we compare our average nitrogen yield for the Wood River Valley watershed to nitrogen yields for 16 watersheds in the northeastern United States. These watershed nitrogen yields were calculated from streamflow nitrogen export data (Boyer et al. 2002).

Table 5-2: Riverine Nitrogen Export from 16 Watersheds in the Northeastern U.S. (adapted from Boyer et al. 2002).

Watershed	***Nitrogen Load (kg N/ha)***
BEANS	**0.98**
James	3.1
Penobscot	3.2
Kennebec	3.3
Saco	3.9
Androscoggin	4.0
Rappahannock	4.7
Hudson	5.0
Merrimack	5.0
Connecticut	5.4
Mohawk	8.0
Potomac	9.0
Delaware	9.6
Susquehanna	9.8
Blackstone	11
Charles	18
Schuylkill	18

The nitrogen yield for the Wood River Valley watershed is smaller than any of the yields calculated for these 16 Northeastern watersheds. The difference can be attributed to a much greater atmospheric deposition in the northeastern U.S. than in the northwestern U.S. (NADP 2003). The average annual atmospheric deposition rate for these 16 watersheds is 9.6 kg N/ha (Boyer et al. 2002) compared to 2 kg N/ha in the Wood River Valley. Additionally, Northeastern watersheds receive more annual precipitation than the Wood River Valley watershed (Figure 5-1). Average annual precipitation in the Wood River Valley is about 37 cm/year (NADP 2003); whereas, average precipitation in the 16 Northeastern watersheds is 111 cm/yr (Boyer et al. 2002), three times that occurring in the Wood River Valley.

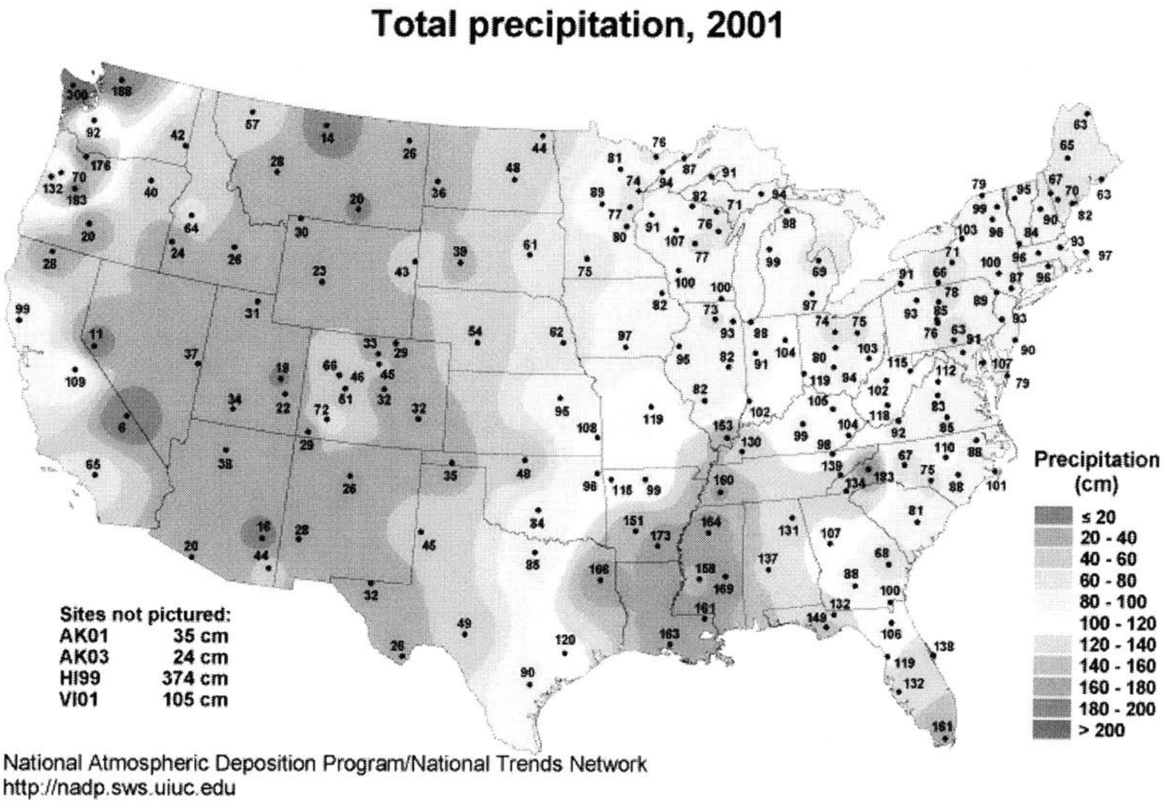


Figure 5-1: Variation in Annual Precipitation Rate

This difference in precipitation rate suggests that a certain nitrogen load in an eastern watershed would create a lower (approximately three times lower) surface water nitrogen concentration than it would in the Wood River Valley Watershed because the volume of water available to dissolve the nitrogen is greater in an eastern watershed.

The calculated nitrogen yield for the Wood River Valley Watershed (0.98 kg N/ha) is most similar to the nitrogen yields from the Penobscot (3.2 kg N/ha), Kennebec (3.3 kg N/ha), and James (3.14 kg N/ha) Watersheds in the Northeast. Because both the calculated nitrogen yield and the precipitation rate in the Wood River Valley are about one third of the nitrogen yields and precipitation rates in these eastern watersheds, we would expect to see relatively similar nitrogen concentrations in the Wood River Valley, Penobscot, Kennebec, and James Watersheds.

5.1. NITROGEN LOAD BY LAND USE

Table 5-3 summarizes the mass of nitrogen that originates from each of the land uses considered in the BEANS model: cropland, rangeland, residential, feed-lots, impervious surfaces, natural vegetation, and water

Table 5-3: Individual Land Use Contributions to the Total Watershed Nitrogen Load

Land Use	*Annual Nitrogen Load (kg)*
Rangeland	302,000
Cropland and Pasture	277,000
Residential	37,000
Feed Lots	13,000
Impervious Surfaces	18,000
Natural Vegetation	16,500
Water	500
Total	**664,500**

Rangeland and agricultural areas contribute the largest portion of the total mass, and their mass contributions are almost equal.

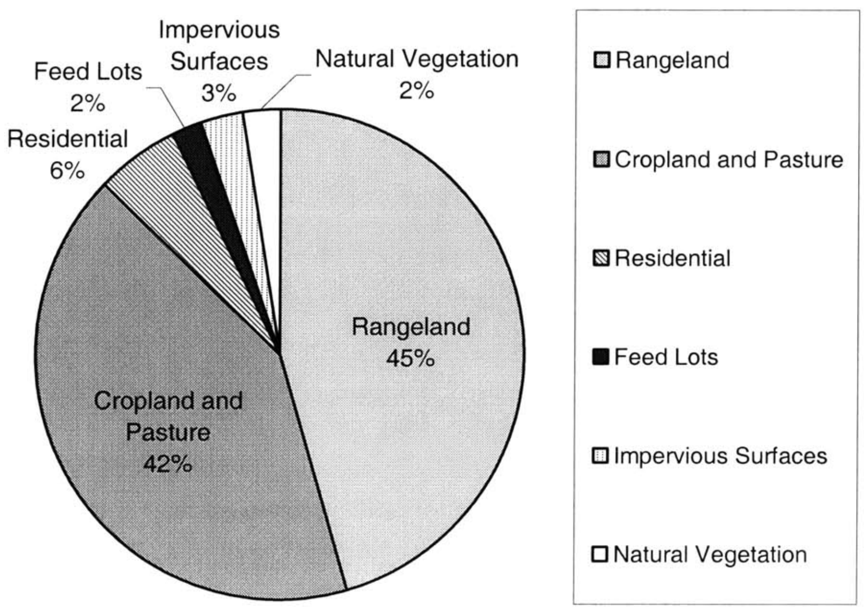


Figure 5-2: Percentage of Nitrogen Load Originating from Each Land Use Type

Overall, rangeland and cropland account for 87% of the total nitrogen load, suggesting that land use changes that convert land used for agriculture to other land uses that have smaller nitrogen yields will have a net effect of decreasing the total watershed nitrogen load. Alternatively, land use changes in areas that are currently residential areas, feed-lots, impervious surfaces, or natural vegetation will have smaller effects on the magnitude of the watershed nitrogen load.

In the GIS, we assign nitrogen yields to specific land areas within the watershed allowing comparison of the relative nitrogen contributions from specific land parcels.

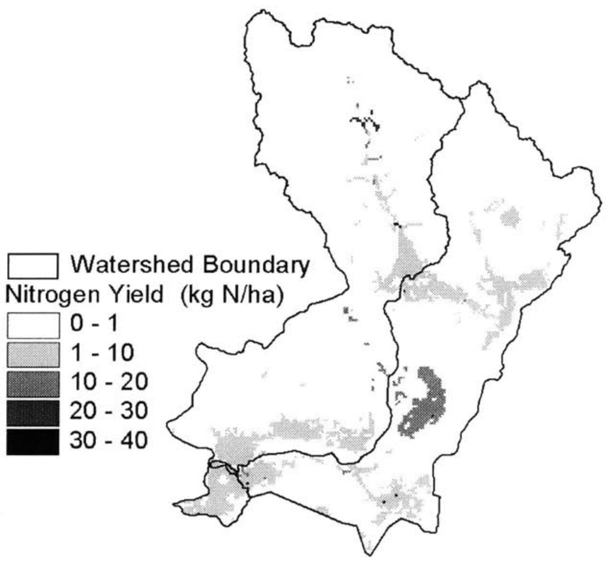


Figure 5-3: Nitrogen Yield throughout the Wood River Valley Watershed

As shown in Figure 5-3, the majority of the land area within the watershed contributes less than 1 kg N/ha, which is consistent with the average watershed nitrogen yield of 0.98 kg N/ha discussed earlier. Additionally, those land use types that contribute the largest total mass of nitrogen to the watershed do not necessarily have the largest nitrogen yields.

Agricultural land and rangeland introduce the largest total mass of nitrogen (Table 5-3) to the watershed, but feed-lots and residential areas have the highest nitrogen yields. Relative nitrogen yields of different land uses determine how converting land areas between land uses will affect the magnitude of the total nitrogen load to the watershed. Converting a given land area to a land use that has a higher nitrogen yield than the current use will increase the total nitrogen load to the watershed; likewise, converting a land area to a land use with a smaller nitrogen yield will reduce the watershed nitrogen load.

The nitrogen yields calculated by the BEANS model are on the low end of ranges reported in the literature and calculated by the USGS SPARROW model; however, the BEANS values appear consistent with these reported values when we consider that atmospheric deposition in the Wood River Valley is significantly lower than it is in other parts of the U.S. and that atmospheric deposition inputs are applied uniformly across all land use types in a watershed.

Table 5-4: Nitrogen Yield by Land Use

Land Use	***BEANS Nitrogen Yields (kg N/ha)***	***Literature Yields [a] (kg N/ha)***	***SPARROW Nitrogen Yields (kg N/ha)***
Feed-Lots	59.8	--	--
Residential	16.9	1.6 – 38.5	3.6 – 175
Cropland	4.4	0.8 – 79.6	2.2 – 42.5
Pasture	4.4	0.1 – 30.8	8.5 – 20.8
Rangeland	0.7	1.5 – 6.8	0.4 – 7.4
Impervious Surfaces	0.4	1.6 – 38.5	3.6 – 175
Natural Vegetation	0.1	0.1 – 10.8	1.8 – 11.2

-- No literature values reported. SPARROW model does not calculate nitrogen yields for agricultural feed lots.

[a] Total nitrogen yields reported in literature reviews (Alexander et al. 2000, Beaulac and Reckhow 1982, Frink 1991, Ritter 1988). The ranges listed for residential and impervious surfaces are those reported for urban areas in the literature.

Table 5-4 reports the average nitrogen yield calculated by the BEANS model for each land use category considered, which allows us to consider how land use changes in the watershed will affect the total watershed nitrogen load. Feed-lots have the highest nitrogen yield in the watershed, so converting land that is now used as a feed-lot into any other land use will reduce the watershed nitrogen load. Residential areas have the second greatest nitrogen yield, so converting residential areas into any land use other than feed-lots will reduce the total nitrogen load. The magnitude of these reductions will depend on the area of land that is converted from feed-lots or residential areas.

Despite the fact that the mass introduced from agricultural areas is much larger than the mass introduced from residential areas, converting agricultural areas into residential areas will increase the watershed nitrogen load because the nitrogen yield from residential areas is greater than that from agricultural areas. On average, each hectare of cropland or pasture that is converted to residential use will add an additional 12.5 kg of nitrogen to the watershed nitrogen load. Currently, there are approximately 84,500 ha of cropland and pasture in the watershed. If this entire area were converted to residential uses, the watershed nitrogen load would increase by 1,100,000 kg N/yr, which is a 160% increase. This increase assumes that population density on newly converted residential lands and the level of wastewater treatment would remain the same as they are currently in residential areas. The effect of population density and wastewater treatment on watershed nitrogen load are discussed in greater detail later in this thesis, as are the effects of other possible agricultural to residential land conversions.

Figure 5-3 illustrates that nitrogen yields are not constant across all parcels of the same type of land use. This is especially noticeable within the residential and agricultural land use categories. The average nitrogen yield for residential areas is 17 kg N/ha, but the specific values range from 11 kg N/ha for residential areas connected to the Hailey wastewater treatment plant to 41 kg N/ha for residential areas in the Town of Bellevue. Differences in the magnitude of nitrogen yields from different residential areas illustrate not only variations in the level of wastewater treatment but also differences in the residential population densities.

Table 5-5: Variation in Nitrogen Yields from Residential Land Areas

Residential Area	***Nitrogen Yield (kg N/ha)***	***Residential Population Density (people/ha)***	***Per Capita N Load (kg N/person)***
Hailey	11	20	0.43
Septic	12	10	1.1
Ketchum/Sun Valley	33	7	4.3
Bellevue	41	18	2.3

The City of Hailey has the highest residential population density, but because its tertiary wastewater treatment plant is effective at removing nitrogen, residential areas within the city maintain the lowest nitrogen yield. Bellevue, which has a similar residential population density to Hailey, has the highest nitrogen yield because its wastewater treatment facility is less efficient at removing nitrogen than the Hailey facility. The Ketchum/Sun Valley treatment plant, which is the least efficient at removing nitrogen on a per capita basis, has a lower nitrogen yield than Bellevue because its residential population density is less than half of Bellevue's. In a linear regression over all the residential land use area in the watershed, nitrogen yield is not correlated to population density ($R^2 = 0.0$) but is positively correlated to per capita nitrogen load ($R^2 = 0.55$).

The average nitrogen yield for agricultural areas is 4.4 kg N/ha, but the specific values range from 0.9 kg N/ha for agricultural area 4 to 7.0 kg N/ha for agricultural area 5. Differences in the magnitude of nitrogen loads from different agricultural areas illustrate variations in localized fertilizer application rates, in the percentage of the area dedicated to nitrogen fixing crops like alfalfa and dry beans, and in the percentage of the area dedicated to pasture, which determines the magnitude of the nitrogen input from animal waste.

Table 5-6: Variation in Nitrogen Yield from Agricultural Land Areas

Agricultural Area	*Nitrogen Yield (kg N/ha)*	*Fertilizer Application Rate (kg N/ha)*	*% N-fixing*	*%-Pasture*
Area 4	0.9	0	0%	100%
Area 3	6.2	112	0%	0%
Area 1	2.0	6	18%	82%
Area 7	4.0	56	9%	53%
Area 2	4.3	49	23%	49%
Area 6	6.7	60	66%	0%
Area 5	7.0	33	100%	0%

Nitrogen yield is negatively correlated to pasture percentage ($R^2 = 0.51$). This is consistent with the fact that area 4, which is dedicated entirely to pasture, has the lowest nitrogen yield. Nitrogen yield is most strongly correlated to nitrogen-fixing percentage ($R^2 = 0.95$). This correlation is positive, so it follows that area 5, which is dedicated entirely to nitrogen-fixing crops, has the highest nitrogen yield. Fertilizer application rate is not correlated to nitrogen yield ($R^2 = 0.0$).

The fact that agricultural nitrogen yields are positively correlated to nitrogen-fixing percentage and negatively correlated to pasture percentage implies how land use changes within agricultural areas will impact the total nitrogen yield from these areas. For example, converting an area that is currently used for pasture into cropland will increase the nitrogen yield, and this effect will be greater if the crop is a nitrogen-fixing crop like alfalfa or dry beans. Likewise, converting an area that is currently used as cropland into pasture will decrease the nitrogen yield, and converting an area from alfalfa or dry beans to pasture will provide an even greater decrease in total nitrogen yield.

5.2. NITROGEN LOAD BY SOURCE

Rather than describing the total nitrogen load from a land use perspective, the nitrogen load can also be categorized based on the sources from which the nitrogen originates (i.e.: atmospheric deposition, fertilizer applications, nitrogen fixation, animal waste, and wastewater). Figure 5-4 illustrates the relative contribution of nitrogen sources to the nitrogen load from each land use type.

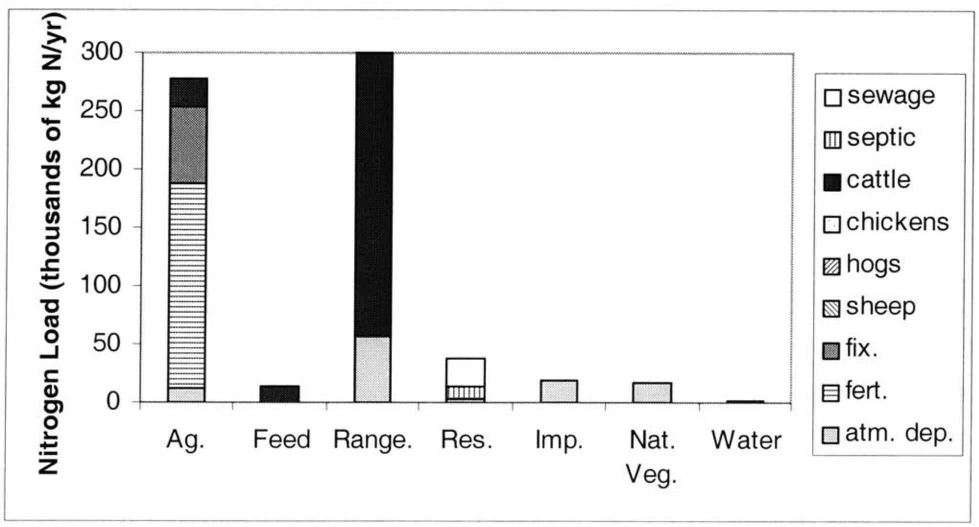


Figure 5-4: Relative Magnitude of Source Nitrogen Loads from Each Land Use Type (Ag. = cropland and pasture; Feed = feed-lots; Range. = rangeland; Res. = residential land; Imp. = impervious surfaces; Nat. Veg. = natural vegetation; fix. = fixation; fert. = fertilizer applications; atm. dep. = atmospheric deposition)

Figure 5-4 shows that fertilizer applications and cattle waste are the largest sources of nitrogen to the watershed. Table 5-7 summarizes the mass of nitrogen that originates from each nitrogen source.

