Appendix A

Water Quality Technical Report

A.C. Kindig & Co.
ENVIRONMENTAL CONSULTING

Trendwest Properties: Cle Elum UGA FEIS Water Quality Technical Report

PREPARED FOR

**Trendwest Resorts, Inc.
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P.O. Box 887
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**Project No. 114
March 3, 2002**

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EXHIBITS

1.0 INTRODUCTION

1.1 Supplemental Evaluation

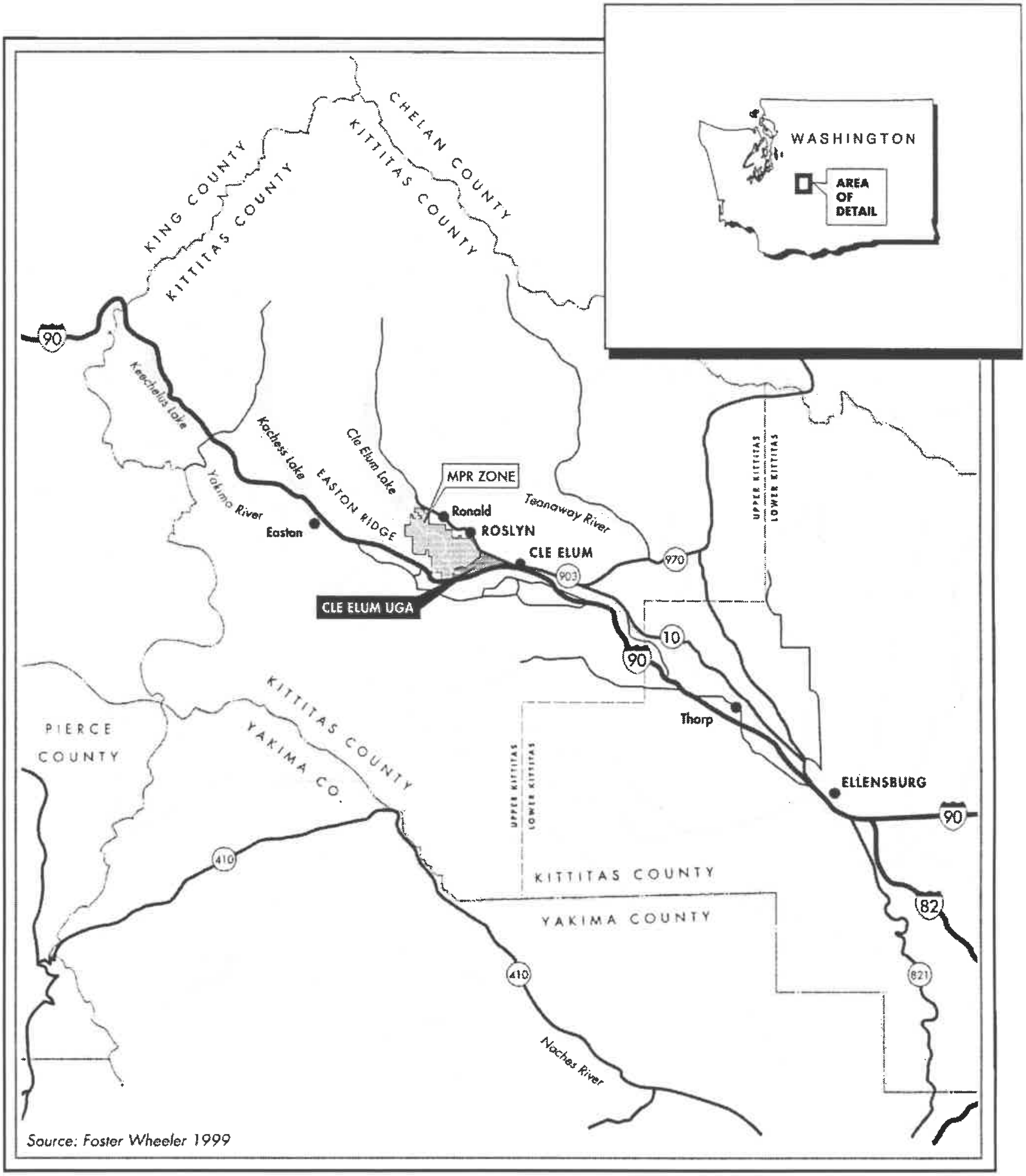
Trendwest proposes to develop 1,106 acres of property within the Cle Elum Urban Growth Area (UGA) located in Kittitas County, Washington (Figure 1-1). The property is bordered by State Route (SR) 903 to the east, Bullfrog Road to the north and west, and Interstate 90 to the south (Figure 1-2). The approved MountainStar Master Planned Resort (MPR) lies adjacent to and northwest of the UGA. The proposal for the UGA consists of a master plan community containing single-family and multi-family residential, commercial, and recreational land uses. Four alternatives were previously evaluated in the Draft Environmental Impact Statement (DEIS) prepared for the Cle Elum UGA (City of Cle Elum 2001). A fifth development alternative has been added since the publication of the UGA Draft EIS. A.C. Kindig & Co. was requested to prepare a supplemental water quality report for the analysis in the Final EIS. The supplemental report adds to the water quality analysis and quantitatively assesses Alternatives 5 (Preferred Alternative). This report contains an additional evaluation of construction impacts, as well as an analysis of post-developed stormwater runoff, treatment, infiltration, and groundwater quality impacts.

1.2 UGA Alternative 5 and MPR Reduced Density Summaries

The Conceptual Site Plan for Alternative 5 represents another site plan variation for development of the Cle Elum UGA that incorporates many of the land uses evaluated under Alternatives 2, 3, and 4 in the Cle Elum UGA Draft EIS. Alternative 5 has been designed to result in a level of environmental impact that falls within the range of environmental impacts evaluated in the Cle Elum UGA Draft EIS. Alternative 5 contains residential land uses (293 acres), a Business Park campus (80 acres), recreation/public facilities (134 acres), open space (418 acres), and a Reserve (175 acres) (Figure 1-3).

Cumulative impacts for Alternative 5 are being evaluated in combination with a Reduced-Density MPR and the September 2000 MPR Conceptual Master Plan submitted to Kittitas County. The reduction in density reflects terms of the settlement agreement between Trendwest and the RIDGE, a Washington non-profit corporation. The density for the MPR has decreased from 4,650 units to 3,785 units.

The primary difference between the September 2000 MPR Master Plan and the conceptual plan included in the MPR FEIS is the timing of golf course construction. In the September 2000 Plan, two golf courses are proposed for Phase I development and a potential golf course is proposed for Phase 3 development. In the MPR FEIS, one golf course was proposed in Phase 1 development with open space shown as a potential golf course.



REFERENCE: SHAPIRO AND ASSOCIATES.

A.C. Kindig & Co.	Water Quality Technical Report Cle Elum UGA FEIS	Figure 1-1 Vicinity Map