Table 5-7: Individual Source Contributions to the Total Watershed Nitrogen Load

Nitrogen Source	***Annual Nitrogen Load (kg)***
Cattle Waste	273,000
Fertilizer	188,000
Atmospheric Deposition	100,000
Nitrogen Fixation	69,000
Sewage	23,000
Septic Systems	9,800
Sheep Waste	760
Hog Waste	150
Chicken Waste	10

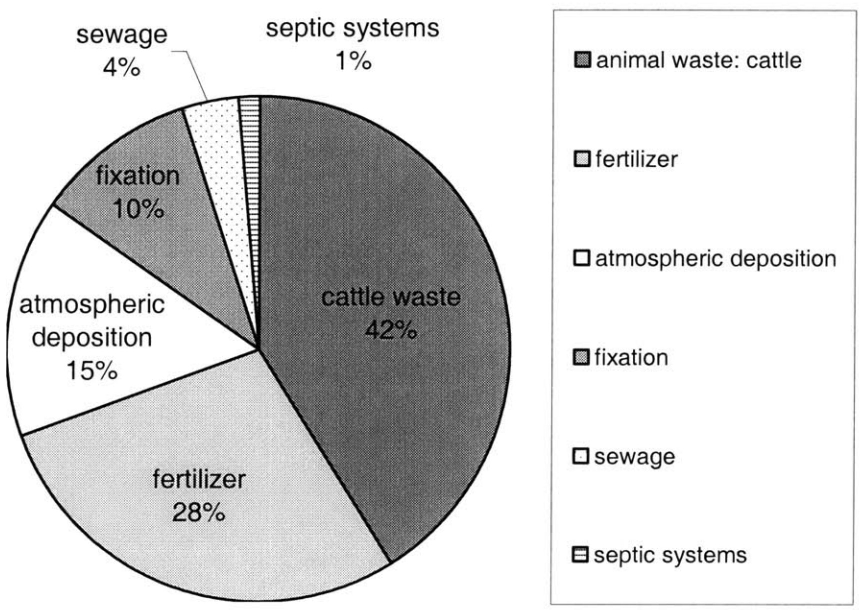


Figure 5-5: Percentage of Nitrogen Load Originating from Each Source

Because the combined inputs from septic and sewage contribute only 5% of the total nitrogen load, changes in watershed population will increase the watershed nitrogen load but the magnitude of this increase will be relatively small. The exception may be the area of the Big Wood River Watershed north of the Town of Bellevue. This sub-watershed houses the majority of the watershed population (79%) but constitutes only 33% of the total watershed area; therefore, it has a much higher average population density than the rest of the watershed. Additionally, this section of the watershed has undergone rapid population growth in recent decades, so it is possible that nitrogen inputs from residential sources will have a larger effect on the magnitude of the nitrogen load to this sub-watershed.

5.3. UPPER VALLEY WATERSHED NITROGEN LOAD

The BEANS model is applied separately to the section of the Big Wood River Watershed that is north of the Town of Bellevue (Figure 5-6).

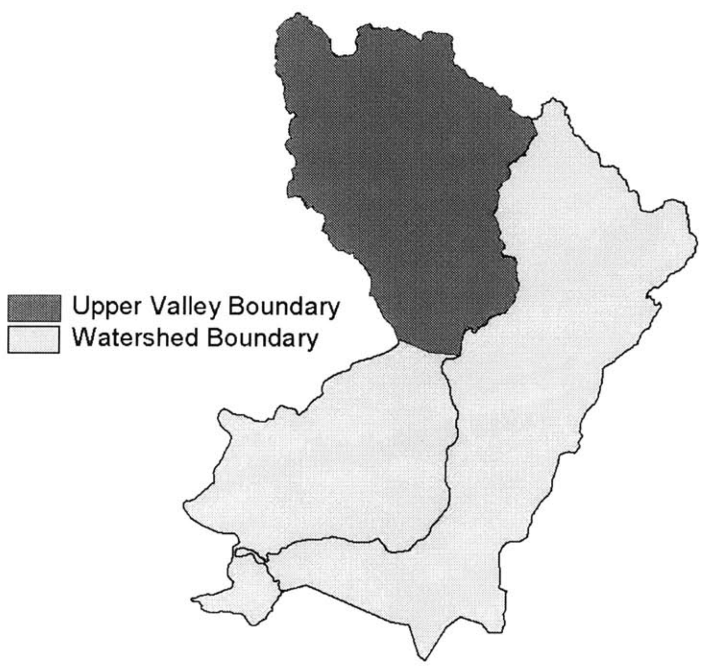


Figure 5-6: Location of the Upper Valley Sub-Watershed

This sub-watershed contains the cities of Ketchum, Sun Valley, and Hailey.

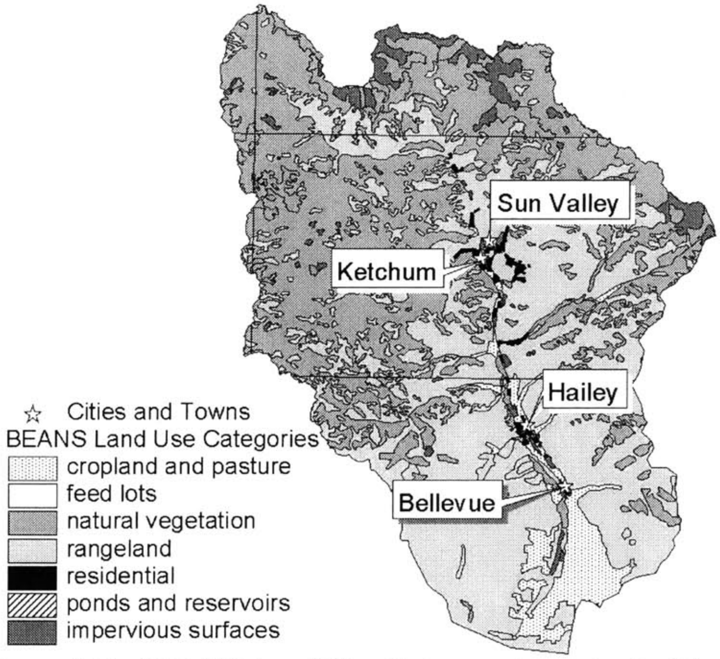


Figure 5-7: BEANS Land Use Category Areas in the Upper Valley Watershed

The total land area in the Upper Valley Watershed is approximately 223,000 ha, which is 33% of the 680,000 ha in the entire watershed.

Table 5-8: Relative Land Use Areas in the Upper Valley Watershed Compared to the Entire Watershed

Land Use	***Upper Valley Watershed Area (ha)***		***Wood River Valley Watershed Area (ha)***	
Rangeland	106,000	47%	428,000	63%
Natural Vegetation	97,300	44%	115,000	17%
Cropland and Pasture	10,800	5%	84,500	12%
Impervious Surfaces	7,300	3%	48,200	7%
Residential Areas	1,700	1%	2,200	0%
Ponds and Reservoirs	34	0%	2,000	0%
Feed lots	6	0%	215	0%

Similar to the watershed as a whole, rangeland is the most common land use within the Upper Valley Watershed but is less prevalent than it is on the watershed scale. Rangeland is almost equal in area to natural vegetation, which is much more prevalent in the upper valley because of the presence of the Sawtooth National Recreation Area. Agricultural land and impervious surfaces are both less common in the Upper Valley, but residential areas are more common, constituting 1% of the sub-watershed area.

The total nitrogen load to the Upper Valley Watershed is estimated by the BEANS model to be approximately 165,000 kg N/year. This mass is approximately 25% of the total watershed nitrogen load. Because the Upper Valley is 33% of the total watershed area, this sub-watershed contributes a smaller than average nitrogen load compared to other sections of the watershed. The average nitrogen yield for the Upper Valley Watershed is 0.74 kg N/ha compared to the average nitrogen yield for the entire watershed of 0.98 kg N/ha.

The Upper Valley nitrogen load originates from all seven land-use types, but the relative magnitudes of land use nitrogen loads are different than they are in the entire watershed. Table 5-9 summarizes the mass of nitrogen that originates from each land use in the Upper Valley.

Table 5-9: Individual Land Use Contributions to the Upper Valley Watershed Nitrogen Load

Land Use	***Annual Nitrogen Load (kg)***
Rangeland	80,900
Cropland and Pasture	34,100
Residential	31,700
Natural Vegetation	15,200
Impervious Surfaces	2,800
Feed Lots	350
Water	8

Rangeland, agricultural land, and residential areas are the largest contributors of nitrogen, as they are in the whole watershed model. The magnitudes of nitrogen inputs from rangeland and agricultural land are approximately equal in the watershed model, but in the Upper Valley cropland contributes less than half the mass of nitrogen that rangeland contributes. Additionally, residential land remains the third largest nitrogen contributor, but the mass of nitrogen from residential land in the Upper Valley Watershed represents 86% of the total watershed residential nitrogen contribution (37,000 kg N/yr). This is consistent with the fact that 19,000 people, 79% of the total watershed population, live within the Upper Valley Watershed boundary.

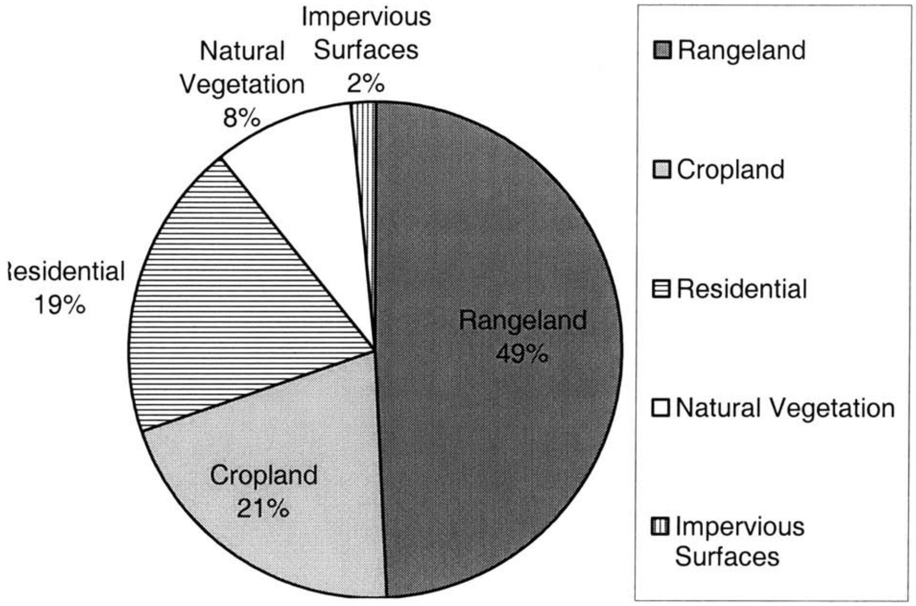


Figure 5-8: Percentage of Upper Valley Watershed Nitrogen Load Originating from Each Land Use Type

In the Upper Valley, residential areas are responsible for 19% of the total nitrogen load compared to only 6% of the entire watershed nitrogen load. Because of the increased significance of residential nitrogen sources when considering only the Upper Valley, it is possible that continuing development and population growth in this area could have a greater effect on the magnitude of the sub-watershed nitrogen load and consequently local surface water quality.

Considering the Upper Valley nitrogen load from a source perspective, the relative magnitude of the nitrogen contribution from wastewater noticeably increases.

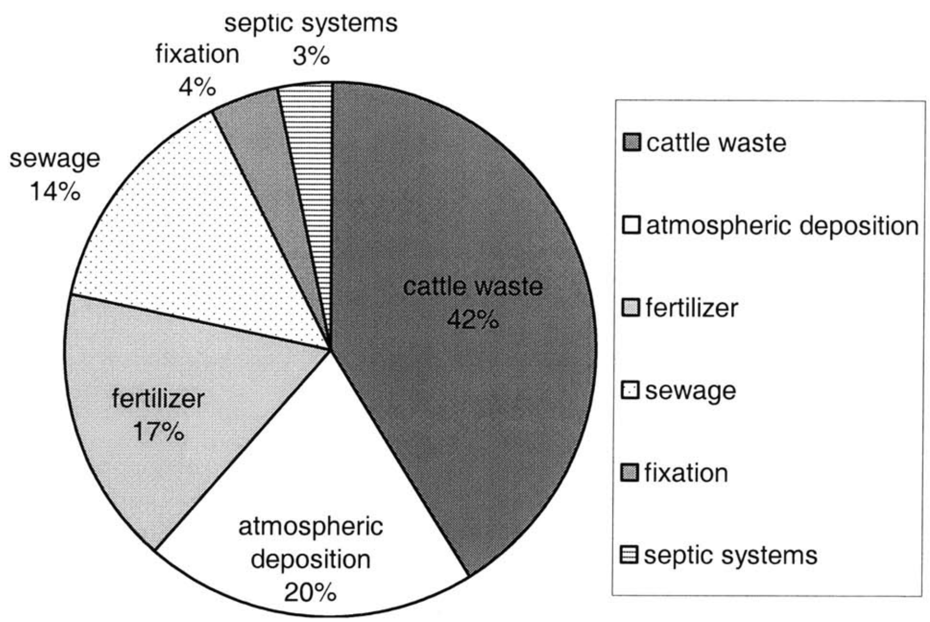


Figure 5-9: Percentage of Upper Valley Watershed Nitrogen Load Originating from Each Source

The septic system nitrogen load is 3% of the total nitrogen load in the Upper Valley compared to 2% in the entire watershed, and the load from wastewater treatment plants is 14% instead of only 4%.

In this sub-watershed, where wastewater nitrogen contributes 17% of the total nitrogen load, the level of wastewater treatment can have a significant effect on the magnitude of the total nitrogen load. Currently in the Upper Valley, 33% of the population receives tertiary treatment of their wastewater from the Hailey treatment plant, which has an annual per capita nitrogen yield of 0.43 kg N/person. Another 31% of the population relies on septic systems, which have an annual per capita nitrogen yield of 1.1 kg N/person. The remaining 36% of the population receives secondary treatment of their wastewater. The Bellevue treatment plant has an annual per capita nitrogen yield of 2.3 kg N/person, while the Ketchum/Sun Valley yield is 4.3 kg N/person.

The BEANS model estimates that the current population contributes a total mass of 28,700 kg of nitrogen to the Big Wood River each year from combined wastewater inputs. To illustrate the importance of the level of wastewater treatment to nitrogen load magnitude, we consider how the Upper Valley nitrogen load would change if the distribution of the population across different levels of wastewater treatment were to change. Table 5-10 shows the magnitudes of the total Upper Valley Watershed nitrogen load and the wastewater nitrogen load if the current population was switched entirely to one type of wastewater treatment plant.

Table 5-10: Effects of Changing Level of Wastewater Treatment on Upper Valley Watershed Nitrogen Load

Level of Wastewater Treatment	***Upper Valley Total Nitrogen Load (kg)***	***Upper Valley Wastewater Nitrogen Load (kg)***
Current Condition	165,000	28,700
All Secondary (Ketchum/Sun Valley)	201,900	65,600
All Secondary (Bellevue)	171,600	35,300
All Septic	153,600	17,300
All Tertiary	142,800	6,500

Switching the entire population over to secondary treatment similar to that performed at the Ketchum/Sun Valley or the Bellevue plants would cause an increase in the Upper Valley nitrogen load. This increase would be smaller for the Bellevue level of treatment because the per capita nitrogen yield from Bellevue is smaller. Switching the entire current population over to septic systems or tertiary wastewater treatment like that performed at the Hailey treatment plant would decrease the Upper Valley nitrogen load.

5.4. NORTHERN VALLEY WATERSHED NITROGEN LOAD

To investigate the effects of residential nitrogen inputs on an even smaller scale, we sub-divide the watershed again to consider a section of the Big Wood River Valley beginning at its northern border and extending south to include the cities of Ketchum and Sun Valley. This sub-watershed is referred to as the Northern Valley Watershed (Figure 5-10).

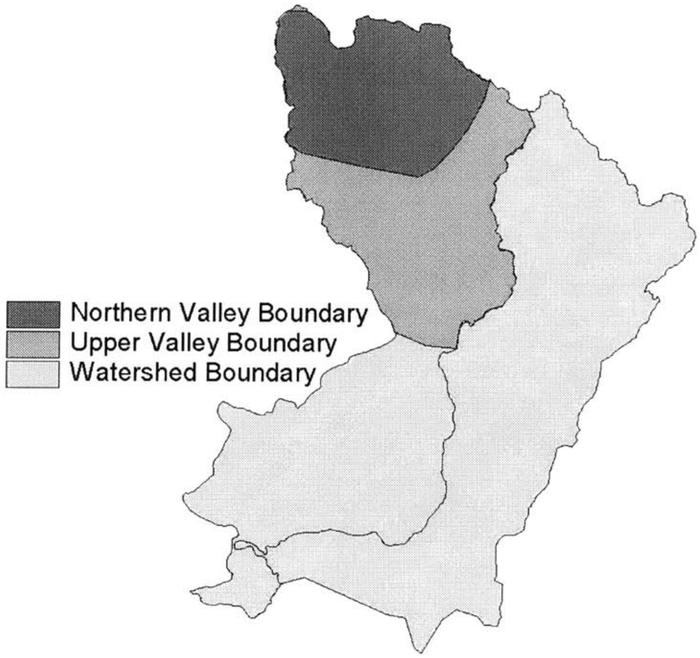


Figure 5-10: Location of the Northern Valley Sub-Watershed

There are two reasons for running the BEANS model this section of the watershed separately from the rest of the watershed:

1. Most of the recent population growth in the watershed has occurred in the Ketchum/Sun Valley area.
2. The valley is very narrow just south of Ketchum, which funnels all of the groundwater and nitrogen that is input into the watershed above this point through a small cross-sectional area of the aquifer. This local hydrogeology could cause elevated nitrogen concentrations at the outlet of this sub-watershed.

The total land area in the Northern Valley is about 101,000 ha, or 15% of the total watershed area.

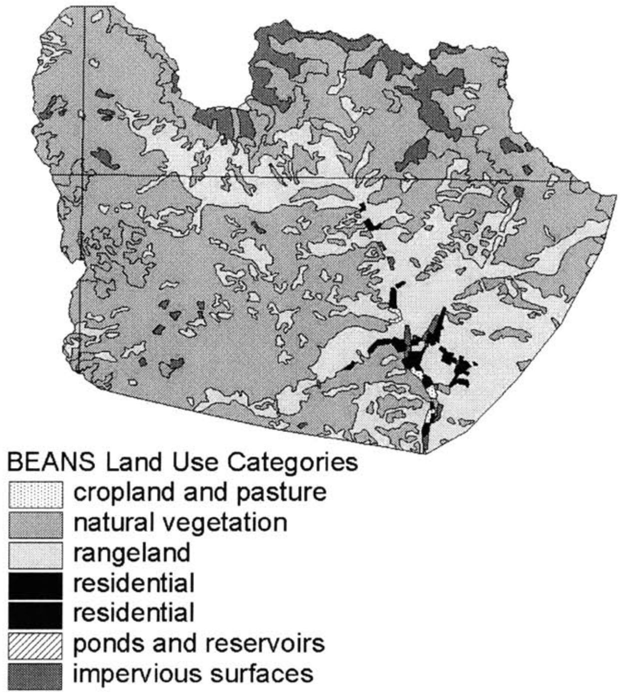


Figure 5-11: BEANS Land Use Areas in the Northern Valley Watershed

The relative magnitude of different land areas in the Northern Valley is noticeably different than it is in both the Upper Valley and the overall watershed. Rangeland is no longer the most abundant land use. Natural vegetation is the most common land use, and no feed-lots are present.

Table 5-11: Relative Land Use Areas in the Northern Valley Compared to the Entire Watershed

Land Use	***Northern Valley Watershed Area (ha)***		***Wood River Valley Watershed Area (ha)***	
Rangeland	29,100	29%	428,000	63%
Natural Vegetation	65,200	65%	115,000	17%
Cropland and Pasture	210	0%	84,500	12%
Impervious Surfaces	5,400	5%	48,200	7%
Residential Areas	1,000	1%	2,200	0%
Ponds and Reservoirs	30	0%	2,000	0%
Feed-lots	0	0%	215	0%

Rangeland and agricultural areas are noticeably less common in the Northern Valley than they are in the overall watershed. Residential areas are more common, but have a similar land area to that observed in the Upper Valley Watershed.

The total nitrogen load to the Northern Valley Watershed estimated by the BEANS model is 55,600 kg N/year. This mass is approximately 8% of the total watershed nitrogen load. Because the Northern Valley is 15% of the total watershed area, this watershed contributes a smaller than average nitrogen load compared to other sections of the watershed. Its relative nitrogen load is also smaller than that contributed by the Upper Valley Watershed. The average nitrogen yield for the Northern Valley Watershed is 0.55 kg N/ha compared to 0.74 kg N/ha and 0.98 kg N/ha for the Upper Valley Watershed and entire watershed respectively.

Because there are no feed-lots in the Northern Valley, the nitrogen load for this sub-watershed originates from only six of the land use categories considered in the BEANS model. Table 5-12 summarizes the mass of nitrogen that originates from each land use in the Northern Valley.

Table 5-12: Individual Land Use Contributions to the Northern Valley Watershed Nitrogen Load

Land Use	***Annual Nitrogen Load (kg)***
Rangeland	22,300
Cropland and Pasture	540
Residential	20,600
Natural Vegetation	10,200
Impervious Surfaces	2,000
Water	7

Rangeland and residential areas are the largest contributors of nitrogen to the Northern Valley Watershed. Natural vegetation areas contribute about half the mass from either of the previous land uses. While cropland and pasture combined is the second largest contributor to both the Upper Valley and the total watershed, these land uses contribute only a small mass of nitrogen to this sub-watershed. Instead, residential areas are now the second largest contributor of nitrogen. The mass input from residential areas in the Northern Valley is 37% of the total watershed residential nitrogen load, but the sub-watershed contains only about 25% of the total watershed population. This is to say that a quarter of the watershed population contributes more than half of the residential nitrogen load to the watershed as a whole.

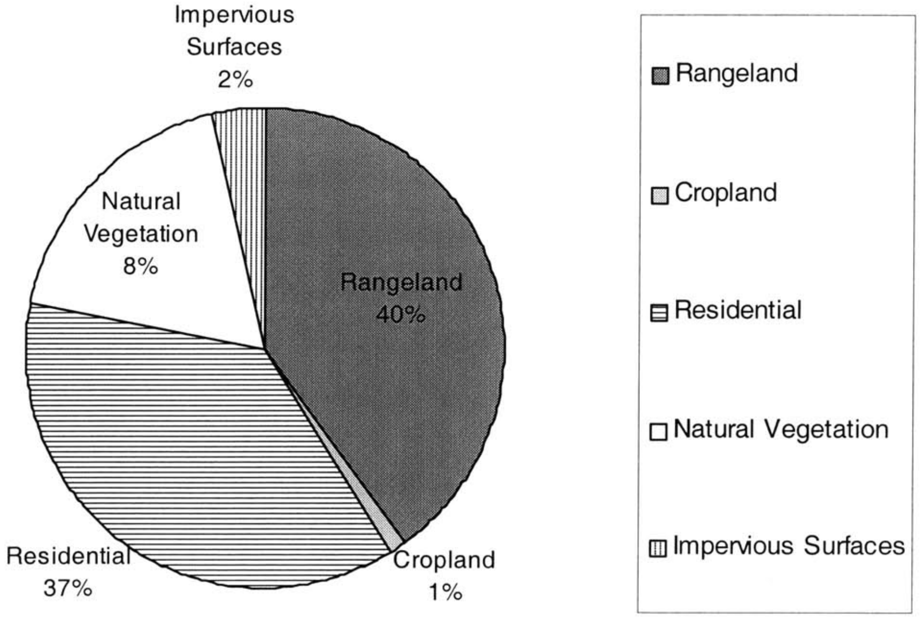


Figure 5-12: Percentage of Northern Valley Watershed Nitrogen Load Originating from Each Land Use Type

In the Northern Valley, residential areas are responsible for 37% of the total nitrogen load compared to 19% and 6% in the Upper Valley and entire watershed respectively. This increase in the relative magnitude of residential nitrogen largely has to do with an increase in the portion of the nitrogen load coming from wastewater inputs.

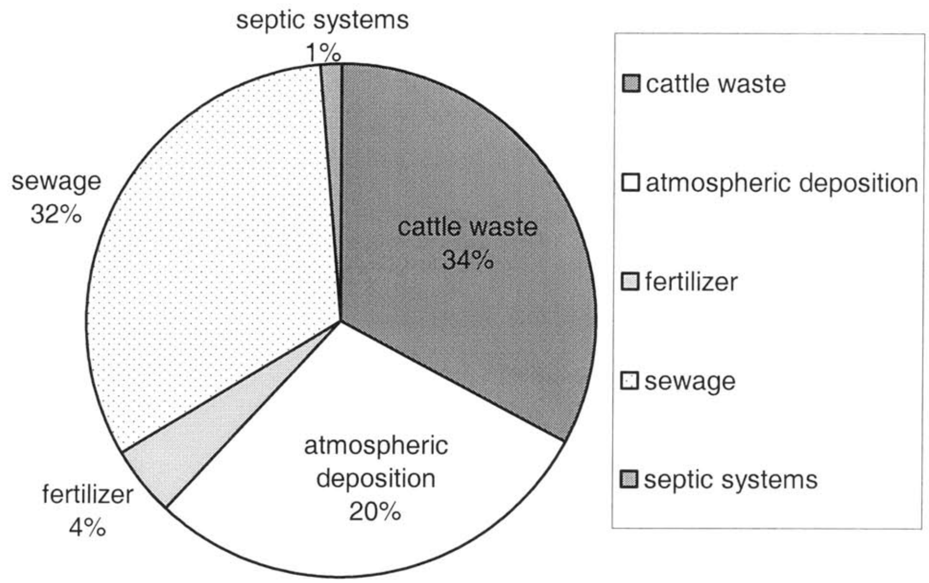


Figure 5-13: Percentage of Northern Valley Watershed Nitrogen Load Originating from Each Source

Septic systems contribute only 1% of the Northern Valley nitrogen load, which is a smaller percentage than its relative contribution to the Upper Valley and overall watershed; however, the sewage input grows to 32% of the total nitrogen load.

In this sub-watershed, where wastewater contributes 33% of the total nitrogen load, the level of wastewater treatment can drastically affect the magnitude of the total nitrogen load. Currently in the Northern Valley, 14% of the population relies on on-site septic systems for treatment of its wastewater and the remaining 86% is connected to the Ketchum/Sun Valley wastewater treatment plant. The Ketchum/Sun Valley treatment plant employs the least effective method of nitrogen removal currently in use in the Wood River Valley. The annual per capita nitrogen yield from this wastewater treatment plant is 4.3 kg N/person.

The Northern Valley Watershed nitrogen load would decrease if the population now using the Ketchum/Sun Valley treatment plant were to convert to any other available wastewater treatment method. To illustrate the importance of wastewater treatment level to the magnitude of the Northern Valley nitrogen load, we consider how this nitrogen would change in response to changes in wastewater treatment level. Table 5-13 shows the magnitude of the Northern Valley nitrogen load and the wastewater nitrogen load for different levels of wastewater treatment.

Table 5-13: Effects of Changing Level of Wastewater Treatment on Northern Valley Watershed Nitrogen Load

Level of Wastewater Treatment	***Northern Valley Total Nitrogen Load (kg)***	***Northern Valley Wastewater Nitrogen Load (kg)***
Current Condition	55,600	18,700
All Secondary (Ketchum/Sun Valley)	57,700	20,800
All Septic	42,400	5,500
All Tertiary	39,000	2,100

Switching the entire population over to septic systems would reduce the current nitrogen load to the Northern Valley by 24%, but could potentially cause extremely high local groundwater nitrogen concentrations. Local septic system conditions are discussed later in this thesis. If the entire population were switched over to tertiary wastewater treatment, the total nitrogen load would be reduced by 30%. If the population that is currently on septic stayed on septic and the Ketchum/Sun Valley plant were upgraded to a tertiary plant like that in Hailey, the total nitrogen load to the Northern Valley would be reduced by 29% to 39,500 kg N/year.

6. OBSERVED NITROGEN CONCENTRATIONS AND LOADS

The United States Geologic Survey (USGS) maintains a stream gaging and water quality monitoring station at the outflow of the Wood River Valley Watershed. The station is named Malad River near Gooding, Idaho (station # 13152500) and is located just below the confluence of the Big Wood and Little Wood Rivers.

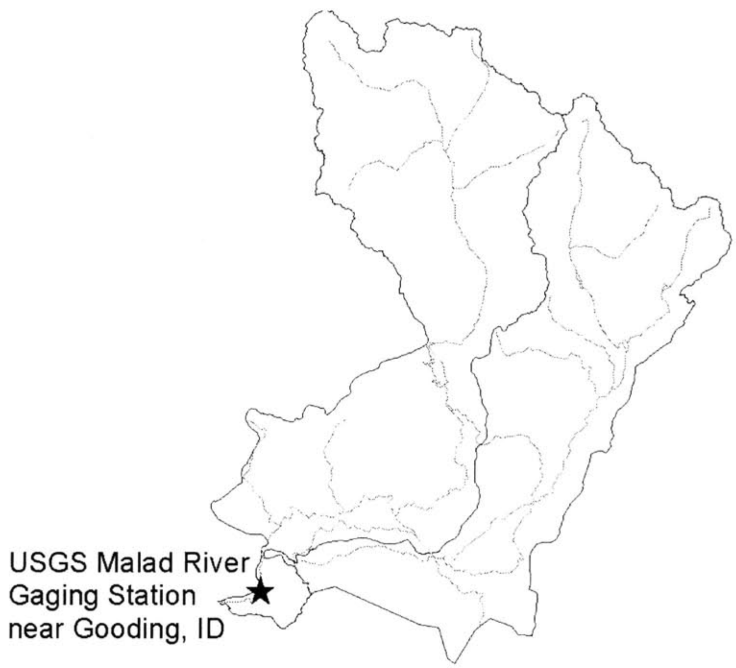


Figure 6-1: Location of USGS Malad River Gaging Station near Gooding Idaho

We use in-stream nitrogen concentrations measured at this station and daily streamflow data to calculate the mass of nitrogen leaving the watershed through streamflow each year. To evaluate the accuracy of the BEANS model, we compare our predicted watershed nitrogen load to masses calculated from USGS nitrogen concentration and streamflow data.

The nitrogen species measured at the Malad River are nitrate (NO_3^-) and Total Kjeldahl Nitrogen (TKN), which includes organic nitrogen and ammonia (NH_3). Total nitrogen concentrations reported here are the sum of nitrate and TKN concentrations in units of mg N/L. Generally, nitrogen concentrations are measured once monthly. There are only six years in which the USGS measured in-stream nitrogen concentrations during at least six months of the year. Table 6-1 lists the nitrogen data available from these six years.

Table 6-1: Available Total Nitrogen Concentration Data from USGS Malad River Gaging Station

Total Nitrogen Concentrations (mg N/L)												
	Jan.	**Feb.**	**Mar.**	**Apr.**	**May**	**June**	**July**	**Aug.**	**Sep.**	**Oct.**	**Nov.**	**Dec.**
1993	--	--	--	1.07	0.65	1.26	0.35	0.46	0.45	0.25	0.54	1.3
1994	1.3	1.7	1.07	0.95	0.55	0.45	0.35	0.35	0.35	0.55	--	1.8
1995	2.01	0.99	0.53	1.13	0.56	0.63	--	0.45	--	--	--	--
1996	--	--	--	0.81	0.59	0.35	0.25	2.2	.57	--	--	--
1997	--	--	--	0.88	0.51	0.53	0.48	0.58	0.39	--	--	--
2000	--	--	--	0.64	0.55	0.50	0.38	0.37	0.43	--	--	--

--: No data available.

The data set is rather limited with respect to calculating the mass of nitrogen leaving the watershed each year. Data are most often missing during the winter months, November to April. In order to calculate annual nitrogen masses, we must make some assumptions about nitrogen concentrations during the times when we do not have specific concentration measurements.

The first assumption we make involves nitrogen concentrations during a month in which an actual measurement has been made. Within a given month, we assume that nitrogen concentrations on days other than the day on which the measurement was made are similar to concentrations on the measured day. For example, we assume that the in-stream nitrogen concentration is 0.64 mg N/L for every day in April 2000. The second assumption involves nitrogen concentrations during months in which no measurement is made. Because no specific data points are available, we assign to these months average nitrogen concentrations derived from the other months of the calendar year when data are available. Rather than using an average annual concentration, we use an average winter concentration for missing concentrations in November through April and an average summer concentration for missing concentrations in May through October. Seasonal averages rather than annual averages are used because variation in nitrogen concentrations shows a seasonal pattern (Figure 6-2).

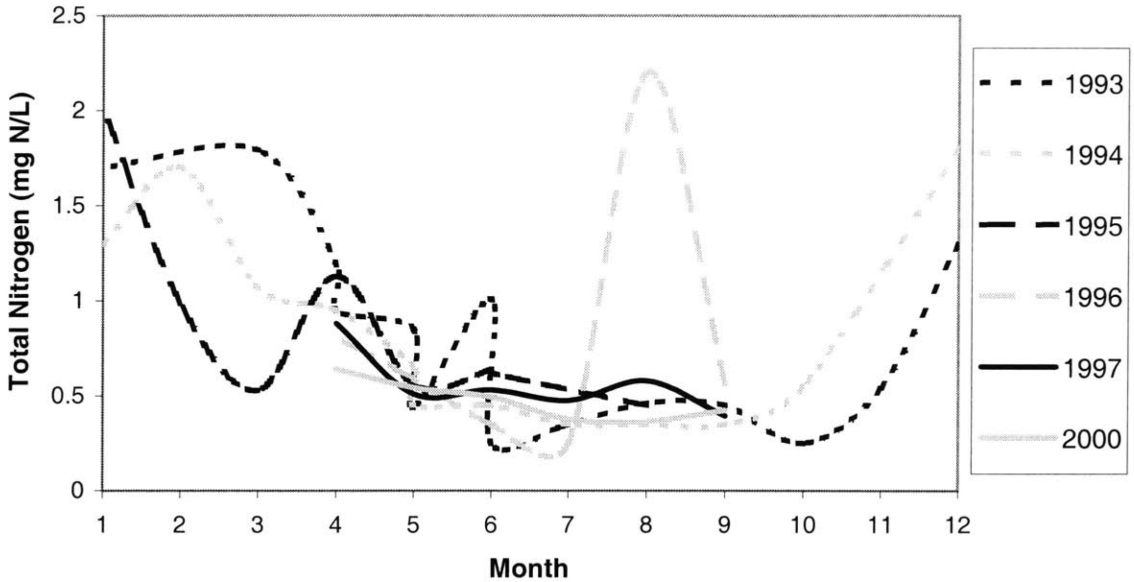


Figure 6-2: Seasonal Variation in Nitrogen Concentration

With the exception of August 1996, nitrogen concentrations are lower in the summer than in the winter. For the few years in which we have winter data, concentrations are highest during December, January, and February, but they seem to increase in November and do not significantly decline until April. In those years where April is the only winter concentration measured (1996, 1997, 2000), the average winter concentration is equal to the April concentration and, consequently, is likely to be considerably lower than true concentrations.

Using seasonally averaged concentrations rather than annual average concentrations is also supported by more comprehensive monitoring that has been conducted in Idaho as part of the National Water Quality Assessment Program (Williamson et al. 1998). This USGS study, summarizing water quality in the Central Columbia River Plateau in Washington and Idaho between 1992 and 1995, reports that nitrogen concentrations are highest during winter months. The explanation for this seasonal pattern is that, in the winter, irrigation water is not delivered to streams and storms large enough to produce runoff are rare; consequently, streamflow is low and groundwater is the predominant source of nitrogen to surface waters (Williamson et al. 1998). When the main source of nitrogen to surface waters is groundwater discharge, instream concentrations are highest because there is little dilution by irrigation canals or return flows (Williamson et al. 1998). Similar factors likely control instream nitrogen concentration in the Wood River Valley Watershed, which explains the seasonal concentration variations observed in the watershed.

Using the available data, the assumptions described above, and daily stream flows, we calculate the mass of nitrogen leaving the watershed each year. Table 6-2 lists these calculated masses.

Table 6-2: Baseflow Nitrogen Masses Calculated from Available USGS Data

	Baseflow Nitrogen Mass (kg N/yr)	***Percent Difference from BEANS Nitrogen Load***
1993	181,500	73%
1994	70,300	89%
1995	224,900	66%
1996	282,600	57%
1997	464,200	30%
2000	82,200	87%

Most of these calculated masses are the same order of magnitude as the BEANS mass of 664,500 kg N/year, but all of the masses are lower than the BEANS model estimate. Calculating annual nitrogen mass using streamflow and in-stream concentration data likely underestimates the true nitrogen mass for two reasons: 1) the available data set is thinnest during winter months which appear to have elevated nitrogen concentrations and 2) the data set does not include peak concentrations or peak flows that are produced during storm events. Smith et al. (1996) and Buffam et al. (2001) have both shown that in-stream nitrogen concentrations are greater during stormflow than during baseflow. Buffam et al. (2001) credit this concentration change during storms to the assumption that hydrologic source areas and flow paths are different between baseflow and stormflow. On average, dissolved organic nitrogen (DON) concentrations doubled during stormflow, and nitrate concentrations increased by slightly more than double (Buffam et al. 2001).

To incorporate the effects of stormflow on annual nitrogen mass, we doubled nitrogen concentrations during storms for each year and recalculated nitrogen mass. Storms were defined in two ways: 1) streamflow greater than 1.5 times the average annual flow rate and 2) streamflow greater than twice the average annual flow rate.

Table 6-3: Baseflow and Stormflow Nitrogen Masses

	Baseflow Nitrogen Mass (kg N/yr)	***Stormflow Nitrogen Mass (kg N/yr) Storms = 1.5 x flow***	***Stormflow Nitrogen Mass (kg N/yr) Storms = 2 x flow***
1993	181,500	257,200	225,800
1994	70,300	123,700	121,400
1995	224,900	338,200	331,400
1996	282,600	512,500	479,400
1997	464,200	744,100	735,000
2000	82,200	146,300	136,000

After adjusting for stormflow, all of the annual nitrogen masses are the same order of magnitude as the BEANS mass, and the BEANS mass falls within the range of calculated values. The calculated concentrations are still extremely variable from year to year. It is unlikely that this variation is the result of drastic changes in nitrogen inputs to the watershed from one year to the next because land use probably remained relatively constant over the seven-year period. Rather, this variation may stem from the fact that a year may be an arbitrary length of time with respect to nitrogen export from the watershed. The BEANS model assumes that all of the nitrogen input into the watershed in a given year is either lost or retained during transport or is flushed from the watershed through streamflow. This assumption may not be true for every year. In years when water is limited, there may not be enough water available to flush the entire annual mass of nitrogen out of the watershed. In years when water is abundant, the river may flush not only nitrogen inputs from the current year but also residual nitrogen from previous years. This argument is supported by the fact that calculated annual nitrogen loads are directly correlated to average annual stream flow.