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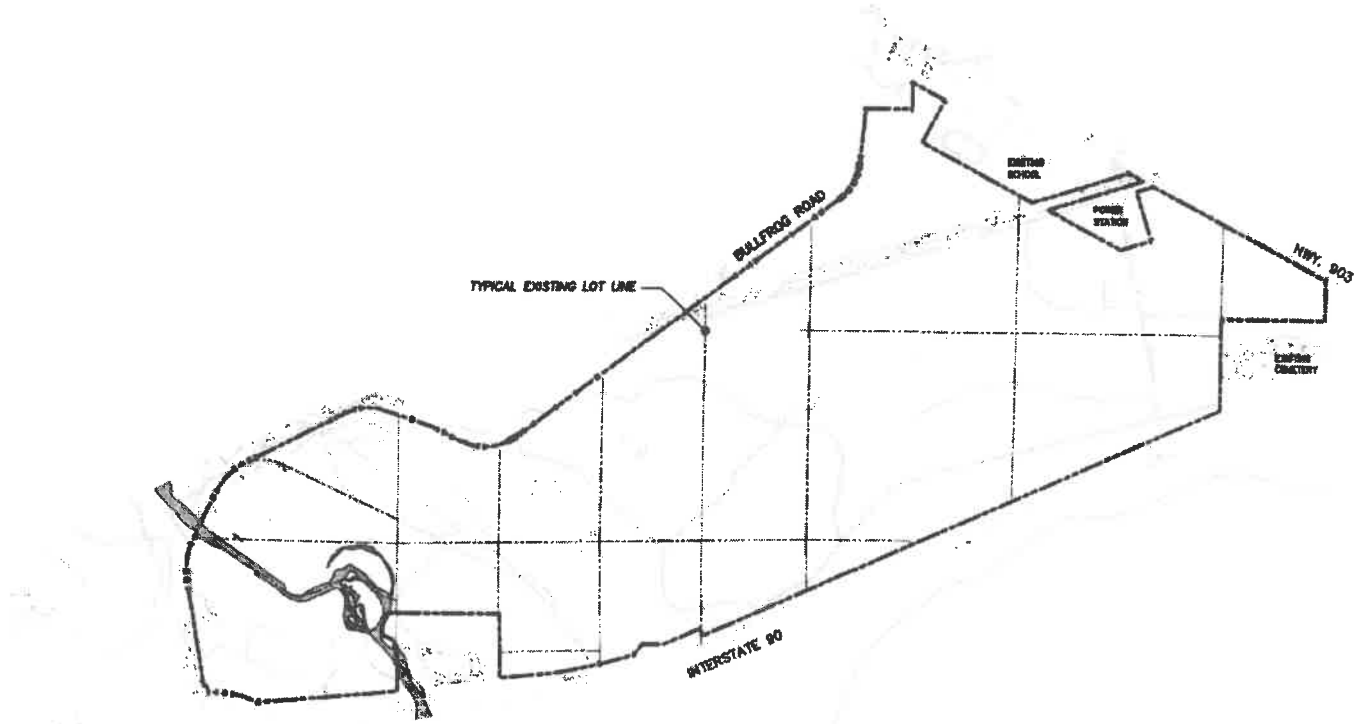

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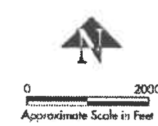
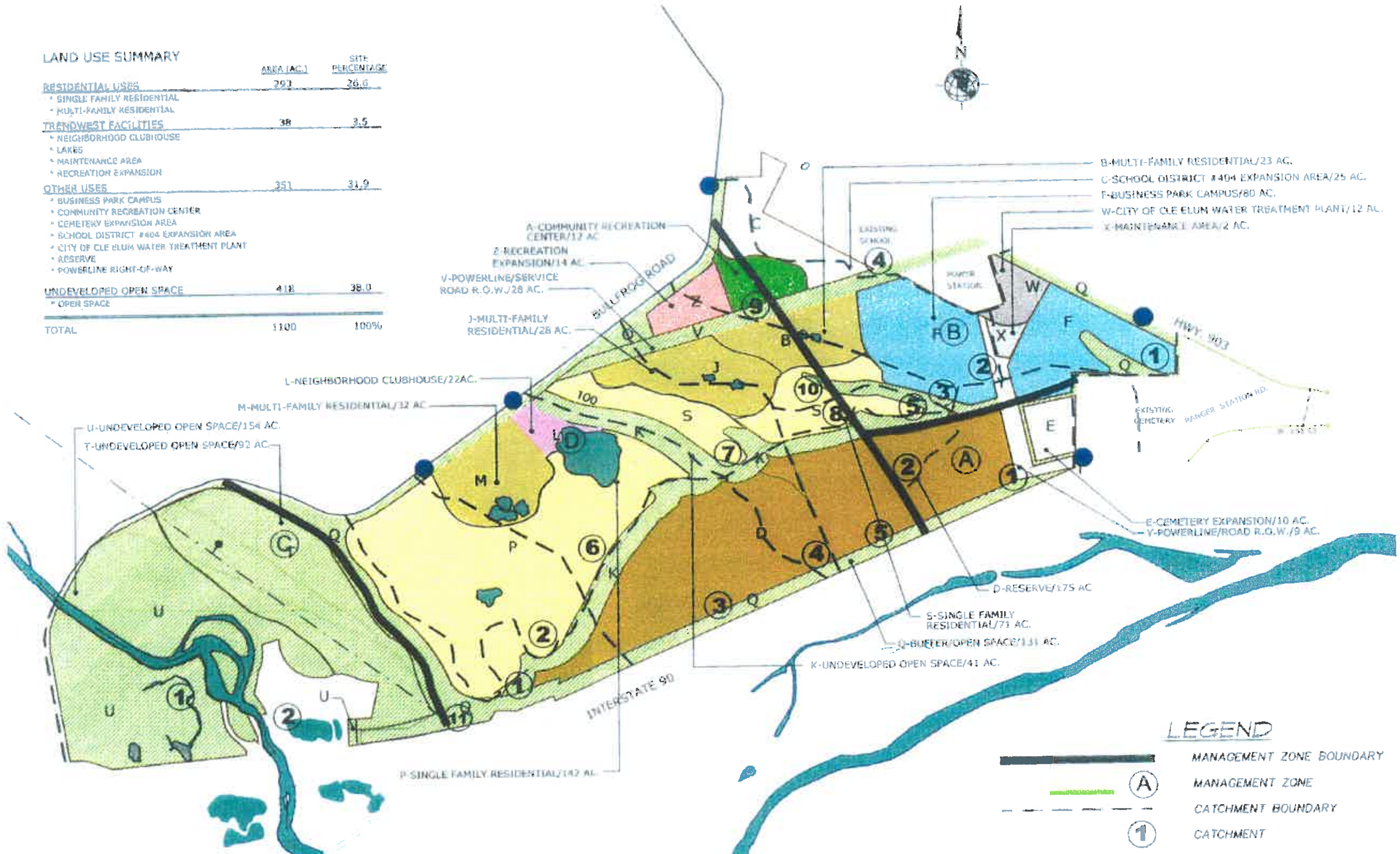


REFERENCE: SHAPIRO AND ASSOCIATES.

A.C. Kindig & Co.	*Water Quality Technical Report* *Cle Elum UGA FEIS*	**Figure 1-2** **Cle Elum UGA Site Map**



REFERENCE: W&H PACIFIC, DATED 8/28/01.

A.C. Kindig & Co.	Water Quality Technical Report Cle Elum UGA FEIS	Figure 1-3 Alternative 5 Land Use Plan

2.0 AFFECTED ENVIRONMENT

2.1 Hydrologic Setting

The Cle Elum UGA Draft EIS previously described the regional hydrologic setting (Section 3.3.1). The proposed Cle Elum UGA site lies within the upper Yakima River drainage basin, which is designated as Water Resource Inventory Area (WRIA) 39 (Washington State Department of Fisheries [WDF] 1975). The UGA site is adjacent to the lower portion of the Cle Elum River between Bullfrog Road and Interstate 90. The Cle Elum River runs along the western boundary of the site and joins the Yakima River at River Mile (RM) 185.6. The Yakima River and Interstate 90 run along the southern boundary of the site. Approximately 750 acres of the UGA site is topographically located within the Yakima River basin, and approximately 350 acres is topographically within the Cle Elum River basin. Due to the nature of surface soils on the site, natural drainage from the site occurs through infiltration and subsurface groundwater flow. The Cle Elum River flows are controlled at the Cle Elum Dam operated by the United States Bureau of Reclamation (USBR). The dam is upstream of the project at RM 8.2. Water impounded by the dam forms Cle Elum Lake, which the USBR uses primarily for storing fall, winter and spring flows to supply late-spring through early fall irrigation demands in the Yakima Valley. A secondary function of the dam is flood control.

2.2 Surface Water Quality

The Cle Elum River from the mouth to Cle Elum Dam (RM 8.2) is identified as water body segment WA-39-1050, and designated Class AA (extraordinary) for water quality (Table 2-1) (Chapter 173-201A WAC).

The Yakima River is designated as Class A (excellent) water quality from its mouth to the confluence with the Cle Elum River, and is designated Class AA (extraordinary) for the reach from the Cle Elum River confluence (RM 185.6) up to its headwaters (Table 2-1) (Washington State Department of Ecology [Ecology] 2000, Chapter 173-201A-130 WAC). A special condition is applied in this reach (RM 185.6 to the headwaters of the Cle Elum River) in that the temperature shall not exceed 21.0°C due to human activities.

2.2.1 Cle Elum River (WA-39-1434)

Water quality data for the Cle Elum River collected by the United States Geological Survey (USGS), Ecology, and Associated Earth Sciences, Inc. (AESI), were summarized in the Draft EIS.