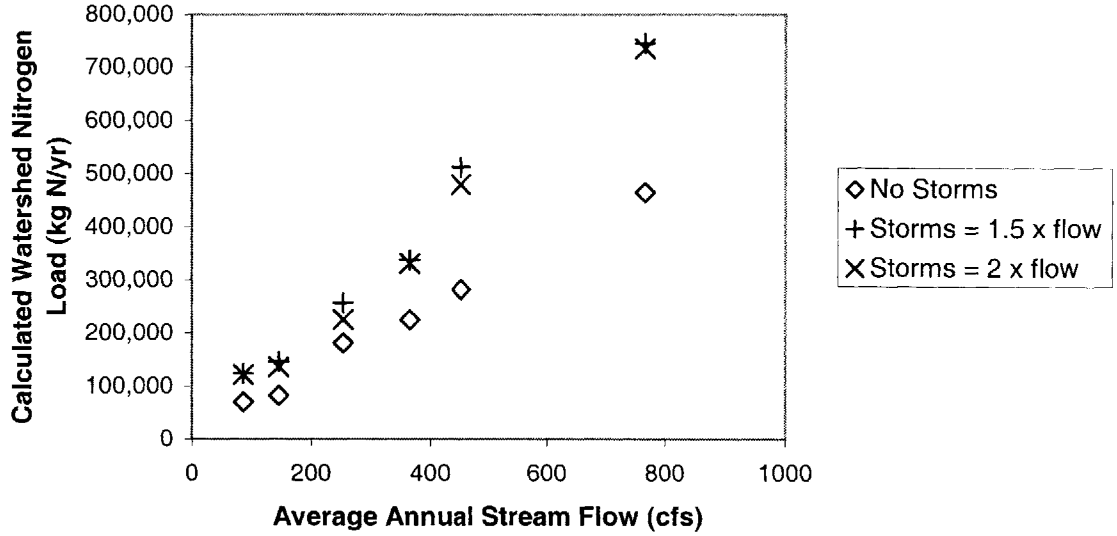


Figure 6-3: Correlation between Calculated Annual Nitrogen Loads and Average Annual Streamflow

Both stormflow and baseflow nitrogen loads increase with increasing average flow rate ($R^2 = 0.99$). This relationship between nitrogen load and stream flow stems from the fact that nitrogen concentrations in the rivers are relatively constant over time. Total nitrogen concentrations observed over the last ten years range from 0.25 mg N/L to 2.2 mg N/L, so as stream flow increases, nitrogen load increases by a relatively constant amount. This small variability likely results from the fact that the majority of nitrogen sources in the watershed are non-point sources. It is difficult to predict the magnitude of annual nitrogen loads from non-point sources because these loads are tied to weather events. "While point sources produce fairly regular flows across seasons and even years, non-point sources do not. Loads are highest during rainy seasons and years with high precipitation, and conversely lower at other times" (Faeth 2000). The correlation between nitrogen load and steam flow makes predicting the annual mass of nitrogen exported from the watershed in a given year difficult because the average annual flow rate cannot be known ahead of time; however, these data do support the value of the annual nitrogen load calculated by the BEANS model. If all of the nitrogen that is input into the watershed (and is not lost during transport) in a given year is exported through the rivers in that same year, then 664,500 kg N/yr is a good estimate of that annual nitrogen load.

7. USGS SPARROW MODEL

The Wood River Valley Watershed nitrogen load predicted by the BEANS model differs significantly from that predicted by the USGS SPARROW model (Smith et al. 1997) for both the total mass of nitrogen delivered to the river and the relative nitrogen source contributions. The SPARROW model uses spatially-referenced land-surface and stream-channel characteristics to describe nitrogen sources, transport, and losses in watersheds. Nitrogen measurements from a national stream-monitoring network are used as a starting point to developing a regression equation that predicts nitrogen transport from watersheds based upon the watershed characteristics. Data from the entire national stream-monitoring network are used to develop the SPARROW model regression equation; however, the SPARROW model can be applied to specific basins within the national stream network. The following watershed characteristics are used in the SPARROW model: population density, point source facilities, farm-animal population, temperature, soil permeability, stream density, irrigated land, and precipitation. A statistical regression of these watershed attributes is used to correlate in-stream nitrogen measurements with the following nitrogen sources: point sources, fertilizer application, livestock waste, atmospheric deposition, and nonagricultural land (Smith et al. 1997). Table 7-1 lists the watershed characteristics and parametric coefficients used in the SPARROW model linear regression and the nitrogen sources to which nitrogen loads are attributed.

Table 7-1: SPARROW Model Parameters (adapted from Smith et al. 1997)

Model Parameters	***Parametric Coefficients***	***Coefficient Units***
Nitrogen Source		
Point Source	0.3464	dimensionless
Fertilizer Application	1.278	dimensionless
Livestock Waste Production	0.9723	dimensionless
Atmospheric Deposition	6.465	dimensionless
Nonagricultural Land	14.67	kg/ha/hr
Land to Water Delivery		
Temperature	0.0196	$°F^{-1}$
Slope		%
Soil Permeability	0.0442	h/cm
Stream Density	0.0215	km^{-1}
Wetland		dimensionless
Irrigated Land		dimensionless
Precipitation		cm
Irrigated Water Use		cm
Instream Decay		
Streamflow < 28.3 m^3/s	0.3758	d^{-1}
28.3 m^3/s < Streamflow < 283 m^3/s	0.1233	d^{-1}
Streamflow > 283 m^3/s	0.0406	d^{-1}
R^2	0.8743	

7.1 Nitrogen Mass Predictions

The BEANS model prediction of the total nitrogen delivered to the Big Wood and Little Wood Rivers is approximately one-fourth of the total nitrogen mass predicted in the SPARROW model. Table 7-2 lists the mass of nitrogen delivered the Wood River system as predicted by the two models, and Figure 7-1 graphically illustrates the predicted nitrogen masses.

Table 7-2: Predicted Mass of Nitrogen Delivered to the Wood River System

Model Nitrogen Source	*SPARROW Model Predicted Nitrogen Mass (kg N/yr)*	*BEANS Model Predicted Nitrogen Mass (kg N/yr)*	*Percent Difference*
TOTAL	2,500,000	665,000	275%
Point Source	10,300	23,400	-60%
Fertilizer	673,000	188,000	260%
Livestock Waste	700,000	274,000	155%
Atmospheric Deposition	132,000	100,000	30%
Nonagricultural Land	978,000	78,400	1,150%

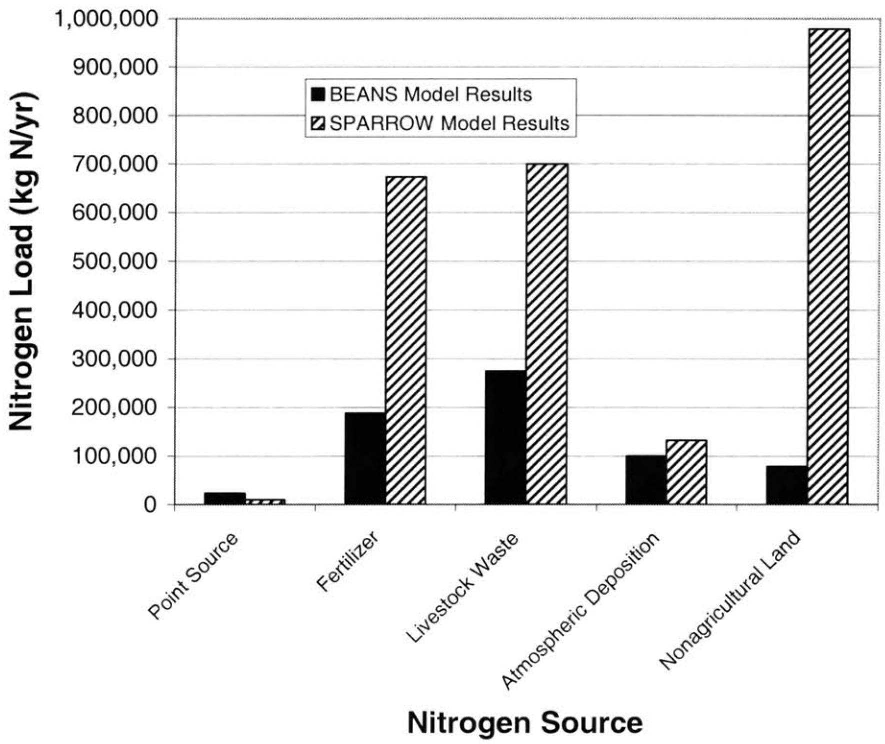


Figure 7-1: Predicted Mass of Nitrogen Entering the Wood River Valley Watershed

The point source category in the two models corresponds to wastewater treatment plant discharges to the Big Wood River. Nonagricultural land is defined in the SPARROW model as urban, forest, and rangeland. Sources of nitrogen in nonagricultural land include urban runoff, septic systems, and nitrogen fixation (Smith et al. 1997). For comparison with the SPARROW model, two nitrogen sources from the BEANS model—

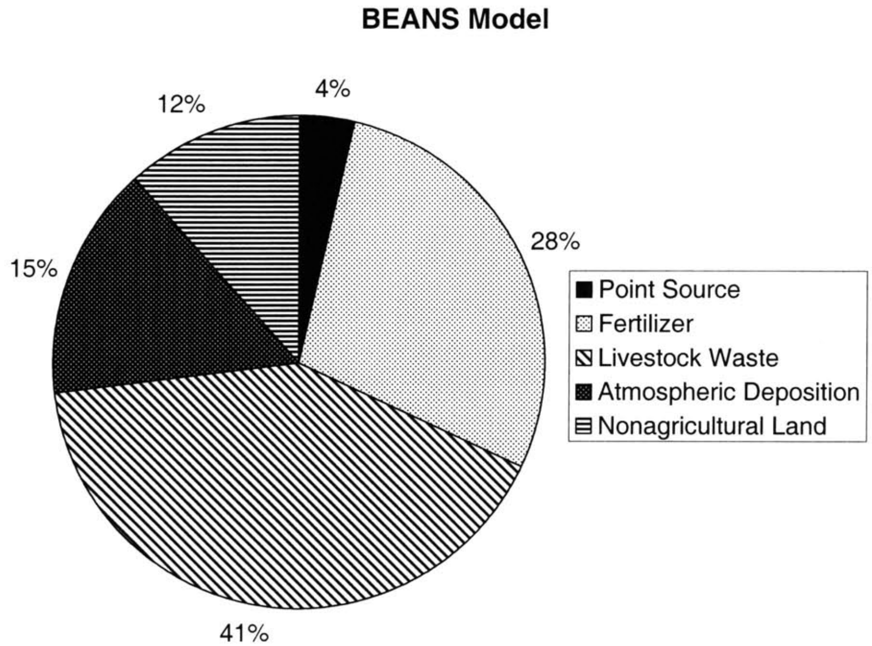
nitrogen fixation and septic systems—were combined to create the nonagricultural land category.

The SPARROW model and the BEANS model differ most significantly in predicted nitrogen mass from fertilizer, livestock waste, and nitrogen contributions from nonagricultural land. The SPARROW model prediction of nitrogen from fertilizer and livestock waste is approximately twice the mass predicted by the BEANS model. Using the BEANS model nitrogen excretion rate for cattle after volatilization and after losses during transport (5.6 kg N/animal/yr), the calculated number of cattle that must be present in the Wood River Valley Watershed to account for the SPARROW model predicted nitrogen mass from livestock waste is 125,000 cattle. This value is larger than the BEANS model estimate of 97,000 cattle in the watershed (Table 4-11).

The largest discrepancy between the two models is the predicted values for nitrogen from nonagricultural land. The SPARROW model prediction of the mass of nitrogen from nonagricultural land is almost ten times that of the BEANS model. The predicted masses of nitrogen from atmospheric deposition are very similar for the two models.

7.2 Relative Nitrogen Source Contributions

The BEANS model and SPARROW model attribute different proportions of the total mass of nitrogen entering the watershed to five sources: point sources, fertilizer, livestock waste, atmospheric deposition, and nonagricultural land. Figure 7-2 shows the sources of nitrogen entering the Wood River Valley Watershed and the percentage of total predicted nitrogen attributed to each source by the BEANS model and the SPARROW model.



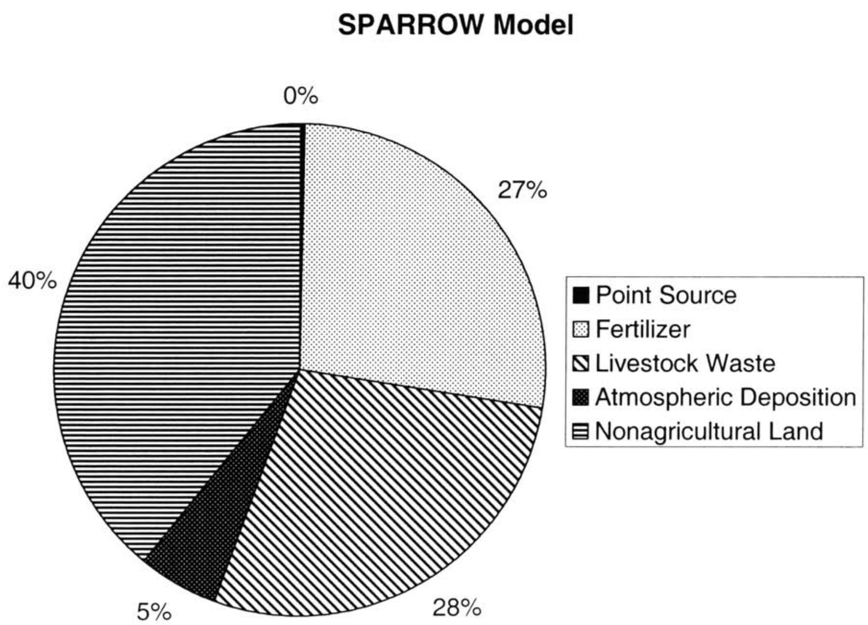


Figure 7-2: BEANS Model and SPARROW Model Predicted Sources of Nitrogen for the Wood River Valley Watershed

The largest contributor of nitrogen in the BEANS model is livestock waste, and the largest contributor in the SPARROW model is nonagricultural land. In both models, point sources are the smallest sources of nitrogen to the watershed. Though the two models predict very similar masses of nitrogen from atmospheric deposition, deposition accounts for 15% of the total nitrogen mass in the BEANS model and only 5% of the total nitrogen mass in the SPARROW model.

7.3 Discrepancies Between the BEANS Model and SPARROW Model

The four-fold difference between BEANS model and SPARROW model predictions of annual nitrogen delivered to the Wood River system is due to differences in the data used in the models and assumptions made in the SPARROW model.

7.3.1 Differences in Model Data

The SPARROW model regression equation was created using water quality and stream flow data from 400 monitoring stations in the National Stream Quality Accounting Network (Smith et al. 1997). The NASQAN network includes every drainage basin in the United States, including the Columbia/Snake River Basin. Though some monitoring stations in this network are located on the Snake River in Idaho, none are located in the Wood River Valley Watershed (NASQAN 2003). Figure 7-3 shows the locations of National Stream Quality Accounting Network (NASQAN) monitoring stations and the approximate location of the Wood River Valley Watershed.

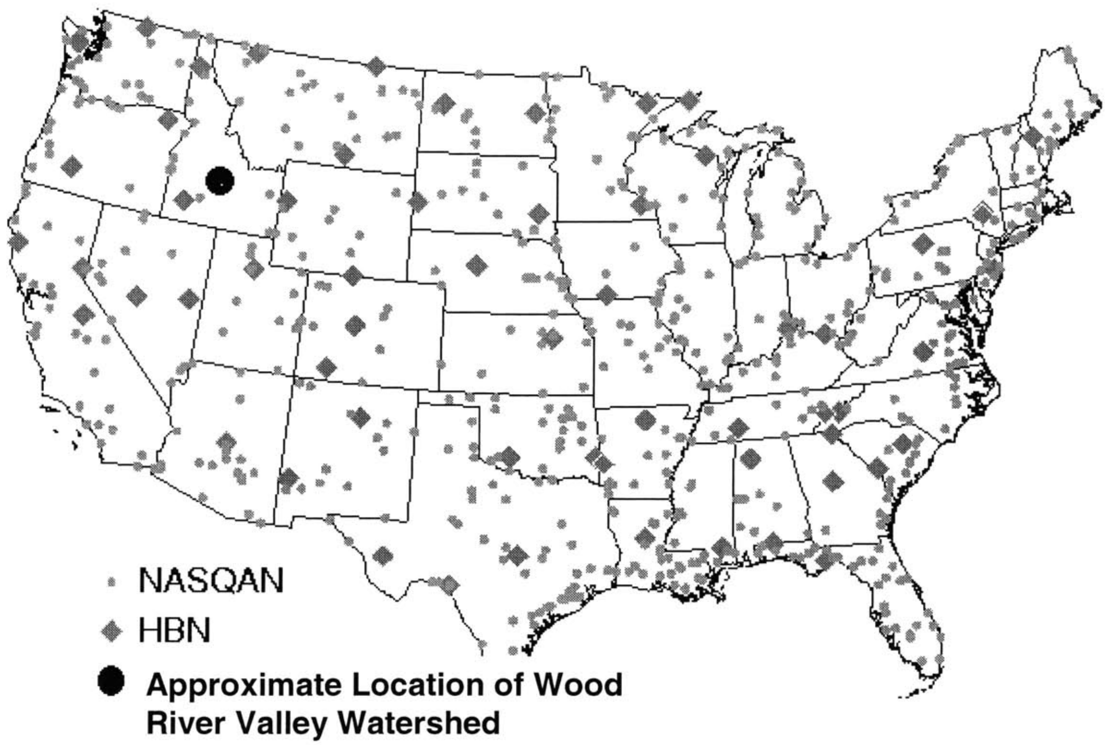


Figure 7-3: NASQAN Monitoring Network

Watershed characteristics data for both models were obtained from national, state, and county-level sources. The BEANS model is more specific to the Wood River Valley Watershed in its use of local data for fertilizer application rates and point source data. Fertilizer application rates for the BEANS model are obtained from local farmers and point source nitrogen data is collected directly from wastewater treatment plants in the watershed. The BEANS model calculations of nitrogen from livestock waste utilize grazing patterns for livestock that are determined from communication with local ranchers.

7.3.2 SPARROW Model Nonagricultural Land Source Assumption

The most significant difference between the BEANS model and SPARROW model nitrogen mass balance predictions is the mass of nitrogen introduced from nonagricultural land. The SPARROW model defines nitrogen contributions from nonagricultural land as nitrogen present in runoff from urban areas, forest, and rangeland. The magnitude of this source in the SPARROW model is assumed to be proportional to the total area of the defined nonagricultural land (Smith et al. 1997). Creators of the SPARROW model developed the nonagricultural land source as a fitting parameter to fit nitrogen predictions from the four mechanistically-determined nitrogen sources (point sources, fertilizer application, livestock waste, and atmospheric deposition) to measured stream nitrogen concentrations for all 400 monitoring stations in the NASQAN (Alexander et al. 2000).

Performance of the SPARROW model and the nonagricultural land source fitting parameter was validated for 16 watersheds in the northeastern United States (Alexander et al. 2002). The SPARROW model is expected to perform well for watersheds in the Northeast both because of the high density of NASQAN stations in that region (Figure 7-3) and because Northeastern watersheds have relatively small areas of nonagricultural land. The SPARROW model may perform differently in watersheds in the western United States because the proportion of nonagricultural land area is typically larger for western watersheds. Any errors in the nonagricultural land fitting parameter are magnified when the SPARROW model is applied to watersheds with greater areas of nonagricultural land.

The creators of the SPARROW model acknowledge that the model overpredicts nitrogen transport rates for relatively small, rural watersheds with large areas of nonagricultural land because no distinction is made between various types of nonagricultural nonpoint sources (Smith et al. 1997). The Wood River Valley Watershed meets the definition of the types of watersheds for which the SPARROW model overpredicts nitrogen masses. The watershed is rural, and over 80% of the watershed area is defined as nonagricultural land (Table 2-3). This fact may explain why the SPARROW model predicts almost ten times the mass of nitrogen from nonagricultural land than predicted by the BEANS model.

In contrast to the SPARROW model, the BEANS model does not attempt to quantify urban areas, forest, and rangeland as one land use and one nitrogen source, but instead the model treats residential (urban) areas, natural vegetation (forest), and rangeland as separate land uses. The BEANS model prediction of nitrogen mass entering the Wood

River Valley Watershed is more accurate than the SPARROW model prediction of nitrogen mass because the BEANS model calculates nitrogen loads from individual land areas by using nitrogen input parameters that are specific to this watershed.

8. ANALYSIS OF LAND USE CHANGES

8.1 AGRICULTURAL FERTILIZER REDUCTION

Agricultural sources (animal waste and fertilizer) constitute approximately 70% of the nitrogen load to the Wood River Valley Watershed. Cattle are the primary animal waste contributor, and five crops are the primary sources of agricultural fertilizer: alfalfa hay, barley, corn, potatoes, and wheat. Fertilization of each of these five primary crops contributes over 500,000 kg N/yr to the land surface, which is over 12% of the total agricultural fertilizer applied to the land surface. Table 8-1 lists the crops grown in the watershed and their relative nitrogen loads to the land surface.