2.2.2 Yakima River

Water quality data from 1997 and 1998 were described in the Draft EIS. Ecology's ambient water quality monitoring program collected water quality data from 1989 through September 2000 (Ecology 2001a). To characterize the existing Yakima River water quality for this report, the Draft EIS data set was updated to include all of Ecology's data collected from October 1994 through 2000. Water quality samples were collected monthly by Ecology from October 1994 through September 2000 at RM 191 located upstream of the UGA site (72 samples).

Table 2-1
Class AA and Class A Water Quality Standards

Class AA (extraordinary) water quality standards for the Cle Elum River from the mouth to the headwaters (Chapter 173-201A-130[17] WAC).

Fecal coliforms	Organism levels shall both not exceed a geometric mean value of **50** colonies/100 mL and not have more than 10 percent of all samples obtained for calculating the geometric mean value exceeding 100 colonies/100 mL.
Dissolved oxygen	Shall exceed **9.5** mg/L. Total dissolved gas shall not exceed 110 percent of saturation at any point of sample collection.
Temperature	Shall not exceed **16.0**°C due to human activities. When natural conditions exceed 18.0°C, no temperature increases will be allowed that will raise receiving water temperatures by greater than 0.3°C.
pH	Shall be within the range of **6.5 to 8.5** with a human-caused variation within a range of less than 0.5 units.
Turbidity	Shall not exceed **5** nephelometric turbidity units (NTU) over background turbidity when the background turbidity is 50 NTU or less, or have more than a 10 percent increase in turbidity when the background turbidity is more than 50 NTU.
Toxic substances	Shall not be introduced above natural background levels in waters of the state that have the potential either singularly or cumulatively to adversely affect characteristic water uses, cause acute or chronic toxicity of the most sensitive biota dependent upon those waters, or adversely affect public health, as determined by the department. (Toxic substances include dissolved metals and ammonia-nitrogen.)

Source: Ecology 2000

Class A (excellent) water quality for the Yakima River from the mouth to the confluence with the Cle Elum River (RM 185.6) (Chapter 173-201A-130[141] WAC).

Fecal coliforms	Organism levels shall both not exceed a geometric mean value of **100** colonies/100 mL and not have more than 10 percent of all samples obtained for calculating the geometric mean value exceeding 200 colonies/100 mL.
Dissolved oxygen	Shall exceed **8.0** mg/L. Total dissolved gas shall not exceed 110 percent of saturation at any point of sample collection.
Temperature	Shall not exceed **18.0**°C due to human activities. When natural conditions exceed 18.0°C, no temperature increases will be allowed that will raise receiving water temperatures by greater than 0.3°C.
pH	Shall be within the range of **6.5 to 8.5** with a human-caused variation within a range of less than 0.5 units.
Turbidity	Shall not exceed **5** nephelometric turbidity units (NTU) over background turbidity when the background turbidity is 50 NTU or less, or have more than a 10 percent increase in turbidity when the background turbidity is more than 50 NTU.
Toxic substances	Shall not be introduced above natural background levels in waters of the state that have the potential either singularly or cumulatively to adversely affect characteristic water uses, cause acute or chronic toxicity of the most sensitive biota dependent upon those waters, or adversely affect public health, as determined by the department. (Toxic substances include dissolved metals and ammonia-nitrogen.)

Source: Ecology 2000

Yakima River temperatures at RM 191 averaged 6.8°C and ranged from 0.20°C to 19.8°C. Two samples exceeded the Class AA standard of 16°C. Dissolved oxygen (DO) ranged from 8.0 mg/L to 13.0 mg/L and averaged 11.1 mg/L. A total of 12 samples within the 1994 to 2000 time frame did not meet the 9.5 mg/L DO minimum criterion. The lower DO values occurred in the summer months of July through September. The pH in the Yakima River ranged from 6.4 to 8.3. One sample was below the minimum 6.5 pH criterion (June 1996). Fecal coliform concentrations averaged 11.1 colonies/100 mL and ranged from 1 to 160 colonies/100 mL. The Class AA standard of 50 colonies/100 mL as a geometric mean was met.

Nutrient concentrations in the Yakima River at RM 191 were low. Total phosphorous (TP), nitrate+nitrite-nitrogen, and ammonia-nitrogen averaged 0.07 mg/L, 0.04 mg/L, and 0.10 mg/L, respectively. Total suspended solids (TSS) were variable, ranging from less than 1.0 mg/L to 62 mg/L. The average TSS was 5.6 mg/L. Turbidity averaged 3.4 NTU and ranged from 0.60 NTU to 30 NTU. The Yakima River upstream of Yakima to its headwaters is being targeted for study and cleanup by Ecology due to high levels of suspended sediment, turbidity, and pesticides (Ecology 2001b). The Yakima River below the project at RM 80.4 (segment WA-37-1010) has a Total Maximum Daily Load (TMDL) for turbidity and the pesticide DDT (See the 303 (d) discussion below).

Dissolved metals were analyzed in the Yakima River at RM 191 by Ecology to verify previously measured violations of state water quality standards from historic USGS survey data (Johnson 2000). All dissolved metals complied with chronic standards when measured six times between March 1999 and January 2000 (Johnson 2000). Dissolved lead was less than the detection level of 0.02 µg/L. Dissolved copper averaged 0.19 µg/L and dissolved zinc averaged 0.75 µg/L. The average hardness at this station was 27 mg/L $CaCO_3$.