Table 8-1: Nitrogen Loads to the Land Surface from Crop Fertilization

Crop	***Current Fertilization Rate (kg/ha)***	***Land in Production, 2000 (ha)***	***Total Nitrogen Input to Land Surface (kg N/yr)***	***Percent of Total Fertilizer Nitrogen***
Alfalfa Hay	**34**	**16,400**	**549,900**	**13 %**
Barley	**112**	**8,400**	**946,700**	**23 %**
Corn	**151**	**4,900**	**738,800**	**18 %**
Dry Beans	80	230	18,600	0 %
Oats	151	2,100	315,600	8 %
Potatoes	**191**	**3,300**	**632,100**	**16 %**
Sugarbeets	120	2,600	306,900	8 %
Wheat	**112**	**5,000**	**565,300**	**14 %**
TOTAL			4,100,000	100 %

Managing the amount, source, placement, form, and timing of nitrogen to the land surface can reduce nitrogen loads from agricultural sources. Recommendations for managing agricultural nitrogen loads are referred to as Best Management Practices (BMPs). The BEANS Model is used to evaluate the impacts of implementation of agricultural fertilizer BMPs on the nitrogen load to the Big Wood and Little Wood Rivers.

Examples of agricultural nitrogen BMPs for different agricultural land uses and activities include the following recommendations:

Rangeland (McFall and Wood 2003)

- Establishment of riparian zones, which have higher rates of denitrification and reduce nitrogen loads to surface water bodies
- Maintenance of adequate vegetation to stabilize soil and prevent erosion, which eliminates soil microorganisms that recycle nitrogen
- Adherence to determined rangeland grazing capacities to maintain rangeland health and reduce the amount of nitrogen introduced from animal waste

Fertilized crops (McFall and Wood 2003)

- Use of an irrigation tailwater recovery system to reuse irrigation water and leached nitrogen fertilizer
- Application of plant residues or mulching material to the soil surface to reduce fertilizer runoff
- Implementation of strip cropping—crop growth in a systematic arrangement of strips or bands to reduce water erosion and fertilizer leaching
- Utilization of agricultural wastes to provide fertility for crop production and decrease nitrogen fertilizer requirements

Fertilizer Application (Mahler et al. 1992)

- Annual soil sampling to determine soil fertility and fertilizer requirements
- Establishment of realistic crop yield goals and corresponding nitrogen fertilization rates
- Application of fertilizer in recommended amounts based upon scientific information
- Timing fertilizer applications to coincide with periods of maximum crop uptake
- Placing fertilizer appropriately (i.e. below seed, top dressed, banded application) for soil conditions and crop
- Use of slow release fertilizers to improve crop nitrogen use efficiency and reduce nitrogen fertilizer loads

Nitrogen loads from crops can also be reduced through the implementation of Precision Agriculture. Precision Agriculture has been developing since the 1980s and is a holistic management strategy that uses information technology to improve decisions relating to agricultural production, marketing, finance, and personnel (Robert 2002). Applying Precision Agricultural techniques to nitrogen fertilization involves using extensive data from soil samples, plant tissue samples, and remote sensing to vary fertilizer application across time and space within a field. Examples of Precision Agriculture include estimating the amount of nitrogen that crops return to the soil with color aerial photography of crop canopies (Sims et al. 2002), detecting the presence of weeds by the reflectance spectra of crop canopies (Vrindts et al. 2002), mapping crop yield variability within a field using airborne color-infrared imagery (Yang and Anderson 2000), and guiding self-propelled forage harvesters using a Real Time Kinematic Global Positioning System (Stoll and Kutzbach 2000). Crop yields can be improved and fertilizer use minimized through Precision Agriculture techniques (Robert 2002).

The effects of agricultural fertilization reduction are examined with the BEANS model by analyzing the use of fertilizer application rate BMPs and Precision Agriculture. Only fertilizer application rate BMPs are selected from all agricultural nitrogen BMPs for analysis because fertilizer application rates are well researched, the effects of different rates can be quantified, and adjusting application rates is a realistic practice that can be easily implemented. With relatively little effort, fertilizer application rate BMPs can maintain current crop yields while reducing nitrogen loads. Unlike fertilizer BMPs, rangeland BMPs applicable to the Wood River Valley Watershed are capital and labor intensive. For example, a USEPA project to restore 1200 ft of riparian zone on the Soque River in Georgia cost $55,000 (USEPA 2003), and a Columbia Basin Fish and Wildlife Authority project to rehabilitate 12 miles of riparian habitat along the Tucannon River is estimated to cost $70,000 (Stendal 2002). We do not know the area of riparian habitat in need of rehabilitation in the Wood River Valley Watershed, and the reductions in nitrogen load resulting from the implementation of this rangeland nitrogen BMP are not easily quantified. Similarly, the extent of adherence to determined grazing capacities and resulting status of natural vegetation in rangeland areas is not known; therefore, the applicability and effectiveness of grazing density and natural vegetation BMPs cannot be estimated.

The nitrogen impacts of Precision Agriculture techniques are analyzed because Precision Agriculture represents a relatively new area of research, preliminary results indicate substantial nitrogen fertilizer reductions are possible, and the fertilizer reductions achieved are quantifiable.

Fertilization rate BMPs are developed for specific crops based upon relationships obtained from soil nutrient tests, crop yield response, and previously planted crops (Mahler 1999). For the five major crop sources, Table 8-2 lists recommended nitrogen fertilizer application rates designed to maximize crop growth and economic return while protecting water quality (Waskom 1994). These rates range from 3% to 18% less than fertilizer application rates currently employed in the watershed.

Due to lack of data and uncertainty about the parameters upon which the fertilizer application BMPs are designed, we assume a fertilizer application reduction of 5 lb N/acre for corn and potatoes. This is consistent with fertilizer reduction BMPs for the other major nitrogen source crops in the watershed.

The impacts to the total watershed nitrogen load of implementing fertilizer BMPs was evaluated using the BEANS Model, and the results (after losses) are shown in Table 8-2. A total reduction of 17,370 kg N/yr can be achieved if BMP fertilizer rates are implemented for all five of the major nitrogen source crops.

Table 8-2: Watershed Nitrogen Load Reduction from Implementation of BMP Nitrogen Fertilization Rates

Crop	*Current Fertilization Rate (kg N/ha)*	*BMP Fertilization Rate (kg N/ha)*	*BMP Source*	*Percent Rate Reduction*	*Reduction in Watershed Nitrogen Load (kg N/yr)*	*Percent Load Reduction*
Alfalfa Hay	34	28	Mahler (1999)	17.6%	7,550	1.1 %
Barley	112	106	Jackson (2000)	5.4%	3,810	0.6 %
Corn	151	146	--	3.3%	2,190	0.3 %
Potatoes	191	185	--	3.1%	1,560	0.2 %
Wheat	112	106	Agrium (2001), Vigil (2003)	5.4%	2,260	0.3 %
TOTAL					**17,370**	**2.5 %**

--: fertilizer rates assumed based on values for other crops

Precision Agriculture is a relatively new area of agricultural research, and preliminary results indicate that implementing Precision Agricultural techniques can maintain agricultural yields while significantly reducing nitrogen fertilizer inputs (Van Alphen and Stoorvogel 2000, Stelljes 2000). Research in the Netherlands reduced fertilizer inputs to winter wheat by 23% using precision fertilization compared to traditional fertilization techniques and improved crop yield by 3% (Van Alphen and Stoorvogel 2000). Preliminary results from Precision Agriculture research conducted by the US Department of Agriculture – Agricultural Research Service in Colorado indicate that nitrogen fertilization rates for corn can be reduced by 35 lb N/acre without reducing crop yield (Stelljes 2000). In the Wood River Valley Watershed, a 35 lb N/acre reduction in nitrogen fertilization rates reduce corn fertilization to 100 lb N/acre. This precision fertilization rate is consistent with research from Iowa State University showing that corn yields of an average of 167 bushels/acre decreased only 3 bushels/acre from 23 test fields across Iowa when fertilizer was applied at an average rate of 101 lb N/acre—a reduction of 50 lb N/acre from traditional fertilization practices in the state (Blackmer and Van De Woestyne 2002). This site-specific research has not yet been accepted as a BMP for corn fertilization, but it indicates that substantial reductions in nitrogen fertilization of corn may be possible.

Research quantifying the fertilization reductions achievable through the use of Precision Agriculture is very site-specific and is currently only available for wheat and corn; however, similar reductions may be possible for other crops in a variety of locations. Based upon these research results, the impacts of reducing fertilizer application in the Wood River Valley Watershed by 20% and 25% through Precision Agriculture techniques are evaluated using the BEANS Model. The impacts of fertilizer reductions for the five primary nitrogen source crops and for all watershed crops are shown in Table 8-3. Reductions in watershed nitrogen load vary depending on the extent of Precision Agriculture use among crops and the fertilizer reduction achieved. A total reduction of 83,750 kg N/yr can be achieved if Precision Agriculture techniques provide a 25% reduction in fertilizer use for all crops in the watershed.

Table 8-3: Watershed Nitrogen Load Reduction from the Implementation of Precision Agriculture Techniques

Crops	***Percent Fertilizer Use Reduction***	***Reduction in Watershed Nitrogen Load (kg N/yr)***	***Percent Reduction***
Primary fertilizer crops	20 %	56,550	8.5 %
Primary fertilizer crops	25 %	70,690	10.6 %
All crops	20 %	67,000	10.1 %
All crops	25 %	83,750	12.6 %

8.2. WATERSHED BUILD-OUT ANALYSIS

Over the last two decades (1980-2000), the population of the Wood River Valley has been growing at a rate of approximately 10% per year (800 people/yr) (U.S. Census 2000). The watershed population is expected to continue to grow in future years. We use the BEANS model to analyze how this population growth will affect the nitrogen load to the Big Wood and Little Wood Rivers. The magnitude of this effect will be determined by the nature of the residential development that is built to accommodate the increasing population. Because agricultural land is currently much more common in the watershed than is residential land (Figure 8-1), it is likely that land for new residential development will come from land that is now used as agricultural land (cropland and pasture).

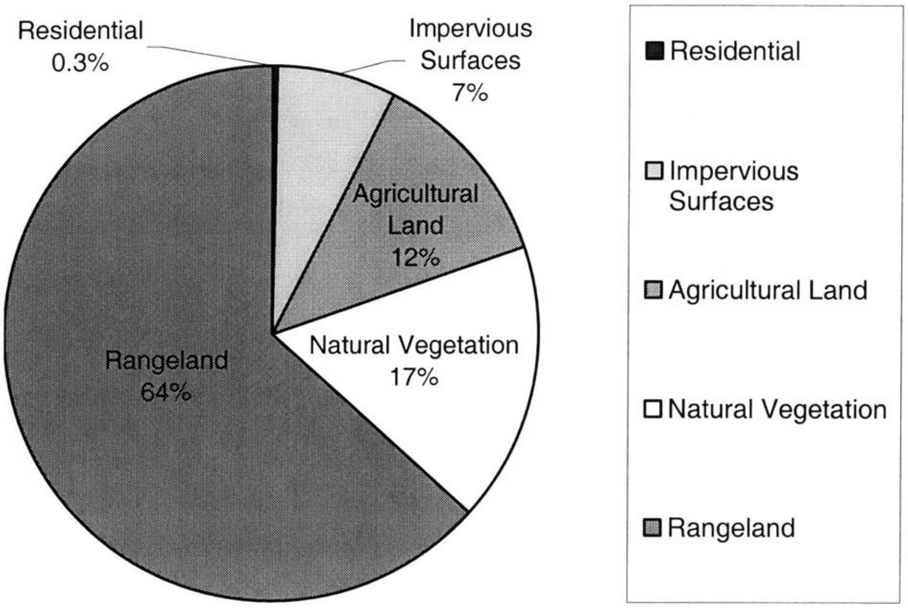


Figure 8-1: Distribution of Land Uses within the Wood River Valley Watershed

Natural vegetation areas and rangeland are much more abundant than both residential land and agricultural lands, but these areas are largely publicly owned by the U.S. Forest Service, the Bureau of Land Management (BLM), and the State of Idaho (Figure 8-2).

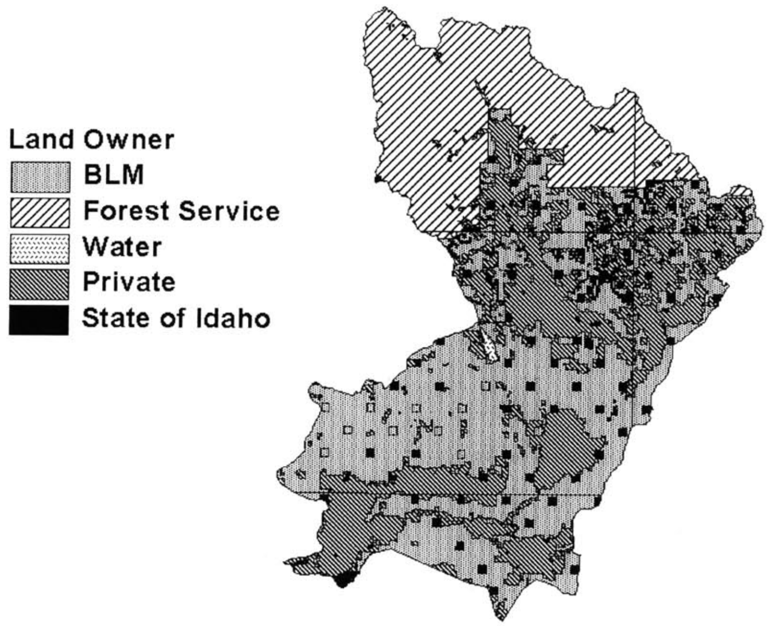


Figure 8-2: Land Ownership in the Wood River Valley Watershed

Agricultural lands are the most probable locations for new residential development because these lands are privately held. Whether converting agricultural land to residential land will increase or decrease the watershed nitrogen load will be a function of the relative magnitudes of nitrogen yields from these different land uses.

Nitrogen yields for agricultural lands are determined by the magnitude of atmospheric deposition, fertilizer application rates, abundance of nitrogen fixing crops, type of livestock, and the density of those livestock. Nitrogen yields for residential lands are a function of atmospheric deposition, lawn fertilizer application rates, the level of wastewater treatment, and residential lot sizes, which control population densities. Figure 8-3 compares current agricultural nitrogen yields to calculated residential nitrogen yields for different combinations of wastewater treatment and lot size. The lot sizes considered in this analyses are those that are currently included in the Blaine County Zoning Code, and the different wastewater treatment possibilities considered are tertiary treatment discharging to a river channel similar to the level of treatment at the Hailey wastewater treatment plant, on-site septic systems, secondary treatment discharging to infiltration lagoons similar to the level of treatment at the Bellevue treatment plant, and secondary treatment discharging to a river channel similar to the level of treatment at the Ketchum/Sun Valley treatment plant.

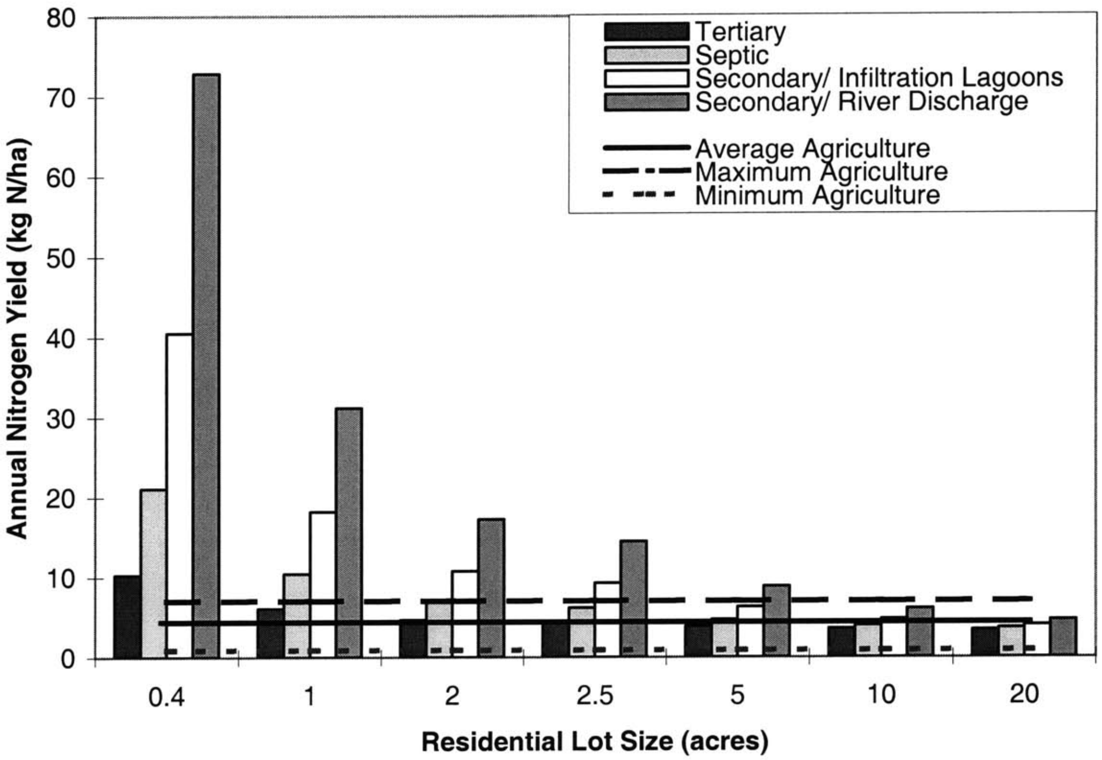


Figure 8-3: Relative Nitrogen Yields from Residential and Agricultural Land Uses

All the possible combinations of lot size and level of wastewater treatment produce residential nitrogen yields that are larger than the minimum agricultural nitrogen yield, so conversion of agricultural lands with the minimum nitrogen yield to any type of residential use will increase the watershed nitrogen load. With respect to the average and maximum agricultural nitrogen yields, some residential combinations have smaller yields and some have larger yields, so the loading changes as a result of converting these areas will be determined by the type of residential development that occurs.

To determine what type of residential development would allow the watershed nitrogen load to remain constant after agricultural land conversions, we calculate the residential lot sizes that would create nitrogen yields equal to agricultural nitrogen yields for each level of wastewater treatment. Figure 8-4 shows the relationship between residential nitrogen yield and lot size for each wastewater treatment option.

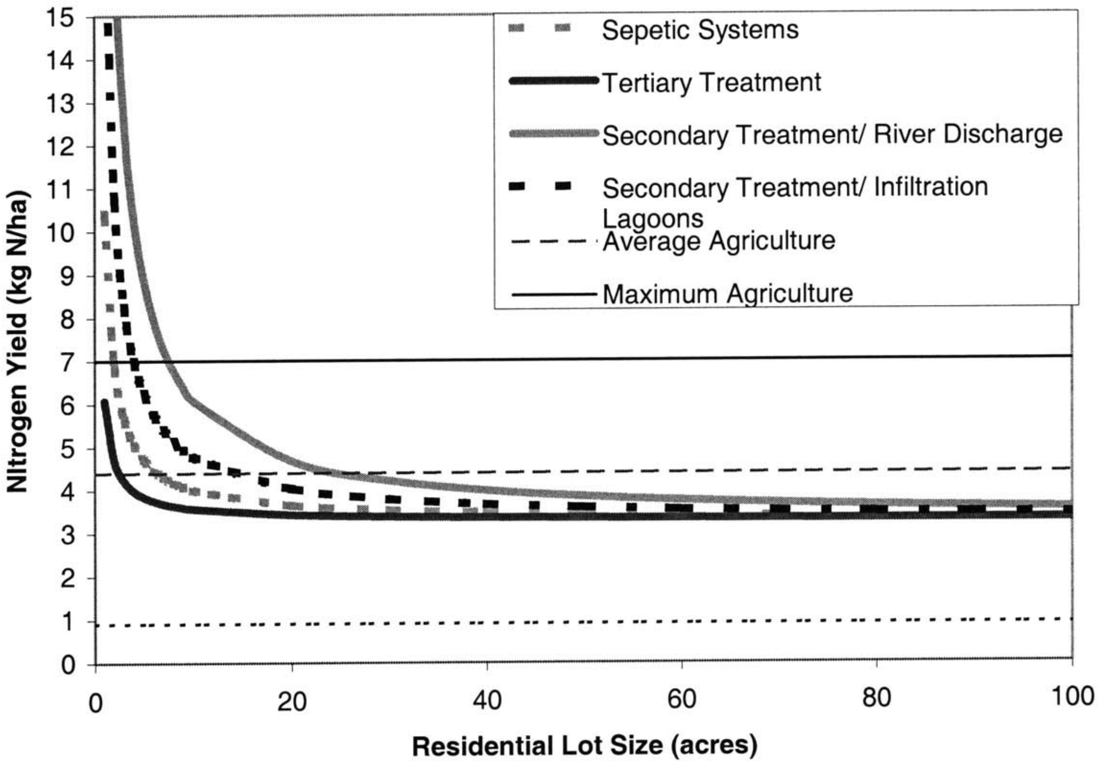


Figure 8-4: Residential Nitrogen Yields as a Function of Lot Size

Residential nitrogen yields decrease as lot sizes increase, but yields all converge to a minimum residential yield of 3.3 kg N/ha at large lot sizes, when the influence of wastewater nitrogen inputs becomes negligible. Currently, residential yields cannot be reduced below this value because this is the minimum yield that is created from atmospheric deposition and fertilizer inputs which are introduced at rates that are proportional to land area. The specific lot sizes that generate nitrogen yields equivalent to current average and maximum agricultural nitrogen yields are summarized in Table 8-4.

Table 8-4: Residential Lot Sizes such that Residential Nitrogen Yields are Equal to Current Agricultural Yields

	Average Agricultural Yield	*Maximum Agricultural Yield*
Tertiary Treatment	2.5 acres	0.75 acres
Septic Systems	6.4 acres	1.9 acres
Secondary Treatment/ Infiltration Lagoons	13 acres	4.0 acres
Secondary Treatment/ River Discharge	25 acres	7.5 acres

With the exception of the lot size required to make a residential use with secondary treated wastewater discharging to a river equivalent to the average agricultural yield, the lot sizes in Table 8-4 are within the range of lot sizes included in the Blaine County Zoning Code; however, no lot size would be big enough to generate a residential yield equal to the minimum current agricultural yield regardless of the level of wastewater treatment.

If lawn fertilizer applications were reduced, the minimum residential yield would decrease in value, but would still be related to land area. Currently, the BEANS model assumes that 25% of a residential lot is fertilized. If this fertilizer percentage is reduced to 0%, significant reductions in residential nitrogen yields will occur. Figure 8-5 shows the relationship between residential nitrogen yield and lot size for each wastewater treatment option assuming that no residential fertilizers are used.

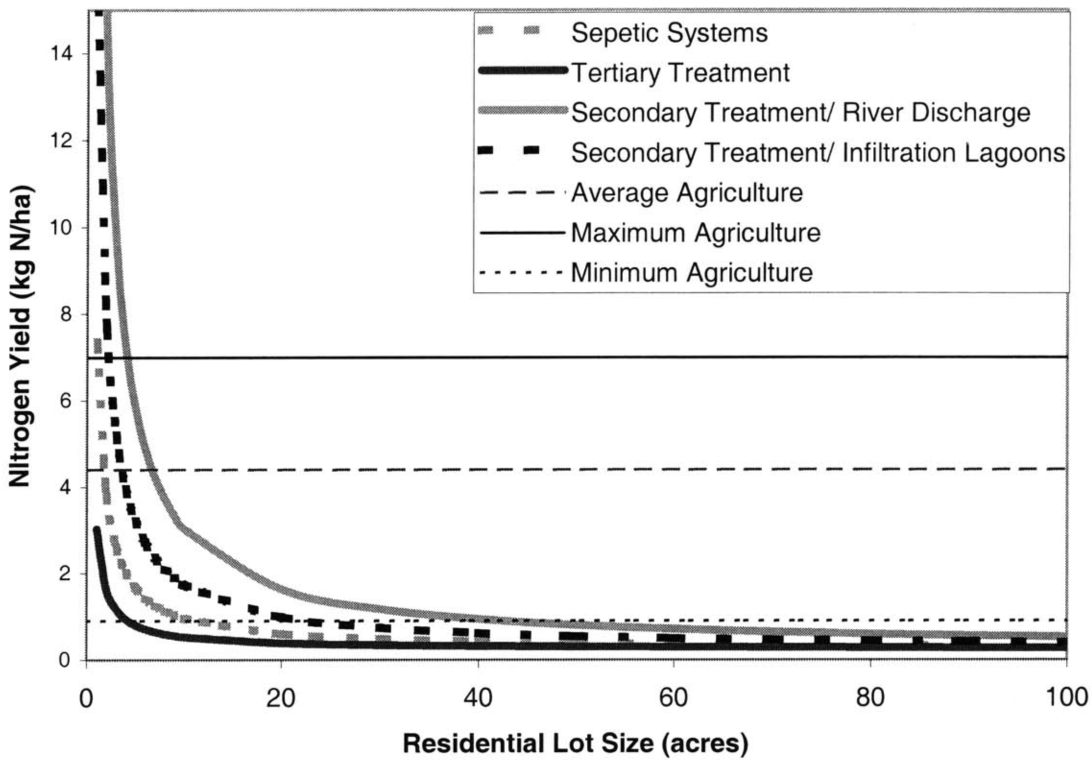


Figure 8-5: Residential Nitrogen Yields as a Function of Lot Size without Residential Fertilizer Applications

Without lawn fertilizer applications, residential nitrogen yields converge to a value of 0.25 kg N/ha at large lot sizes when wastewater inputs become negligible. This yield is significantly smaller than all the current agricultural nitrogen yields. The specific lot sizes that generate nitrogen yields equivalent to current average and maximum agricultural nitrogen yields are summarized in Table 8-5.

Table 8-5: Residential Lot Sizes without Lawn Fertilizer Applications such that Residential Nitrogen Yields are Equal to Current Agricultural Yields

	Minimum Agricultural Yield	*Average Agricultural Yield*	*Maximum Agricultural Yield*
Tertiary Treatment	4.3 acres	0.7 acres	0.4 acres
Septic Systems	11 acres	1.7 acres	1.1 acres
Secondary Treatment/ Infiltration Lagoons	23 acres	3.6 acres	2.2 acres
Secondary Treatment/ River Discharge	43 acres	6.7 acres	4.1 acres

Removing lawn fertilizers as a nitrogen input to residential sources substantially reduces the magnitude of nitrogen yields from these areas. In the following build-out analyses, we assume that residential fertilizer application rates remain constant; however, it should be recognized that this source presents an additional opportunity for reducing nitrogen yields from residential sources.

8.2.1. Watershed: 2025 Population

Based on local population growth rates during the 1990s, we estimate that the watershed population will grow to approximately 40,000 people by the year 2025. This growth would introduce 16,000 additional people to the watershed, requiring that new land areas be developed to house the expanding population. Policy decisions made by local officials and citizens with respect to population densities and wastewater treatment methods will control how a growing population will impact the nitrogen water quality of the Big Wood and Little Wood Rivers. We use the BEANS model to calculate the 2025 watershed nitrogen load assuming that new residential areas are developed from current agricultural areas.

There are currently 84,000 ha of cropland and pasture in the watershed. The lot sizes used in new residential developments will determine how much agricultural land is ultimately converted. We assume that each residence will house approximately 2.6 people (Blaine County 1999). Table 8-6 summarizes how much agricultural land will be needed to accommodate the new residential development under each of Blaine County's zones that allow for residential land uses.

Table 8-6: Agricultural Land Requirements to Accommodate 2025 Residential Build-Out

Blaine County Zone	*Residential Lot Size (acres)*	*Land Area Converted (ha)*	*Percent of Current Agricultural Land*
R-0.4	0.4	980	1%
R-1	1	2,400	3%
R-2	2	4,900	6%
R-2.5	2.5	6,100	7%
R-5	5	12,200	15%
A-10	10	24,400	29%
A-20	20	48,900	58%

The nature of wastewater treatment used in new developments will determine the magnitude of the nitrogen load originating from each house. Five wastewater treatment possibilities are considered in the watershed build-out analysis:

1. Current Treatment Distribution: This option assumes that 45% of the population will continue to use on-site septic systems, 26% will be connected to a tertiary wastewater treatment plant that discharges to the river (similar to the current Hailey treatment plant), 22% will be connected to a secondary treatment plant that discharges to the river (similar to the Ketchum/Sun Valley treatment plant), and 7% will be connected to a secondary treatment plant that discharges to infiltration lagoons (similar to the Bellevue treatment plant).
2. Tertiary Treatment: This option assumes that all new development will be connected to a wastewater treatment plant, that all current and any new treatment plants will provide tertiary wastewater treatment, and that those households that currently use on-site septic systems will continue to rely on those systems for their wastewater treatment.
3. Septic Systems: This option assumes that all new residential development will rely on on-site septic systems for their wastewater treatment. Those households currently connected to the Hailey, Bellevue, and Ketchum/Sun Valley wastewater treatment plants will stay connected to these plants, but no new households will be connected to any wastewater treatment plants.
4. Secondary Treatment/ Infiltration Lagoons: This option assumes that all new residential development will be connected to secondary wastewater treatment plants that discharge to infiltration lagoons. Households currently using septic systems will continue to use septic systems.
5. Secondary Treatment/ River Discharge: This option assumes that all new residential development will be connected to secondary wastewater treatment plants that discharge directly to the Big or Little Wood Rivers. Households currently using septic systems will continue to use septic systems.

Figure 8-6 shows the 2025-watershed nitrogen load calculated using the BEANS model considering each combination of residential lot size and wastewater treatment options.

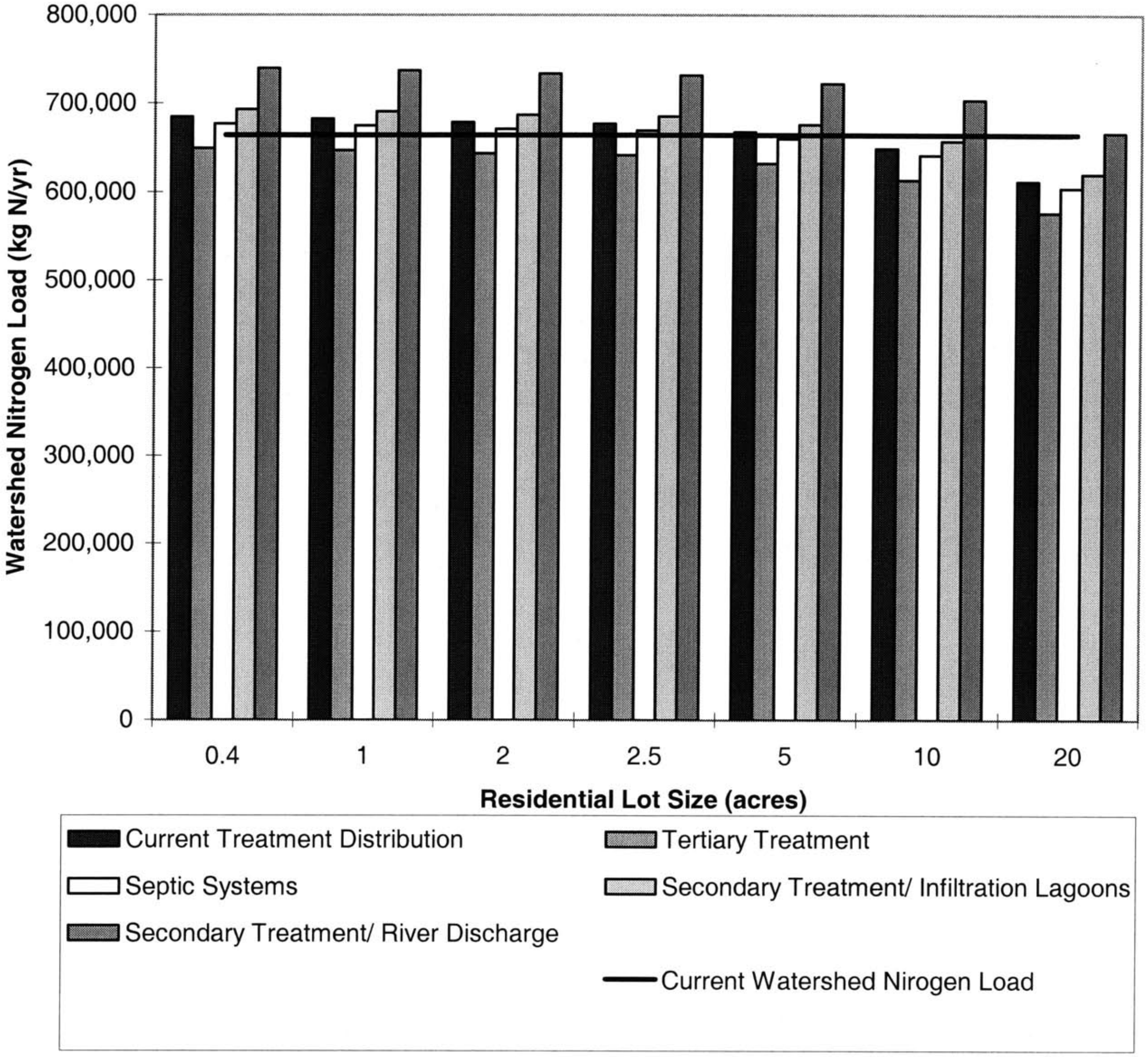


Figure 8-6: Possible 2025 Watershed Nitrogen Loads

Future development will probably include a combination of different lot sizes, but these calculated nitrogen loads provide an estimate of how the average future lot size is related to surface water quality.

Regardless of future residential lot size, expanding the availability of tertiary wastewater treatment will decrease the watershed nitrogen load by 2025. The septic system option may also be an effective possibility for reducing the future nitrogen load to the rivers, but the negative impacts on local groundwater quality caused by septic systems may outweigh the gains in surface water quality. Expanding the availability of secondary wastewater treatment facilities that discharge directly to the rivers will not decrease the watershed nitrogen load by 2025 under any of the population density options considered in the Blaine County Zoning Code. For the majority of the lot sizes considered, expanding the availability of secondary wastewater treatment of any kind will cause an increase in the watershed nitrogen load by 2025.

8.2.2. Upper Valley: 2025 Population

Because most of the watershed's population is contained in the Upper Valley, most of the future population growth is expected to occur in this sub-watershed as well. We conduct a similar build-out analysis to assess how population growth will affect the Upper Valley nitrogen load. Similar assumptions to those used in the entire watershed build-out analysis are applied again to the Upper Valley. The 2025 population is estimated to be approximately 29,000 people (Parsons Brinckerhoff Quade & Douglas 2002). This population increase adds 10,000 people to the Upper Valley over approximately 25 years and accounts for 63% of the population growth estimated to occur in the entire watershed during this period.

There are currently 10,800 ha of cropland and pasture in the Upper Valley. Again, we assume that there will be an average of 2.6 people residing in each new house (Blaine County 1999). Table 8-7 summarizes how much of the Upper Valley agricultural land will need to be converted to residential land to accommodate the 2025 population considering the same Blaine County zones.

Table 8-7: Agricultural Land Requirements to Accommodate 2025 Residential Build-Out

Blaine County Zone	***Residential Lot Size (acres)***	***Land Area Converted (ha)***	***Percent of Current Agricultural Land***
R-0.4	0.4	640	5%
R-1	1	1,600	15%
R-2	2	3,200	30%
R-2.5	2.5	4,000	37%
R-5	5	8,000	74%
A-10	10	16,000	148%
A-20	20	32,000	297%

The 10-acre and 20-acre lot size developments would require more agricultural land than is available in the Upper Valley, so these lot sizes are not considered in the sub-watershed build-out analysis. Average lot sizes of approximately 6.7 acres would convert 100% of the current agricultural land to residential land, so this lot size is considered in addition to the possible zoned lot sizes.

The assumptions about wastewater treatment are essentially the same as they were in the entire watershed build-out analysis except the current distribution of wastewater treatment levels is slightly different. The Upper Valley current distribution option assumes that 33% of the population will be connected to a tertiary wastewater treatment plant that discharges to the river (similar to the current Hailey treatment plant), 31% will continue to use on-site septic systems, 27% will be connected to a secondary treatment plant that discharges to the river (similar to the Ketchum/Sun Valley treatment plant), and 9% will be connected to a secondary treatment plant that discharges to infiltration lagoons (similar to the Bellevue treatment plant).

Figure 8-7 shows the 2025 Upper Valley nitrogen load calculated using the BEANS model considering each combination of residential lot size and wastewater treatment options.

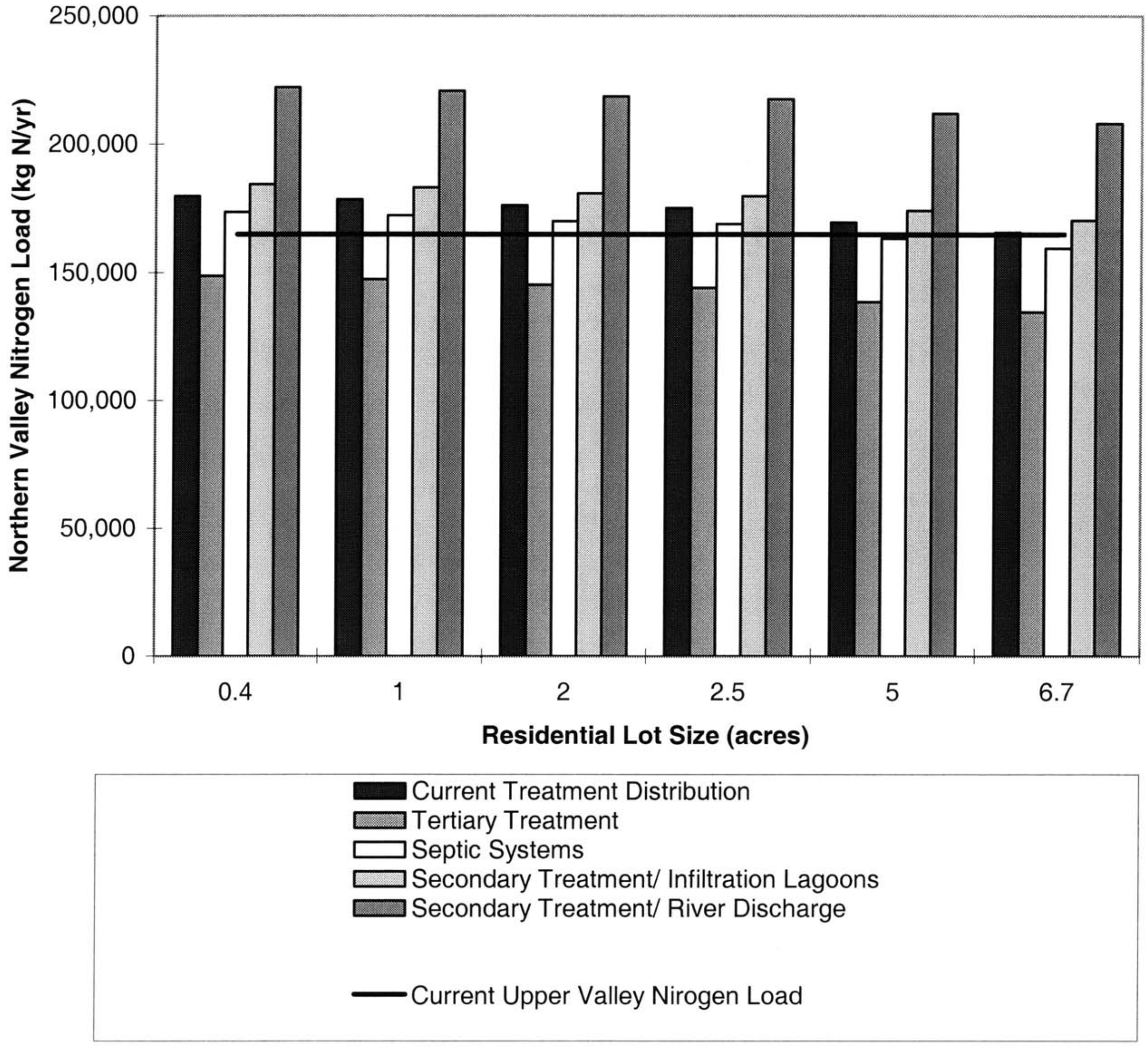


Figure 8-7: Possible 2025 Upper Valley Nitrogen Loads

Tertiary treatment is clearly the only wastewater treatment option that consistently reduces the Upper Valley nitrogen load under each population density considered. The septic system option enables the 2025 nitrogen load to drop below the current load at the largest possible lot size, but again the costs of increased contamination of local groundwater around septic systems may outweigh the benefits from a slight reduction in the surface water nitrogen load. Similar to the dynamics observed for the watershed as a whole, connecting more households to secondary wastewater treatment plants will only increase the current Upper Valley nitrogen load.