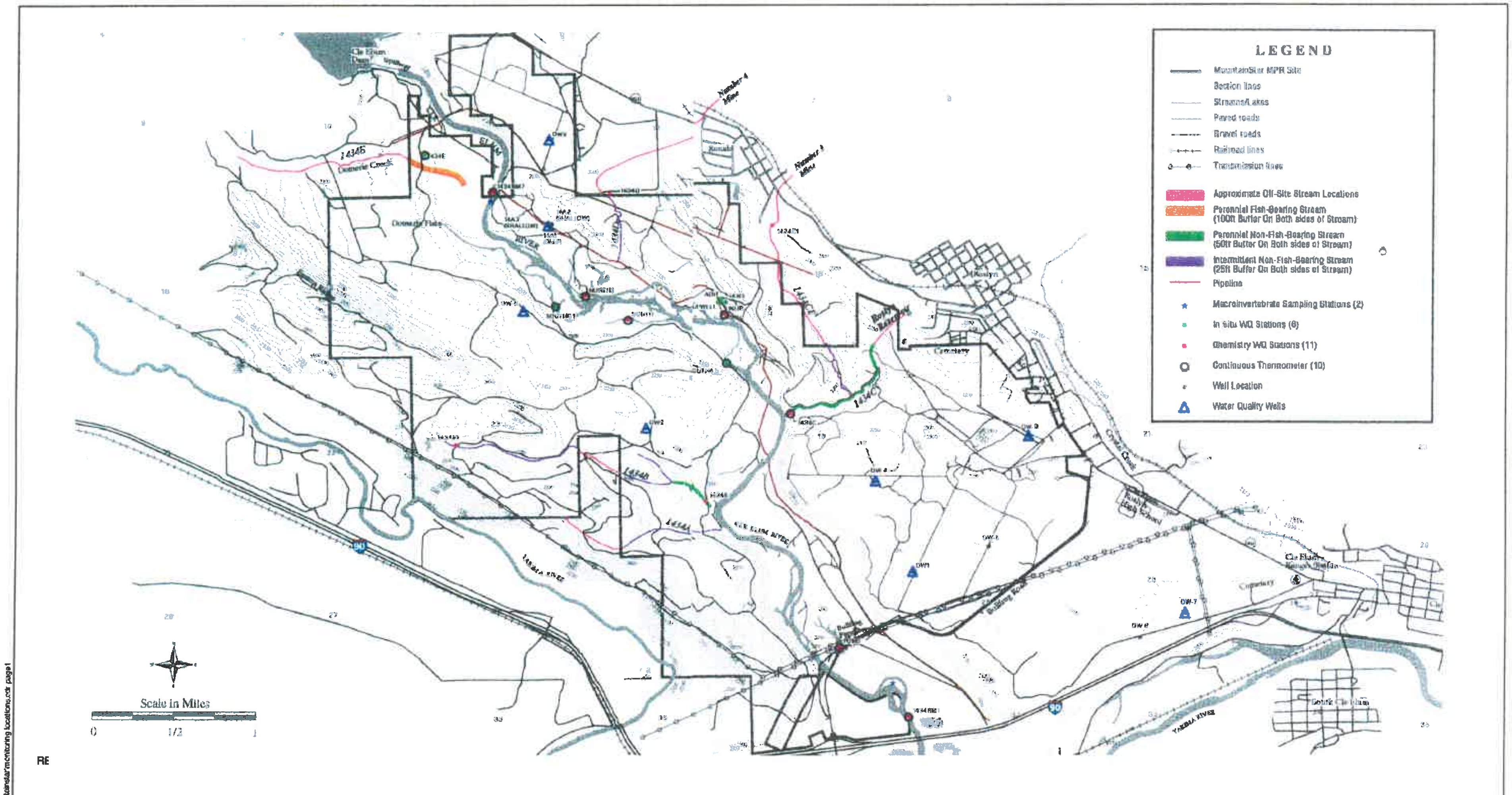
2.3 Groundwater Quality

Groundwater quality was described in the Draft EIS, and was based on results from four wells drilled (OW-7, OW-1, OW-4, and OW-9) for the MPR (Figure 2-1). Wells OW-9, OW-7 and OW-4 were sampled eight times between October 1998 and September 1999 (AESI 1999c). Well OW-1 was dry during five of the eight sampling events, therefore the water quality record is for three occasions (October and November 1998 and April 1999) (AESI 1999c).

2.4 Water Quality-Related Regulations

2.4.1 State Water Quality Standards

Surface waters in the State of Washington are regulated for quality by Chapter 173-201A WAC, administered through Ecology. These regulations classify surface waters into water quality categories, each with a defined range or maximum value for a variety of water quality constituents. The state water quality standards are intended to protect all beneficial uses of surface waters, including the protection of aquatic biota. As established above in Section 2.2,



Andy Kindig & Co.	*Water Quality Technical Report* *Cle Elum UGA FEIS*	***Figure 2-1*** ***Surface and Ground Water Quality Monitoring Station Locations***

surface waters near the site have a Class AA (extraordinary) and Class A (excellent) classification (Table 2-1).

2.4.2 Washington State Groundwater Standards (Ecology 1990, Chapter 173-200 WAC)

The goal of the Washington State groundwater quality standards is to protect groundwater quality and existing and future beneficial uses through an antidegradation policy (Chapter 173-200-030 WAC) and definition of maximum contaminant level (MCL) criteria (Table 2-2) (Chapter 173-200-040 WAC) (Ecology 1990). Existing water quality is to be protected and all contaminants proposed for entry to groundwater shall be provided with all known, available and reasonable methods of prevention, control and treatment prior to entry.

2.4.3 Washington State Drinking Water Standards (WDOH 1992, Chapter 246-290-300 WAC)

The purpose of drinking water regulations is to ensure health quality standards are maintained for public drinking water supplies. Drinking water standards established by the Washington Department of Health (WDOH) comply with the Federal Safe Drinking Water Act of 1974 and subsequent 1986 amendments (WDOH 1992, Chapter 246-290-300 WAC). The standards outline monitoring protocols and maximum contaminant levels (MCLs) for bacteriological, inorganic chemical and physical parameters, turbidity, trihalomethanes, pesticides, and volatile organic chemicals. The MCLs are divided into primary and secondary categories. Primary standards are based on chronic, non-acute, or acute human health effects. Secondary standards are based on factors other than human health effects. Groundwater standards and drinking water standards are similar, but not identical.

2.4.4 Section 303(d) Threatened and Impaired Water Bodies

Section 303(d) of the 1972 Federal Clean Water Act (CWA) requires states to identify and list threatened and impaired water bodies (Ecology 1998). The CWA requires the list to be updated and submitted for review and approval to the U.S. Environmental Protection Agency (EPA) every 2 years. The purpose of the listing is to identify segments where, with technology-based pollution control measures, the identified segments are not expected to meet the applicable standard(s) for the listed water quality parameter(s). The EPA allowed states to skip the year 2000 303(d) list due to the ongoing development of new federal rules affecting the listing process and the TMDL program. The next 2002 303(d) list is due in April of 2002. Under the amended CWA, the list is now required every 4 years instead of every 2 years.

The 1998 water quality limited list for Washington was approved by the EPA on January 28, 2000, and is the current active listing. The 1998 303(d) list includes the Cle Elum River as limited for temperature (Ecology 1998). The temperature listing was based on 26 excursions beyond the criterion from the mouth of the river up to Cle Elum Lake (segment WA-39-1050).

Table 2-2
Water Quality Standards for Groundwater in the State of Washington[1]

Contaminant	Maximum Contaminant Level (MCL) Criterion
Primary Contaminants	
Barium	1.0 mg/L
Cadmium	0.01 mg/L
Chromium	0.05 mg/L
Lead	0.05 mg/L
Mercury	0.002 mg/L
Selenium	0.01 mg/L
Silver	0.05 mg/L
Fluoride	4 mg/L
Nitrate-Nitrogen	10 mg/L
Total Coliform Bacteria	1/100 mL
Secondary Contaminants	
Copper	1.0 mg/L
Iron	0.30 mg/L
Manganese	0.05 mg/L
Zinc	5.0 mg/L
Chloride	250 mg/L
Sulfate	250 mg/L
Total Dissolved Solids	500 mg/L
pH	6.5-8.5 standard units
Color	15 color units
Radionuclides	
Gross Alpha Particle Activity	15 pico curie/L
Gross Beta Activity	50 pico curie/L

Source: Ecology 1990; Reference: Chapter 173-200-040 WAC.

(1) Partial listing of primary and secondary contaminants and radionuclide standard criteria. All heavy metal standard criteria are for total metals.

The 1998 303(d) list includes the Yakima River from RM 147 upstream to the Cle Elum River confluence at RM 185.6 for DDT, mercury, copper, cadmium, and the herbicide 4.4'-DDE (segment No. WA-39-1030). The Yakima River upstream of the site (segment WA-39-1060) is listed as limited for DO and temperature. The 1998 303(d) lists additional parameters for the Yakima River downstream of RM 147.

A TMDL for TSS and DDT was approved by the EPA for the Yakima River, segment WA-37-1010, on November 25, 1998. This water body segment extends from its mouth at the Columbia

River to RM 80.4, which is approximately 100 RM downstream from the UGA site. Ecology conducted a TMDL evaluation of the lower Yakima River basin in 1994 and 1995, in cooperation with the EPA and the Yakama Indian Nation. The TMDL evaluation focused on TSS and associated DDT loads from irrigated fields. Consequently, it is limited to the lower Yakima River basin during irrigation season. Turbidity targets for the mainstem Yakima and tributary sites are being instituted over an implementation schedule spanning 15 years (Joy 1997).

2.4.5 Washington State Water Quality Assessment 305(b) Report

Section 305(b) of the 1972 CWA requires all states to prepare biennial reports assessing the water quality of defined water bodies within the state. The 1998 report has been adopted, but the 1998 report is generic in scope with no specific water body information. Therefore, the 1994 list is used to characterize specific water bodies. The 1994 report prepared by Ecology addresses supported and impaired uses, sources, and causes of documented impairments of the Yakima River upstream and downstream of the project site (Ecology 1995). Yakima River supported uses upstream and downstream of the project include rearing, harvesting, and other fish spawning; salmonid spawning; and salmonid and other fish migration. Yakima River supported uses upstream of the project include primary and secondary contact recreation. Salmonid spawning is cited as impaired upstream and downstream of the project (Ecology 1995).

The Yakima River approximately 39 RM downstream of the site (water body segment No. WA-39-1030) is listed as impaired for rearing, harvesting, salmonid and other fish spawning, and migration. The source of this impairment is attributed to unspecified agriculture, irrigated crop production, unspecified hydro- or habitat-modification, and removal of riparian vegetation. Specific causes of the impairment are pesticides, priority organics, ammonia, pH, DO / organic enrichment, thermal modifications, and flow and habitat alterations (Ecology 1995).

2.4.6 Stormwater Management Manual for the Western Washington (Final Draft 2001 Ecology and the Puget Sound Water Quality Authority [Chapter 173-275-360 WAC])

Kittitas County has adopted the 1992 Ecology *Stormwater Management Manual for the Western Washington*, which lists the required stormwater management for drainage within the County (Ecology 1992). Trendwest will exceed that minimum requirement by complying with the standards detailed in the Final Draft 2001 Ecology Manual (Ridge Agreement, Section 1.9) (Ecology 2001c). The 2001c Ecology Manual specifies temporary erosion and sediment control (TESC) measures, storm drainage management (quantity and quality), and stormwater facility maintenance. The Ecology 2001c Manual has three water quality "Menus" describing treatment requirements for a variety of situations: the Basic, Enhanced, and Phosphorus Menus, each of which has options intended to provide equivalent removal of the pollution target(s) in Volume V. In addition, there is an Oil Control Menu for those projects deemed to require oil control.

2.4.7 Stormwater Runoff National Pollutant Discharge Elimination System (NPDES) Permit

For all new construction activity exceeding 5 acres in size, a Notice of Intent (NOI) must be filed for a National Pollutant Discharge Elimination System (NPDES) General Permit with Ecology for clearing, grading, and silt control. New legislation to be implemented by the EPA will require construction activities that disturb more than 1 acre to get a permit. Ecology expects to be issuing permits under the new rules by March 2003 (Austin, personal communication 2000). A public notice must be published at least once a week for 2 consecutive weeks in a newspaper that has general circulation in the county in which the site is located. The NOI must be received by Ecology prior to the publication of the public notices. Ecology will notify the applicant on coverage within 10 days of receiving a completed application; however, Ecology will not issue a permit until after the 30-day public comment period, which starts on the date of the last public notice publication.

Ecology may determine that an individual NPDES permit is required for the UGA site. An individual permit is written for a single facility and includes a description of the individual facility, its processes and discharge requirements in a fact sheet. This evaluation of the facility and legal requirements leads to a permit that specifies discharge treatment, monitoring, and reporting requirements tailored to an individual facility. Individual permits take a slightly longer time frame to issue than general permits because they assess the precise link between the discharge characteristics and permit requirements of the facility.

3.0 IMPACTS EVALUATION

3.1 Construction Impact Evaluation

Areas presenting risks of erosion and landslides exist within the UGA along the Cle Elum River, the West Ridge, Central Ridge, and East Ravine (Figures 1-2, 3-1, and 3-2). The area along the Cle Elum River contains low slopes ranging from 0 to 5 percent, but is determined to be at moderate to high risk for erosion because of the soils present and their location within the river floodplain (City of Cle Elum 2001). The Central Ridge contains soils with moderate erosion potential and slopes between 15 and 40 percent. The East Ravine is characterized by slopes between 15 and 40 percent and is considered to have a moderate to high risk erosion potential. The West Ridge contains slopes greater than 40 percent and is considered to have a high risk for both erosion and landslides.

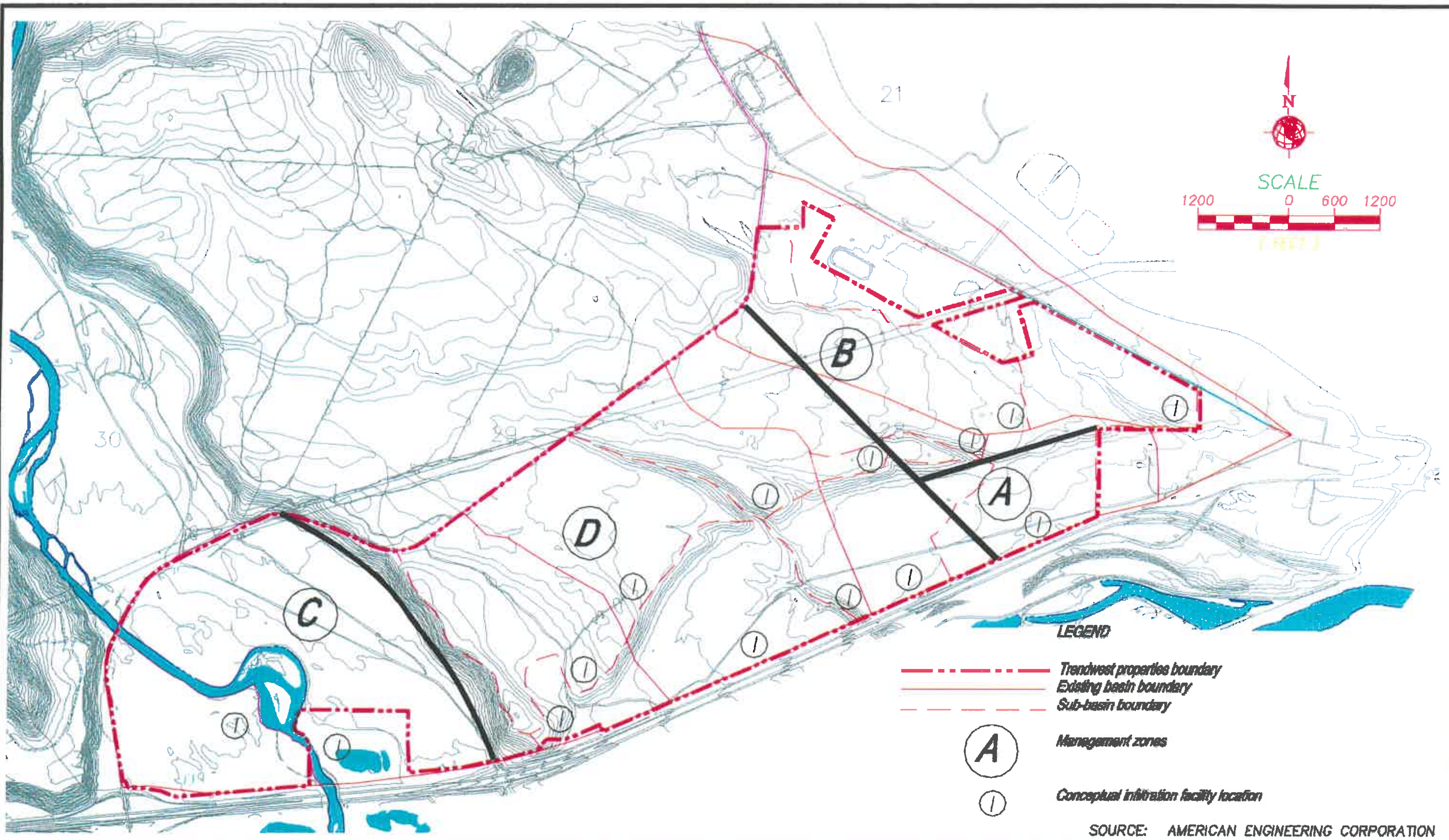
3.1.1 No Action Alternative

Since publication of the Draft EIS, approximately 23.0 acres within the Cle Elum UGA have been rezoned from Forest and Range 20 (20 acre lots) to Suburban 1 (one acre lots). The property is located north of Interstate 90 and east of Bullfrog Road, adjacent to the western boundary of Trendwest-owned property within the UGA. Kittitas County issued a SEPA Determination of Non-Significance (DNS) on the non-project action on October 10, 2001, with conditional language stating that any future development proposal would be subject to environmental review under SEPA at the time development was proposed. As such, potential impacts to water quality are addressed qualitatively in this EIS. Refer to Chapter 1, Summary of the Final EIS for additional description of the No Action Alternative.