8.3 Septic System Considerations

Septic systems are a relatively small source of nitrogen to the Wood River Valley Watershed. BEANS Model results attribute to septic systems a watershed nitrogen load of 9,800 kg N/yr, which accounts for 1% of the total nitrogen load. Though septic systems may have a relatively small impact on nitrogen water quality in the Wood River system compared to other nitrogen sources in the watershed, septic systems can pose a threat to both human and ecosystem health as a result of other contaminants that they introduce to ground and surface water. Septic systems are a potential public health concern because they can cause nitrate groundwater contamination in excess of the USEPA drinking water standard and bacterial and viral contamination of groundwater (USEPA 2001). Septic systems can also have negative impacts on ecosystem health because of phosphorus loading to surface waters.

8.3.1 Nitrogen Contamination of Groundwater

Septic systems in the Wood River Valley are potential sources of nitrate groundwater contamination in excess of the USEPA drinking water standard of 10 mg/L NO_3^- - N. Concentrated plumes of septic discharge may persist in watershed aquifer systems with very little dispersion of nitrate. Research conducted on two septic systems located in sand aquifers concluded that the dispersive capabilities and contaminant dilution potential of many sand and gravel aquifers are much less than previously thought (Robertson et al. 1991). The studied aquifers were primarily composed of fine sand—similar to the Wood River Valley alluvial aquifers—and had very low capacities for vertical and horizontal dispersion. The study authors found the results to be consistent with other aquifer dispersivity studies in Borden and Twin Lakes, Ontario, and Cape Cod, Massachusetts (Robertson et al. 1991).

Nitrogen concentrations in septic system plumes can be estimated from the size of septic system leach fields and the average per capita nitrogen excretion rate. Human nitrogen inputs into the septic system are diluted by septic system flow and precipitation infiltration into the leach field. Required leach field size is determined by the number of bedrooms in the home and the type of soil in which the leach field is located (IDHW 2000).

For estimating the nitrogen concentration in septic plumes in the Wood River Valley Watershed, we use an average per capita nitrogen excretion rate of 13,150 mg N/person/day (Valiela et al. 1997). A twenty-year average precipitation rate (NADP 2002) is used to calculate infiltration into septic system leach fields. The calculated infiltration rate is 0.00008 m/day, 8.8% of the average precipitation over the leach field area (Lopez-Bernal 2003). The primary flow diluting the human nitrogen inputs is the septic flow from the home. A daily domestic water usage of 0.51 m^3/person/day (USGS 2003) is estimated specifically for the Wood River Valley, and a more general usage of 0.18 m^3/person/day is estimated by USEPA (1995). Septic system flowrates are assumed to be equal to domestic water usage. Because septic system flow is significantly (two orders of magnitude) greater than infiltration into the septic system leach field, calculated nitrogen concentrations in septic plumes become a function only of the number of residents per household. Nitrogen losses of 35% from denitrification occurring within the leach field are applied to the calculated nitrogen concentrations (Valiela et al. 1997). Table 8-8 lists septic plume concentrations after denitrification losses for two estimates of septic system flowrates. Nitrate concentrations far exceed the USEPA drinking water standard of 10 mg N/L (USEPA 2002) for both septic flowrate estimates.

Table 8-8: Calculated Nitrogen Concentrations in Septic System Groundwater Plumes

Septic System Flowrate (m^3/person/day)	***Plume Nitrate Concentration (mg N/L)***
0.51	17
0.18	48

No measurements of nitrogen concentrations in septic system leach field plumes are available from homes in the Wood River Valley, so we are unable to verify these calculated nitrogen plume concentrations; however, groundwater nitrogen concentrations are available from monitoring wells at the Bellevue wastewater treatment plant. At the plant, wastewater is treated in aerated ponds and then pumped to a final percolation pond, where it infiltrates into the groundwater. Groundwater samples taken from monitoring wells near the percolation pond have elevated nitrogen concentrations of 7 mg N/L (Turner 2003). This value is the only direct measurement of groundwater nitrogen concentrations originating from the discharge of domestic wastewater to the subsurface. While the Bellevue wastewater treatment plant differs from septic systems, monitoring data from this plant show that the discharge of wastewater to the subsurface generates elevated groundwater nitrogen concentrations.

Research conducted on septic systems in unconfined sand and gravel aquifers in Ontario, Canada and Western Australia verifies the existence of persistent, concentrated nitrogen plumes originating from septic system leach fields. Nitrate concentrations above 100 mg N/L were measured at the water table beneath a septic leach field in an unconfined sand aquifer in Ontario, Canada. Concentrations of 28 mg N/L persisted 50 m downgradient in the center of the leach field plume (Harman et al. 1996). Similar results were obtained from the study of two septic systems in unconfined sand aquifers in Ontario, Canada. At both sites studied, nitrate concentrations were measured above 25 mg N/L in plumes 25 m downgradient of the septic system leach fields (Wilhelm et al. 1996). At a study site in Western Australia, nitrate concentrations of 70 mg N/L were measured in a septic system leach field located in a sand and gravel aquifer (Gerritse et al. 1995).

8.3.2 Bacterial and Viral Contamination of Groundwater

Septic systems are a significant source of groundwater contamination and can lead to waterborne disease outbreaks and other adverse health effects. The bacteria, protozoa, and viruses in wastewater can cause numerous diseases, including gastrointestinal illness, cholera, and typhoid (USEPA 2001). Research conducted on four septic systems located in sandy soils in coastal North Carolina concluded that malfunctioning septic systems are a risk to public health by contaminating ground and nearby surface waters with pathogens (Scandura and Sobsey 1997). The virus studied (bovine enterovirus type 1) was capable of surviving septic system treatment and migrating into groundwater. Viral contamination of groundwater was related to distance from the septic system leach field and pH of the groundwater (Scandura and Sobsey 1997), with viruses occurring more frequently in groundwater sampled near the septic system leach fields and in groundwater with higher pH.

Studies of viral transport from two septic systems in sand aquifers in western Montana and south-central Wisconsin also concluded that septic systems are a source of infectious viruses entering the groundwater. Coliphage viruses (viruses that infect coliform bacteria in the human intestinal tract) were studied as an indicator of pathogenic viruses in a septic system leach field plume in western Montana (DeBorde et al. 1998). Coliphage viruses were consistently detected in groundwater samples along a 17 m flowpath downgradient of the leach field. Coliphage concentrations were highest where septic waste concentrations were determined to be at a maximum (DeBorde et al. 1998). Transport of polioviruses from septic tank effluent was studied in south-central Wisconsin (Alhajjar et al. 1988). An average of 88% of polioviruses escaping from the septic tank to the leach field was transported to the groundwater. Polioviruses were determined to move freely in groundwater with little or no retardation (Alhajjar et al. 1988).

Fecal coliform bacteria, indicators of the presence of pathogens, were detected in the groundwater samples near four septic system leach fields in coastal North Carolina (Scandura and Sobsey 1997), while another study concluded that fecal coliform bacteria were removed completely by soil underneath the septic system leach field and were not transported to groundwater (Alhajjar et al. 1988). Other investigations have found limited mobility and survival of fecal coliforms in soils and concluded that coliform bacteria were unlikely to be transported into the groundwater. Extensive movement of bacteria in the subsurface is possible depending on the soil and geological features of the area (Canter and Knox 1985).

The results of septic system studies are applicable to the predominantly coarse soils of the Wood River Valley alluvial unconfined aquifer, and they indicate that septic system leach fields are a potential source of bacterial and especially viral contamination of groundwater. Future planning decisions for the watershed must consider the potential human health effects of septic tanks in addition to their effect on watershed nitrogen loads.

8.3.3 Phosphorus Loading to Surface Water

In addition to nitrogen, bacteria, and viruses, septic systems can contribute phosphorus to the groundwater and surface waters of the Wood River Valley Watershed. Normal septic tank effluent phosphorus concentrations (5 –20 mg P/L) are several orders of magnitude greater than environmental background levels (Robertson and Harman 1999). Phosphorus in water does not pose a direct threat to human health (Carpenter et al. 1998), and no drinking water standard has been established by USEPA; however, phosphorus, like nitrogen, is a nutrient that can cause eutrophication of surface water.

Eutrophication accounts for half of the impaired lake area and 60% of the impaired river reaches in the U.S., and excessive phosphorus inputs are a major cause of freshwater eutrophication (Carpenter et al. 1998). Pursuant to Section 303(d) of the CWA, a total phosphorus TMDL has been established for the Big Wood River (IDEQ 2002) because the river does not meet water quality standards. In addition to depleting oxygen and causing eutrophication, freshwater algal blooms caused by phosphorus loading can be toxic to aquatic life (Carpenter et al. 1998).

The mechanism of phosphorus transport from septic systems to surface water is well described by research conducted on ten septic system plumes in Ontario, Canada (Robertson et al. 1998). Average phosphate concentrations ranging from less than 1 mg P/L to 5 mg P/L were measured in plumes from septic system leach fields. Phosphorus plume lengths varied from less than 1 m to greater than 25 m. Migration of all of the phosphate plumes studied was significantly retarded (retardation = 20 to 100) compared to groundwater velocity because of the strong affinity of phosphate for sorption onto the subsurface material. Although substantial phosphate retardation was observed, phosphate migration velocities at sites with sandy aquifers were sufficiently fast (approximately 1 m/yr) to be of concern for phosphorus loading to surface water bodies (Robertson et al. 1998).

Continued study by Robertson and Harman (1999) at two of the septic system sites in Ontario revealed that two to four years after septic system decommissioning, groundwater phosphate concentrations persisted at the same concentrations observed during active sewage loading. The sorption reactions that hold phosphate to the subsurface solids are readily reversible, and phosphate is released from the subsurface solids long after the sewage source is removed (Robertson and Harman 1999). While sorption initially substantially retards phosphate migration velocity in the subsurface, phosphate may still migrate downgradient and impact receiving water bodies after septic system decommissioning.

Septic systems are not the only source of phosphorus to surface water bodies. Wastewater treatment plants also contribute phosphorus to rivers. Phosphate in directly-discharged wastewater treatment effluent is not subject to sorption in the soil; therefore, treatment plants that do not employ tertiary treatment to remove phosphorus contribute more phosphorus per person to receiving waters than do septic systems. Surface water phosphorus loading from septic systems is similar to nitrogen loading from septic systems because nutrient concentrations are reduced during groundwater transport. Because phosphorus does not pose a direct threat to human health as a groundwater contaminant, septic systems may be a more efficient method for treating domestic wastewater with respect to phosphorous than would primary or secondary wastewater treatment plants; however, like nitrogen, the phosphorus attenuation capacity of septic systems must be considered in the context of other contaminants that are introduced to groundwater by these systems.

9. CONCLUSIONS AND RECOMMENDATIONS

9.1 MAGNITUDE OF NITROGEN SOURCES

Significant reductions in the nitrogen load to the Wood River Valley Watershed must involve reductions in nitrogen exported from agricultural sources.

9.1.1 Dominance of Agricultural Sources

Agricultural sources are the largest nitrogen contributors to the Wood River Valley Watershed. Cropland and pasture, rangeland, and feed-lots contribute 89% of the watershed nitrogen load. From these different land uses, cattle waste and fertilizer are the two largest sources. Cattle waste alone accounts for 42% of the nitrogen load to the watershed, and fertilizer and cattle waste together contribute 70%. If the more residential Upper Valley and Northern Valley sub-watersheds are considered separately, rangeland and cattle waste are still the single largest nitrogen-contributing land use and source in those sub-watersheds.

9.1.2 Achieving Agricultural Source Reductions

The primary agricultural sources of nitrogen—fertilizer and cattle waste—are directly related to agricultural production. Fertilizer use increases crop yields in the watershed, and cattle waste is a by-product of grazing and feeding operations for beef production. Because methods for maintaining production and reducing agricultural nitrogen loads are limited, significant nitrogen reductions in the Wood River Valley Watershed will involve a trade-off with agricultural production. Substantial reductions from the two primary agricultural land uses (rangeland and cropland) can be achieved by limiting production.

Rangeland: Currently, there are approximately four times as many cattle in the watershed as there are people. Unlike human nitrogen contributions in residential areas, no economically feasible methods exist for capturing and treating the rangeland cattle waste that constitutes a significant portion of the watershed nitrogen load. Nitrogen reduction from implementation of rangeland nitrogen BMPs is not easily quantifiable and may have a limited effect because of the magnitude of the cattle waste source.

Significant reductions in rangeland nitrogen loads from cattle waste must involve reducing the cattle density and related beef production in the watershed. Most rangeland in the watershed is held in the public trust by BLM and the U.S. Forest Service; consequently, animal grazing densities are not locally controlled. Local governments, ranchers, and citizens groups can encourage less dense grazing in federally-controlled lands for the reduction of the watershed nitrogen load.

Cropland: Fertilizer application rate BMPs are well researched, but the implementation of BMPs in the Wood River Valley Watershed will result in a predicted total nitrogen load reduction of only 2.5%. Precision Agricultural techniques, which represent the forefront of current agricultural nitrogen research, will achieve a predicted 12.6% reduction in total watershed nitrogen load if implemented to their full extent. Precision Agriculture represents the best-case scenario for reducing agricultural fertilizer inputs while maintaining current crop yields. More significant reductions can be achieved by changes in cropping patterns and crop yield expectations, both of which may involve producing fewer crops. Changes in cropping patterns that would reduce nitrogen fertilizer use include letting fields lie fallow or growing crops that have a smaller nitrogen requirement. Lowering crop yield expectations (and therefore nitrogen fertilization rates) can significantly reduce nitrogen yields. Whereas fertilizer BMPs and Precision Agriculture seek to maintain or improve crop yields while reducing fertilizer application, lowering crop yield expectations will lower production as a means to significantly reduce fertilization rates and subsequent nitrogen load. Attaining a voluntary reduction in the fertilizer nitrogen load requires local farmers to evaluate the costs of decreased production versus the benefits to surface water quality. Because no direct incentives exist for farmers to sacrifice their income from crop production for the benefit of local water quality, government regulation is the only realistic mechanism for reducing cropland nitrogen loads.

9.2 WASTEWATER TREATMENT

The level of wastewater treatment is critical in determining the effects of continued development on nitrogen water quality in the watershed. Expanding the availability of tertiary wastewater treatment to a larger portion of the watershed population will accommodate future population growth with the smallest nitrogen loads to ground and surface waters. Secondary wastewater treatment plants that discharge directly to surface waters offer essentially no reduction in wastewater nitrogen concentrations, and are therefore the least efficient treatment method with respect to riverine nitrogen loads. Septic systems offer some nitrogen removal through aquifer denitrification as septic plumes travel through the subsurface; consequently, septic systems may reduce the impact of an increasing population on the magnitude of future watershed nitrogen loads. Gains in surface water quality from the use of septic systems must be evaluated in the context of viral and bacterial contamination of groundwater from these systems. Policymakers must evaluate the risks and benefits of septic system use to both public health and ecosystem health in making decisions about the nature of future residential development in the watershed.

9.3 NITROGEN MONITORING

Monitoring of nitrogen concentrations in ground and surface water should be expanded. Currently, surface water monitoring in the Wood River Valley watershed is conducted primarily by USGS, and groundwater monitoring is conducted by USGS, IDEQ, and Blaine County. In general, both of these monitoring schemes are inadequate in providing a complete understanding of the current nature and extent of nitrogen contamination in the watershed. Monitoring schedules and methods should be redesigned to incorporate monitoring during times and in areas that are suspected to have elevated nitrogen concentrations.

9.3.1 Surface Water Monitoring

As discussed in Chapter 6 of this thesis, elevated nitrogen concentrations in surface waters are likely to occur during the winter months as a result of seasonal variations and during storm events as a result of altered hydrologic dynamics that occur during storms.

In an average year, surface water nitrogen concentrations are currently measured once a month during six months of the year (April through September). Calculating the mass of nitrogen exported from the watershed in a given year based on only these six data points provides an inaccurate estimate of the true magnitude of the rivers' nitrogen load. The current monitoring schedule likely does not capture the vulnerability associated with winter and storm discharges during which the highest nitrogen concentrations are expected to occur.

Monthly, scheduled measurements of in-stream nitrogen concentrations should be expanded to include measurements during winter months, and unscheduled monitoring during storm events should be introduced to the surface water monitoring scheme. Because USGS maintains many monitoring stations within the state, it may not be feasible for this agency to expand its current monitoring. Local sources (i.e. local governments or citizens groups) may need to accept responsibility for introducing additional monitoring. USEPA provides resources for volunteer water monitoring groups, including factsheets to introduce citizens to water monitoring and manuals describing monitoring methods. National Volunteer Monitoring Conferences are also sponsored by USEPA to encourage information sharing among water monitoring volunteers (USEPA 2003). Expanding the extent of surface water monitoring in the watershed will enable better calibration of the BEANS model to observed in-stream concentration data. A broader nitrogen concentration data set that incorporates both seasonal and storm-related dynamics will also provide insight into the relationship between annual nitrogen loads and streamflow that we already observe with only a limited data set.

9.3.2 Ground Water Monitoring

While expanded groundwater monitoring is not specifically necessary to understand how accurate the BEANS model is in predicting the annual riverine nitrogen load, groundwater data are crucial in determining the magnitude of the public health effects associated with different nitrogen sources. Elevated nitrogen concentrations in groundwater are likely to occur in shallow groundwater below land uses that have high nitrogen yields (i.e. feed-lots, residential land, cropland and pasture). Groundwater flowpaths increase in depth as they travel from their source toward a receiving water body; consequently, the longer groundwater must travel, the deeper the flowpaths go (Figure 9-1).

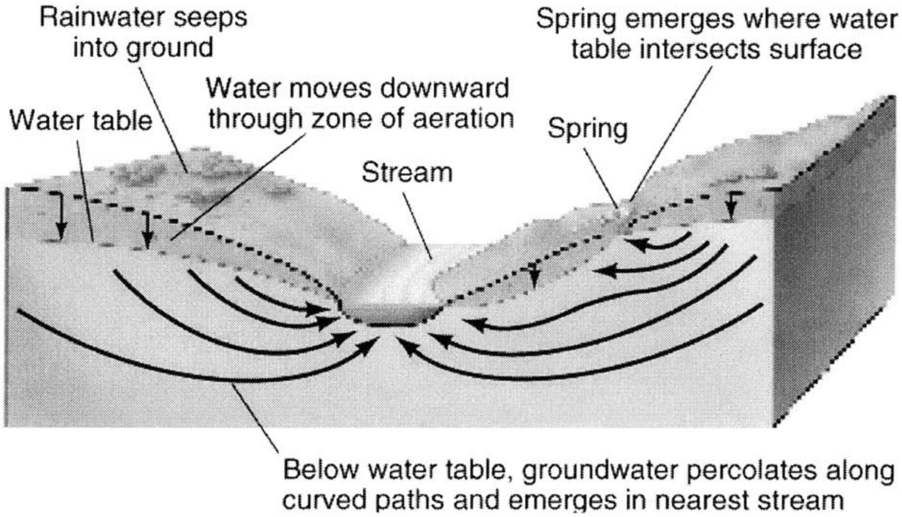


Figure 9-1: Groundwater Flowpaths from Land Surface to Receiving Water Body (source: Skinner, Porter, and Botkin 1999)

Contaminants that enter groundwater close to a river will likely not enter into deeper aquifer sections because they do not have the time to travel significant vertical distances before discharging into neighboring surface water bodies. Elevated nitrogen concentrations are not expected in deep groundwater in the Wood River Valley Watershed because of the close proximity of most nitrogen sources to river or stream channels. Currently, the Blaine County groundwater monitoring program relies primarily on relatively deep wells (usually drinking water wells) to assess the nature of groundwater nitrogen contamination. This expectation of lower nitrogen concentrations in deeper wells is observed in eastern Idaho in the Central Columbia Plateau NAWQA study unit (Williamson et al. 1998).