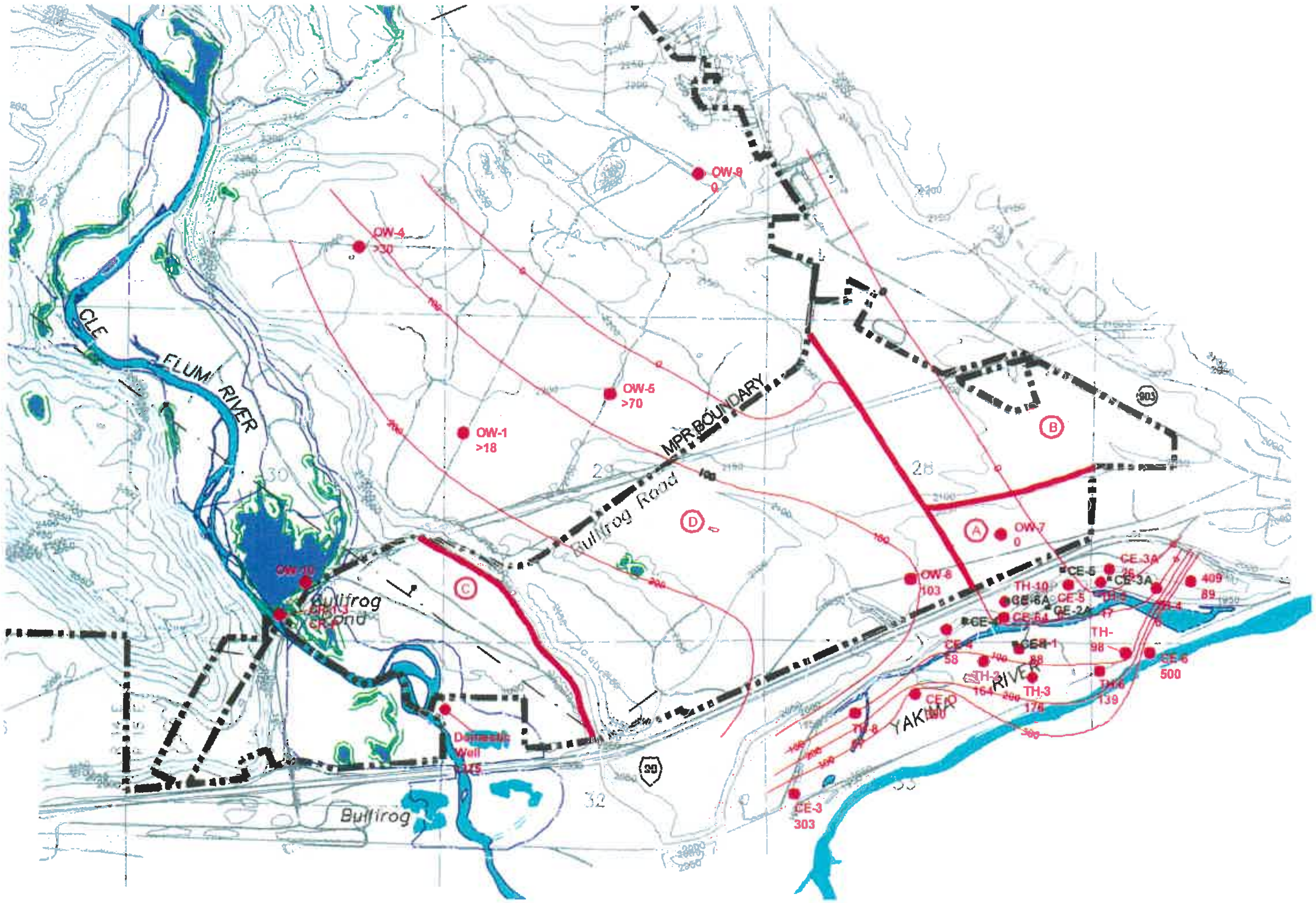
Under the No Action Alternative, the rezone of 23 acres to 1 acre lots would increase the potential for water quality impacts to receiving waters during construction. These impacts are addressed in the context of cumulative impacts, in Section 3.1.4 of this report.

3.1.2 Alternative 5 Construction Impacts

The preferred Alternative 5 lowers construction risk by avoiding development near the Cle Elum River in the highest risk portion of the UGA. This area, labeled Management Zone C, is described in the section entitled "*Stormwater Management Zones*" below. This would reasonably be expected to prevent sediment introduction to the river as a result of clearing and grading. Alternative 5 also has a lower cut and fill quantity than Alternative 2 previously described in the Draft EIS (City of Cle Elum 2001). Infrastructure and building construction would expose erodible soils and could increase the rate of surface water runoff from storms as a result of exposure and compaction. Although nearly all of the site infiltrates without generating surface runoff under normal conditions at present, localized erosion from surface water flows could occur if TESC measures were not implemented, or if they were implemented but failed. Uncontrolled sediment release to on-site wetlands could decrease water quality as well as fill localized portions of the wetlands if water is channelized and contains silts and sands. Risk of



A.C. Kindig & Co.	Water Quality Technical Report Cle Elum UGA FEIS	Figure 3-1 Slope Map



LEGEND

CE-1 ■ Cle Elum Hatchery production well

(A) Aquifer Susceptibility Zone

The following table lists the susceptibility (low, moderate, high) for transport of contaminants to the shallow and deep aquifer.

Aquifer Susceptibility Zones

Zone	Shallow Aquifer	Deep Aquifer
A	High	High
B	Moderate to High	Moderate to High
C	High	Low
D	Low to Moderate	Low

N

0 800 1600
FEET

REFERENCE: W&H PACIFIC.

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Figure 3-2

Management Zones, Aquifer Susceptibility these impacts would be increased during construction in the wet season when it was warm enough to rain (October through March) because of the increased difficulty in preventing erosion when saturated or frozen soils are exposed during warmer stormy weather; however, rare summer storms could also have the same result. Minor turbidity and minor sediment-related impacts to wetlands are generally not severe and long-lasting, since wetlands are naturally deposition environments. However, short-term water quality impairment and resulting habitat degradation could occur if inputs were sustained. Short-term water quality impacts to wetlands could include increases in turbidity, suspended and settleable solids, and phosphorus loading from eroded soils.

The use of heavy equipment during construction requires fueling and often limited storage of petroleum hydrocarbon products, which creates a risk of accidental spills of petroleum hydrocarbons. Unintended release of fuels, oil, or hydraulic fluid could contaminate soils and, if unintended or uncontrolled, migrate to groundwater or into surface water resources. Such water quality impacts, although locally severe, can typically be prevented with adequate construction site control measures and spill response planning required by the NPDES permit for construction discharge.

3.1.3 Alternative 5 Construction Mitigation

To avoid construction impacts, a Stormwater Pollution Prevention Plan would be prepared that employs applicable portions of Volume II of Ecology's (2001c) *Stormwater Management Manual for the Western Washington*. The manual includes Best Management Practices (BMPs) for control of sediments and construction waste during construction. Measures include required and recommended BMPs for control of sediment and for controlling pollutants other than sediment from construction sites. The proposed TESC measures were described in the UGA Draft EIS (City of Cle Elum 2001).

3.1.4 Cumulative Construction Impacts with the MPR and the Peare-O'Rourke Property

MPR and UGA cumulative impacts during construction were previously analyzed in the MPR DEIS and in the Draft EIS. Cumulative impacts on water quality as a result of construction within the UGA in conjunction with the MPR are not anticipated because nearly all water would be infiltrated, with very little potential for surface discharge reaching the Cle Elum River. The UGA would have little construction in the vicinity of the Cle Elum River under Alternative 2 previously described in the Draft EIS, or no construction in the vicinity of the river under Alternative 5. Most of the UGA property extends well to the northeast of the Cle Elum River, with little to no potential for construction impact to the river corridor. Cumulative impacts during construction would take two forms. With regard to individual wetlands or side-channels, there would be little to no potential for cumulative impacts because of the distance between the MPR and the UGA. Overland runoff would not extend from one site to the other. The Cle Elum corridor has been preserved as undeveloped open area for the MPR, which greatly reduced the risk of sediment delivery to the Cle Elum River (AESI 1999a). If any sediment was introduced to the Cle Elum River at the MPR site it would not reasonably be expected to have any impact on the quality of water downstream at the UGA site. With regard to the Cle Elum River and the Yakima River downstream, the amount of released sediments that could reasonably be expected would be unlikely to have a combined effect measurably different from either action alone, due to: (1) the high hydraulic energy flushing small particles through the river system and (2) high-flow rates in the Cle Elum River system during most of the active construction season rendering introduced fines to immeasurable levels over background (AESI 1999a). The risk of accidental spills during construction might increase on a per-incident basis. However, individual spills, once contained in accordance with TESC requirements, would have no potential to combine to create greater impact, even if they were to occur at the same time. Therefore, no adverse cumulative impacts are reasonably expected from the MPR and UGA properties during construction if TESC plans are written and implemented as required under the separate NPDES permits for these two projects.