The groundwater monitoring network should be expanded to incorporate shallower wells. Because drinking water wells are designed to be deep enough to avoid contamination from surface sources, the construction of new shallow wells may be necessary. New shallow wells should be positioned so that they are likely to intercept nitrogen plumes from land uses with high nitrogen yields. One location that may be a good candidate for the introduction of new monitoring wells may be the cross-section of the valley at the outlet of the Northern Valley Watershed (Chapter 5). Because the aquifer is very narrow at this point, all of the groundwater from the Northern Valley is funneled through this cross-section. The installation of multi-level sampling wells across the valley at this location will provide an improved understanding of groundwater nitrogen contamination in the Northern Valley. Because residential land is the highest nitrogen yielding land use in the Northern Valley, much of the nitrogen contamination present at this cross-section likely originates from on-site, residential septic systems.

9.4 MAGNITUDE OF FUTURE NITROGEN LOADS

Watershed residents and public officials are in a position to control whether the magnitude of the nitrogen load to the Big and Little Wood Rivers increases or decreases in future years. Possibilities for nitrogen source reductions exist both in agricultural and residential sectors. As discussed earlier in this chapter, agricultural sources are currently the largest contributor of nitrogen to the watershed, and, consequently, individuals involved in agricultural production have the greatest potential to impact the magnitude of future nitrogen loads. Alternatively, because the watershed population is expected to continue to grow in future years, residential nitrogen sources represent the largest new potential source of nitrogen to the watershed. In making decisions about future development and land use changes in the watershed, residents and policymakers should keep in mind how these decisions will affect livestock densities, the acreage of high nitrogen-demanding crops, residential densities, and the availability of different wastewater treatment technologies to watershed residents.

LITERATURE CITED

33 USC § 1251-1387. Federal water pollution control act (Clean Water Act).

Agrium. 2001. Economics of Nitrogen Fertilizer and Crop Production. Retrieved: April 2003 from http://www.agrium.com/customer_centre/publications/1621.cfm.

Alexander, R., P.J. Johnes, E.W. Boyer, and R.A. Smith. 2002. A comparison of models for estimating the riverine export of nitrogen from large watersheds. *Biogeochemistry* 57/58: 295-339.

Alexander, R.B., R.A. Smith and G.E. Schwarz. 2000. Effect of stream channel size on the delivery of nitrogen to the Gulf of Mexico (Supplementary information). *Nature* 403: 758-761.

Alexander, R.B., J.R. Slack, A.S. Ludtke, K.K. Fitzgerald, and T.L. Schertz. 1998. Data from selected U.S. Geological Survey national stream water quality monitoring networks. *Water Resources Research* 34 (9): 2401-2405.

Alhajjar, B.J., S.L. Stramer, D.O. Cliver, and J.M. Harkin. 1988. Transport Modeling of Biological Tracers from Septic Systems. *Water Research* 22 (7): 907-915.

American Water Resources Association. 1987. Regional Aquifer Systems of the United States: Aquifers of the Western Mountain Area. Eds. John S. McLean and A. Ivan Johnson. AWRA Monograph Series No. 14. Bethesda, D.

Baldwin, J., D. Brandt, E. Hagan, and B. Wicherski. 2000. Idaho Department of Environmental Quality. Cumulative Impacts Assessment, Thousand Springs Area of the Eastern Snake River Plain, Idaho. Ground Water Quality Technical Report No. 14. Boise, ID.

Beck, G. Telephone Interview. 14 January 2003.

Blackmer, A.M., and B. Van De Woestyne. 2002. Corn yields at reduced rates of spring nitrogen in 2001. *Integrated Crop Management* 488 (3). Retrieved: April 2003 from http://www.ipm.iastate.edu/ipm/icm/2002/3-18-2002/reducedn.htm.

Blaine County. 1999. Blaine County, Idaho Land Capacity Study. Prepared for Board of Commissioners. Blaine County, ID.

Blaine County. 2002. Blaine County, Idaho. County Code. Retrieved: October 2002 from http://66.113.195.234/ID/Blaine%20County/index.htm.

Boyer, E.W., C.L. Goodale, N.A. Jaworski, R.W. Howarth. 2002. Anthropogenic nitrogen sources and relationships to riverine nitrogen export in the northeastern U.S.A.. *Biogeochemistry* 57/58: 137-169.

Brawley, J.W., G. Collins, J.N. Kremer, C.H. Sham, and I. Valiella. 2000. A time dependent model of nitrogen loading to estuaries from coastal watersheds. *Journal of Environmental Quality* 29: 1448-1461.

Brown, L. Personal Interview. 7 January 2003.

Buffman, I., J.N. Galloway, L.K. Blum, and K.J. McGlathery. 2001. A stormflow/baseflow comparison of dissolved organic matter concentrations and bioavailability in an Appalachian stream. *Biogeochemistry* 53: 269-306.

Canter, L.W., and R.C. Knox. *Septic Tank System Effects on Ground Water Quality.* Lewis Publishers, Inc. Chelsea, MI: 1985.

Carpenter, S.R., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley, and V.H. Smith. 1998. Nonpoint Pollution of Surface Waters with Phosphorus and Nitrogen. *Ecological Applications* 8 (3).

Cherkauer, D.S., P.F. McKereghan, and L.H. Schalch. 1992. Delivery of chloride and nitrate by ground water to the Great Lakes: Case study for the Door Peninsula, Wisconsin. *Ground Water* 30: 885-894.

DeBorde, D.C., W.W. Woessner, B. Lauerman, and P.N. Ball. 1998. Virus occurrence and transport in a school septic system and unconfined aquifer. *Ground Water* 36 (5): 825-834.

Delwiche, C.C. ed. *Denitrification, Nitrification, and Atmospheric Nitrous Oxide.* John Wiley & Sons, Inc. New York, NY: 1981.

Domenico, P.A., and F.W. Schwartz. *Physical and Chemical Hydrogeology.* John Wiley & Sons, Inc. New York, NY: 1998.

Environmental Defense. 2002. Scorecard. Retrieved: December 2002 from http://www.scorecard.org/.

Ericson, R. Personal Interview. 9 January 2003.

Ernst, R.A. 1995. Chicken Meat Production in California. Poultry Fact Sheet No. 20. Cooperative Extension. University of California. Retrieved: January 29, 2003 from http://animalscience.ucdavis.edu/Avian/pfs20.htm.

Faeth, P. *Fertile Ground: Nutrient Trading's Potential to Cost-Effectively Improve Water Quality.* World Resources Institute. Washington, DC: 2000

Gardner, R. Personal Interview. 15 January 2003.

Gerritse, R.G., J.A. Adeney, and J. Hosking. 1995. Nitrogen Losses From a Domestic Septic Tank System on the Darling Plateau in Western Australia. *Water Research* 29 (9): 2055-2058.

Gillham, R.W., and J.A. Cherry. 1978. Field evidence of denitrification in shallow groundwater flow systems. *Water Pollution Research in Canada* 13: 53-71.

Gillham, R.W. 1991. Nitrate contamination of ground water in southern Ontario and the evidence for denitrification. Pp. 181-198 in Bogárdi, I. and R.D. Kuzelka, editors. Nitrate Contamination. NATO ASI Series G, Vol. 30. Springer-Verlag, Berlin.

Group of Experts on the Scientific Aspects of Marine Pollution (GESAMP). 1990. State of the Marine Environment. Reports and Studies Number 39. United Nations Environment Programme.

Hansen, T. Telephone. 16 January 2003.

Harman, J., W.D. Robertson, J.A. Cherry, and L. Zanini. 1996. Impacts on a Sand Aquifer from an Old Septic System: Nitrate and Phosphate. *Ground Water* 34 (6): 1105-1114.

Hendry, M.J., R.G.L. McCready, and W.D. Gould. 1984. Distribution, source, and evolution of nitrate in a glacial till of southern Alberta, *Canada. Journal of Hydrology* 70: 177-198.

Howarth, R. 1988. Nutrient limitation of net primary production in marine ecosystems. *Annual Review of Ecology and Systematics* 19: 89-110.

Howarth, R.W., G. Billen, D. Swaney, A. Townsend, N. Haworski, K. Lajtha, J.A. Downing, R. Elmgren, N. Caraco, T. Jordan, F. Berendse, J. Freney, V. Kudeyarov, P. Murdoch, and Zhu Zhao-Liang. 1996. Regional nitrogen budgets and riverine N & P fluxes for the drainages to the North Atlantic Ocean: Natural and human influences. *Biogeochemistry* 35: 75-139.

Hursch, K. 2002. Hursch Consulting and Communications. Retrieved: January 29, 2003 from http://www.hursh.ca/2002_07_14_archive.asp.

Hyde, R. Personal Interview. 6 January 2003.

Idaho Department of Environmental Quality. 2002. The Big Wood River Watershed Management Plan. Prepared by Balthasar B. Buhidar. Twin Falls, ID.

Idaho Department of Health and Welfare. 2000. Technical Guidance Manual for Individual and Subsurface Sewage Disposal Systems. I.D.H.W. Division of Environmental Quality.

Idaho Department of Water Administration. 1972. Water resources of the Big Wood River – Silver Creek area, Blaine County, Idaho. By Paul M. Castelin and Sherl L. Chapman. Water Information Bulletin No. 28. Boise, ID.

Idaho Department of Water Resources. Water Use Information. Retrieved: April 2003 from http://www.idwr.state.id.us/water/wateruse.htm.

International Fertilizer Industry Association. 1998. Fertilizer Statistics: Fertilizer Use by Crop. Retrieved: November 2002 from http://www.fertilizer.org/ifa/statistics.asp.

Jackson, G.D. 2000. Nitrogen Fertilization of Dryland Malt Barley for Yield and Quality. Retrieved: April 2003 from http://landresources.montana.edu/FertilizerFacts/.

Jones, R. Telephone Interview. 20 January 2003.

Kaplan, O.B. 1991. *Septic System Handbook.* Lewis Publishers.

Kent, K.M. 1972. National Engineering Handbook. Section 4: Hydrology. Chapter 15: Travel time, time of concentration, and lag. Natural Resource Conservation Service. Retrieved: January 2003 from www.wcc.nrcs.usda.gov/water/quality/common/neh630/630ch15.pdf

Lopez-Bernal, K. 2003. Design of a Watershed-Based Nitrogen Trading System for the Big and Little Wood Rivers Watershed. Thesis, Massachusetts Institute of Technology. Cambridge, MA.

Mahler, R.L., T.A. Tindall, and K.A. Mahler. 1992. Best Management Practices for Nitrogen Management to Protect Groundwater. University of Idaho, College of Agriculture. Cooperative Extension System. Current Information Series No. 962. Moscow, ID.

Mahler, R.L. 1999. Northern Idaho Fertilizer Guide: Alfalfa. University of Idaho, College of Agriculture. Cooperative Extension System. Current Information Series No. 447. Moscow, ID.

McFall, W., and J. Wood. 2003. Idaho OnePlan. Retrieved: March 2003 from http://www.oneplan.org.

McFarland, J.A. 1989. Nitrogen transport in groundwater in two geologic settings, Patuxet River Basin, Maryland. Pp. 105-124 in Ground Water Issues and Solutions in the Potomac River Basin/Chesapeake Bay Region. National Water Well Association. Dublin, OH.

Moreland, J.A. 1977. United States Geological Survey. Idaho Department of Water Resources. Ground water – surface water relations in the Silver Creek area, Blaine County, Idaho. Water Information Bulletin No. 45. Boise, ID.

NASS, 1997. United States Census of Agriculture. 1997. National Agricultural Statistics Service. Retrieved: October 2002 from http://www.nass.usda.gov/census/.

National Atmospheric Deposition Program. 2002. NADP/NTN Monitoring Location ID03. Retrieved: October 2002 from http://nadp.sws.uiuc.edu/sites/siteinfo.asp?net=NTN&id=ID03

National Stream Quality Accounting Network. 2003. NASQAN and HBN Station Map. Retrieved: April 2003 from http://water.usgs.gov/nasqan/progdocs/index.html.

Pabich, W.J., H.F. Hemond, and I. Valiela. 2003 (submitted). An empirical model to predict groundwater denitrification rates, Cape Cod, USA: substrate limitation by nitrate and DOC. Submitted to *Environmental Science and Technology.*

Parsons Brinckerhoff Quade & Douglas. 2002.Timmerman to Ketchum Environmental Analysis. Revised SH-75 Corridor Year 2025 Population and Employment Forecasts. Retrieved: January 2003 from http://www.sh-75.com/.

Prairie Swine Center. 2003. Retrieved: January 29, 2003 from http://adminsrv.usask.ca/psci/default.htm

Purdy, N. Personal Interview. 15 January 2003.

Reddell, D.L., G.G. Wise, R.E. Peters, and P.J. Lyerly. 1973. Water quality hydrology of lands receiving farm animal wastes. Technical Report No. 50. Retrieved: January 29, 2003 from http://twri.tamu.edu/reports/1973/50.html.

Robert, P.C. 2002. Precision agriculture: a challenge for crop nutrient management. *Plant and Soil* 247: 143-149.

Robertson, W.D., J.A. Cherry, and E.A. Sudicky. 1991. Ground-Water Contamination from Two Small Septic Systems on Sand Aquifers. *Ground Water* 29 (1): 82-92.

Robertson, W.D. and J. Harman. 1999. Phosphate plume persistence at two decommissioned septic system sites. *Ground Water* 37 (2): 228-236.

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

Rupert, M.G. 1996. United States Geological Survey. Major sources of nitrogen input and loss in the Upper Snake River Basin, Idaho and western Wyoming, 1990. Water Resources Investigations Report 96-4008. Boise, ID.

Scandura, J.E., and M.D. Sobsey. 1997. Viral and Bacterial Contamination of Groundwater from On-site Sewage Treatment Systems. *Water Science and Technology* 35 (11-12): 141-146.

Sims, A.L., J.T. Moraghan, and L.J. Smith. 2002. Spring Wheat Response to Fertilizer Nitrogen Following a Sugar Beet Crop Varying in Canopy Color. *Precision Agriculture* 3 (3): 283-295.

Skinner, B., S. Porter, and D. Botkin. *The Blue Planet: An Introduction to Earth System Science*. John Wiley & Sons, Inc. New York, NY: 1999.

Smith, R.A., G.E. Schwarz, and R.B. Alexander. 1997. Regional interpretation of water-quality monitoring data. *Water Resources Research* 33 (12): 2781-1798.

Starr, R.C. and R.W. Gillham. 1993. Denitrification and organic carbon availability in two aquifers. *Ground Water* 31 (6): 934-947.

Stelljes, K.B. 2000. New Systems Research Targets Precision Agriculture's Effectiveness. *Agricultural Research Magazine*. Retrieved: April 2003 from http://www.ars.usda.gov/is/AR.

Stendal, G. 2002. FY 2002 Columbia Plateau Province Proposal: Restore Tucannon River Riparian Habitat: Wooten Wildlife Area. Retrieved: May 2003 from http://www.cbfwa.org/files/province/plateau/projects/25072.htm.

Stoll, A. and H.D. Kutzbach. 2000. Guidance of a Forage Harvester with GPS. *Precision Agriculture* 2 (3): 281-291.

Swindell, D. Personal Interview. 8 January 2003.

Trudell, M.R., R.W. Gillham, and J.A. Cherry. 1986. An in-situ study of the occurrence and rate of denitrification in a shallow unconfined aquifer. *Journal of Hydrology* 83: 251-268.

Turner, R. Personal Interview. 15 January, 2003

United States Census Bureau. 2000. Census 2000 data for the state of Idaho. Retrieved: October 2002 from http://www.census.gov/census2000/states/id.html.

United States Environmental Protection Agency. 1990. National Pesticide Survey: Nitrate. U.S. E.P.A. Office of Water, Office of Pesticides and Toxic Substances. p. 7.

United States Environmental Protection Agency. 1995. Cleaner Water Through Conservation. Retrieved April 2003 from http://www.epa.gov/OW/you/chap1.html.

United States Environmental Protection Agency. 2001. Source Water Protection Practices Bulletin: Managing Septic Systems to Prevent Contamination of Drinking Water. U.S. E.P.A. Office of Water.

United States Environmental Protection Agency. 2002. Groundwater and Drinking Water: Current Drinking Water Standards. Retrieved: November 7, 2002 from http://www.epa.gov/safewater/mcl.html#1.

United State Environmental Protection Agency. BASINS. 2002. BASINS GIS data. Retrieved: November 2002 from http://www.epa.gov/waterscience/ftp/basins/gis_data/huc/.

United States Environmental Protection Agency. Natural Channel Restoration on the Soque River, Georgia. Retrieved: May 2003 from http://www.epa.gov/region4/water/wetlands/projects/soqueepa.html.

United States Environmental Protection Agency. 2003. Volunteer Monitoring. Retrieved: May 2003 from http://www.epa.gov/owow/monitoring/vol.html.

United States Geological Survey. 2003. Idaho Water Use Data. Retrieved: January 2003 from http://idaho.usgs.gov/public/water.use.html.

Valiela I., G.Collins, J. Kremer, K. Lajtha, M. Geist, B. Seely, J Brawley, CH Sham. 1997. Nitrogen loading from coastal watersheds to receiving estuaries: New method and application. *Ecological Applications* 7(2): 358-380.

Valiela, I., J. Costa, K. Foreman, J.M. Teal, B.L. Howes, and D. Aubrey. 1990. Transport of groundwater-borne nutrients from watersheds and their effects on coastal waters. *Biogeochemistry* 10: 177-197.

Van Alphen, B.J., and J.J. Stoorvogel. 2000. A Methodology for Precision Nitrogen Fertilization in High-Input Farming Systems. *Precision Agriculture* 2 (4): 319 332.

Vigil, M.F. 2003. Conservation Tillage Fact Sheet. Retrieved: March 2003 from http://www.akron.ars.usda.gov/fs_fert.html.

Vrindts, E., J. De Baerdemaeker, and H. Ramon. 2002. Weed Detection Using Canopy Reflection. *Precision Agriculture* 3 (1): 63-80.

Waskom, R.M. 1994. Best Management Practices for Nitrogen Fertilization. Colorado State University Cooperative Extension. Fort Collins, CO.

Webb, D. Telephone Interview. 20 January 2003.

Wilhelm, S.R., S.L. Schiff, and W.D. Robertson. 1996. Biogeochemical evolution of domestic waste water in septic systems: 2. Application of conceptual model in sandy aquifers. *Ground Water* 34 (5): 853.

Williamson, A. K., M.D. Munn, S.J. Ryker, R.J. Wagner, J.C. Ebbert, and A.M. Vanderpool. National Water Quality Assessment Program. 1998. Water Quality in the Central Columbia River Plateau, Washington and Idaho, 1992-1995. United States Geological Survay Circular No. 1144. Denver, CO.

Wright, D. Personal Interview. 16 January 2003.

Yang, C. and G.L. Anderson. 2000. Mapping Grain Sorghum Yield Variability Using Airborne Digital Videography. *Precision Agriculture* 2 (1): 7-23.