The property zoned Suburban is located in the Cle Elum River corridor area of the UGA that is geologically and proximally determined to be at highest risk among all the areas within the MPR and UGA study areas (see Section 3.2.3). The risk for construction-related impacts on this property is higher because it is located where the Cle Elum River is in direct continuity with an underlying shallow alluvial aquifer. In general, if construction on the Suburban zoned property occurred concurrently with that of the proposed development under Alternative 5 and the MPR, potential cumulative construction-related water quality impacts to the Cle Elum River would increase above those described for the MPR and Alternative 5 of the UGA alone.

Clearing, grading, and construction of residential and supporting infrastructure (access roadway and utilities) could potentially deliver fine sediments, accidental spills of petroleum products, or construction waste such as concrete leachate to the river via the underlying alluvial aquifer. Short term impacts would likely be localized, but could include increased turbidity or direct localized toxicity to aquatic organisms from accidental spills or releases of fuels, hydraulic fluids, or concrete wash water or leachate. The proposed rezone would be responsible for acquiring the necessary NPDES permit(s) for construction, if such were required. Through February of 2003, construction of property under single ownership and less than 5 acres in size would not require an NPDES permit from Ecology. After March of 2003, that size exemption is expected to be reduced to less than 1 acre. If required, the NPDES permits would include provisions for stormwater pollution prevention plans. If NPDES permits are not required due to the size exemption, which could occur if individual lots were sold to builders prior to construction, then building and grading permits would be required from Kittitas County. Erosion control and spill prevention measures for construction for the Suburban 1 zoned property, the UGA, and the MPR, could and would be implemented independently; no regional construction measures or runoff control measures are anticipated practical or necessary for all three areas combined.

3.1.5 Unavoidable Adverse Impacts

Isolated and localized releases of turbid water could occur to on-site wetlands or to the Cle Elum River under Alternative 5 if the TESC measures failed or were inadequately maintained.

Impacts to water quality or wetlands, if any, would be expected to be short-term with no broad or cumulative effects. Isolated spills of petroleum products could occur from construction equipment. These spills would be toxic if released to surface water and could contaminate shallow groundwater on a localized basis if uncontrolled. Accidental spill plans could adequately provide for containment and cleanup within constructed areas of the preferred Alternative 5, which are outside of the alluvial floodplain. Thus, no large-scale or long-term impacts are reasonably expected to occur away from the floodplain. No construction would occur within the floodplain.

3.2 Developed Water Quality Impact Evaluation

3.2.1 No Action Alternative

Direct operation impacts under the No Action Alternative were discussed in Section 3.3 of the Draft EIS. As described above under Construction Impacts, 23 acres of the UGA have been rezoned from Forest and Range 20 to Suburban 1, since publication of the Draft EIS. Potential operational impacts from development of this property, in combination with impacts from development of the Trendwest MPR and UGA properties, is provided in Section 3.3.4 of this report.

3.2.2 Alternative 5: Stormwater Contaminants

Metals

Roadways are a source of inorganic and organic materials that run off during storms. Roadways can contribute an array of pollutants to surface and groundwater resources (Maestri et al. 1988). Roadway runoff contains heavy metals, rubber, polycyclic aromatic hydrocarbons (PAHs), petroleum products, and solid materials. Stormwater washes these contaminants from roadways and deposits them downslope. The concentrations of these pollutants are highly variable by site, and are affected by numerous factors such as traffic characteristics, climate, maintenance, and adjacent land use (Maestri et al. 1988).

Lead was historically present in roadway runoff at higher concentrations than any other priority pollutant (Municipality of Metropolitan Seattle ([Metro] 1982), although it has declined considerably in the environment since its removal as a gasoline fuel additive. More recent data for developed runoff has shown a dramatic decline in metals and other automotive pollutants due to automotive emission controls and catalytic converters (CH2M Hill 1992). Lead in stormwater runoff originating from streets is associated with solid material; dissolved lead in storm runoff comprised less than 10 percent of the total lead found in runoff (Metro 1982).

The primary source of roadway copper is wear from vehicle parts, such as brakes, alternators, and radiators. Low concentrations of the cupric ion of copper are extremely toxic to phytoplankton (Metro 1982).

Tire rubber is the source of cadmium on roadways. Cadmium is less affected by sorption than other trace metals (Metro 1982). Salmonids are extremely sensitive to waterborne cadmium ions (Metro 1982).

Zinc is an abundant trace mineral that occurs naturally in water bodies. Studies of lakes adjacent to roadways show increases in zinc concentrations in lake sediments (Gjessing et al. 1984). Zinc is not considered a carcinogenic metal and federal agencies have no specified health limits for zinc. However, Washington State water quality standards for zinc do exist.

Street dust collects fuel combustion byproducts. Tire and mechanical wear are also concentrated in street dust and urban soils. Metro (1982) found all priority pollutant metals except selenium in street dust samples. Also, Metro (1982) determined that the six metals found in highest concentration in street dust also appeared in highest concentration in stormwater runoff from the same areas.

Oil and Grease, and Total Petroleum Hydrocarbons (TPH)

Oil and grease and total petroleum hydrocarbons (TPH) result from spills, leaks, antifreeze, hydraulic fluids, and asphalt leachate (Washington State Department of Transportation [WSDOT] 1997). Oil and grease have poor solubility in water and are hydrophobic, which means they readily separate from the aqueous phase and adhere to solid surfaces. Appreciable amounts of oil and grease can remain dispersed in water in emulsified form. The hydrophobic nature of these compounds is the basis for BMPs to remove them from stormwater. Supplying a large surface contact area (such as grasses in biofiltration or emergent vegetation in shallow ponds) allows the oil and grease and TPH to adhere to emergent surfaces, where they are degraded by microbial digestion, sunlight (photochemical degradation), and volatilization. TPH is a subset of oil and grease derived solely from petroleum products. TPH is more volatile than oil and grease; therefore it has at least equivalent and usually greater removal rates than oil and grease in stormwater facilities.

Suspended Solids

Sediment is comprised of inorganic and organic material and can be transported by, suspended in, or deposited by stormwater. Suspended sediment is generally considered to be one of the most substantial non-point source contaminants (Waters 1995, Crawford and Mansue 1996). Many contaminants including some metal ions, organic chemicals, and phosphorus bind to and are transported by fine sediments (Waters 1995, Simmons 1993).

Nutrients

Nitrogen and phosphorus occur in stormwater runoff from roadways, from fertilizers used in residential lawns and landscaping, and from sediment erosion. Phosphorus also originates from detergents used to wash cars or home exteriors. Phosphorus readily binds to sediments.

Consequently, if particulate settling is provided, much of the phosphorus can be settled out of the water bound to finer sediments.

Fecal Coliforms, Biochemical Oxygen Demand (BOD), and Temperature

Increased fecal coliforms are a generalized result of development related to residential and land use density. The construction of impervious surfaces collects organics from pet and wildlife waste that are then available to be washed off during storms. Fecal bacteria densities have been shown to be related to housing density, percent impervious surface, and domestic animal density (Young and Thackston 1999). Generally, stormwater runoff from residential and urban development carries a low biochemical oxygen demand (BOD) concentration, unlike runoff from agricultural areas with significant livestock use. Urban runoff can influence stream temperatures during summer storms because of the warmer impervious surfaces, reduced tree cover, and use of ponds to remove contaminants and/or detain flows. However, most storm runoff events and the vast majority of runoff volume occur during the cooler weather seasons, and stormwater on the UGA would be infiltrated, eliminating concern for temperature impacts.

3.2.3 Stormwater Management for Alternative 5

Stormwater Facilities Changes

Stormwater facilities for the UGA were previously described in Section 3.3 of the Draft EIS and in the MDP (AEC 1999). Since the Draft EIS was published, the proposal for water quality treatment of stormwater runoff has been modified, in most cases increasing treatment prior to infiltration. For this purpose, and publication of the Draft EIS, the UGA site was subdivided into stormwater management zones based on underlying geology and groundwater flow paths. Treatment is now proposed in accordance with the water quality risk in each zone (flow route, infiltration soil type, groundwater receptors, and receptor proximity). The project has also incorporated provisions of the 2001 Department of Ecology Stormwater Management Manual (Ecology 2001c) produced since Draft EIS publication, which has altered treatment requirements. The facilities proposed at the time of the Draft EIS are summarized below, along with the basis for the change in treatments proposed in the Final EIS (Table 3-1).

Stormwater Management Zones

The proposed UGA site is divided into four water quality management zones named A, B, C, and D, largely as a result of underlying geology and the groundwater flow patterns (AESI 2001) (Figure 3-2). The Cle Elum River lies to the west of the UGA, and is in direct continuity with a shallow aquifer in alluvial soils immediately adjacent to the river. The portion of the site overlying these alluvial deposits is Management Zone C. Underneath the shallow alluvial aquifer, the Cle Elum River, and the main central portion of the UGA site, there is an aquitard approximately 200 feet or more in depth that is formed of silt and clay lacustrine deposits. This aquitard protects the underlying deep aquifer from land surface influences. The main central portion of the UGA site is Management Zone D, which has outwash soils at the surface. Till soils occur over the outwash on the highest elevations of Management Zone D to the north, nearest Bullfrog Road. The outwash supports a shallow aquifer, which again is separated from the deep aquifer by the same very deep lacustrine aquitard as occurs under Management Zone C.

Table 3-1
Changes to the Draft EIS Stormwater Treatment Proposal for the Cle Elum UGA

<table>
<tr><th>DEIS and MDP Treatment Category</th><th>Land Uses Served</th><th>Treatment Proposed in DEIS</th><th>Basis for Treatment Changes Incorporated in this FEIS Supplemental Report</th></tr>
<tr><td>Level I</td><td>Roof, Golf</td><td>Natural Infiltration (No golf is proposed under Alternative 5)</td><td rowspan="4">1. “Basic Treatment Menu” options from the 2001 Department of Ecology Stormwater Management Manual (for all land uses) in the lower risk Management Zones (Zones C and D as described below). These facility options are intended for projects that discharge to the ground. (Note: no development is proposed in Zone C under Alternative 5.)

2. “Enhanced Treatment Menu” options from the 2001 Department of Ecology Stormwater Management Manual (for all land uses) in the higher risk Management Zones (Zones A and B as described below). These facility options are intended for projects that discharge to waters tributary to fish-bearing streams. Although Management Zones A and B would discharge to the ground and the groundwater path is proximate to the drawdown Yakama Hatchery wells. Thus, these areas are proposed for a higher level of treatment where development is proposed.</td></tr>
<tr><td>Level II</td><td>Residential Driveways</td><td>Filter Strip Treatment and Natural Infiltration</td></tr>
<tr><td>Level III</td><td>Roadways</td><td>Filter Strip Treatment and Natural Infiltration</td></tr>
<tr><td>Level IV</td><td>Parking Lots, High Density</td><td>Bioswale Treatment and Natural Infiltration</td></tr>
</table>

Further east, under Management Zones A and B, the surface soils are the same as for Management Zone D. However, Zones A and B are distinguished from D because the thick lacustrine aquitard is absent. Thus, the deep aquifer is more vulnerable to surface activity. Zone A is more proximate to the Yakima River, and thus the Yakama Hatchery intake wells, than Zone B, which is why they were separately distinguished.

In summary:

- Management Zone A is characterized as having a high risk for transport of contaminants to the underlying deep aquifer and thus to the Yakama Hatchery wells, due to proximity and the lack of an intervening aquitard;

- Management Zone B is characterized as having a high to moderate risk for transport of contaminants to the underlying deep aquifer; the geology is the same as Management Zone A, but B is more distant from the Yakama Hatchery wells;

- Management Zone D is characterized as having a low to moderate risk to the shallow aquifer due to soils and a very low risk to the deeper aquifer due to an intervening aquitard; and

- Management Zone C is characterized as having a high risk to the shallow alluvial aquifer and the Cle Elum River in direct continuity, but a very low risk to the deeper aquifer.

There is approximately 1.4 acres of an existing gravel roadway in Management Zone C, but no development is proposed by Trendwest in Management Zone C under Alternative 5. Zone C is not discussed further in this section, because the preferred Alternative 5 would have no direct influence on groundwater quality from this zone.

Stormwater Facilities Proposed

Treatment of Management Zone D runoff would meet the Ecology Manual requirements for a site proposing to infiltrate stormwater more than _ mile from a fish-bearing stream, a tributary to a fish-bearing stream, or a lake (Ecology 2001c). This basic level of treatment would be provided prior to infiltration in Zone D, which can be satisfied by biofiltration swales, filter strips, basic wet ponds, wet vaults, stormwater wetlands, combined detention and wet ponds, or sand filters. Stormwater wetlands are recommended, and for the purpose of this analysis are assumed to be the treatment facility type employed. Roof runoff is not considered to be pollution generating by the Ecology manual and could be infiltrated directly.

Zones A and B have less natural filtration afforded by the underlying sediments. Runoff from Zones A and B under the preferred Alternative 5 would have enhanced treatment to further reduce dissolved metals and other contaminants prior to infiltration. This proposed level of treatment would exceed the Ecology Manual requirements for water quality treatment (Ecology 2001c). The "Enhanced Treatment Menu" (Volume V) in the Ecology Manual incorporates two-facility treatment "trains" for enhanced dissolved metals removal. The "Enhanced Treatment Menu" provides a variety of facilities to satisfy this objective. The treatment facility could be any of the following:

- A biofiltration swale followed by a sand filter or media filter;
- A filter strip followed by a sand filter (or in reversed order);
- A basic wet pond followed by a sand filter or media filter;
- A wet vault followed by a sand filter or media filter;
- A basic combined detention and wet pond followed by a sand filter or a media filter; or a
- A presettling or detention basin, sand filter, and media filter in that sequence.

A treatment train consisting of a wet pond followed by a basic sand filter is recommended and for the purpose of this analysis was evaluated for all runoff from Management Zones A and B.

Non-Point Source Controls Proposed

The following non-point source are proposed under Alternative 5:

- Residential Landscaping: Residential owners would be provided with educational literature on means to reduce or eliminate landscape chemical use and encourage use of adapted landscaping plants.

- Non-Residential Landscaping: Common landscaping emphasizing native or adapted vegetation would be used in public areas to the extent feasible, and encouraged in private areas (for example business parks, multi-family landscaping).

- Residential and Commercial Structures: External unsealed copper ornamentation and the use of exposed galvanized metals would be prohibited (except where the latter may be required by City codes for construction of utilities).

Enhanced Treatment Facilities

Wet ponds followed by sand filters were analyzed for the enhanced treatment proposed for all runoff infiltrated in Management Zones A and B. Each portion of this dual facility is described separately below, followed by expectations for its combined function.

Wet Ponds

Wet ponds (or combined wet detention ponds) maintain a dead storage volume of water that removes dissolved phosphorus and nitrogen by settling of fine particles, nutrient uptake by algae and fringing vegetation, denitrification, microbial degradation of organics, and sequestering of phosphorus and metals in the sediments (Nussbaum 1990). The dead storage volume also dilutes the first portion of stormwater inflow with the higher quality residual water since the previous storm. Quiescent, or between-storm settling of particulates and attached contaminants, is very effective if the time between storms is a period of days. However, most runoff volume is treated under dynamic conditions during storms while pond inflow and outflow is occurring. Overall performance in removing particulates and the dissolved constituents that they attract and bind is a function of hydraulic residence time and dissipation of hydraulic energy, which relates to the amount of time slowly settling small particles will have to settle below the dead water storage at the pond's bottom. Time between storms is the largest factor in residence time, given the fixed wet pond sizing criteria in the Ecology Manual. The size, shape, and dead water depth in wet ponds are designed to dissipate hydraulic energy of water passing through the pond to promote settling of particulates, as well as prevent re-suspension of settled material. Efficiency for fine particle removal measured as the percentage of outflow over inflow concentrations is extremely variable from start to finish of a given storm and from storm to storm depending upon the runoff hydrographs, and upon the inflow concentrations of stormwater constituents. Thus, stormwater facility functions are presented in the literature as averaged percent removal efficiencies for individual contaminants (Table 3-2).

Table 3-2
Wet Pond Contaminant Removal Efficiencies (%) Reported in the Literature

Reference	TSS	TP	NH3-N	NO3-N	Pb	Zn	Cu	FC	Oil and Grease
U.S. EPA 1993	80	65			75	60			
City of Austin 1990	46	37		36	40-60	40-60	40-60	20-40	
Martin 1989a	66	50	55		42	50			
King County 1995	50-90	50	72	67	25-80	30			
Urbonas 1994	90	40			10-95	0-70			
Schueler et al. 1991	50-90	50			40-60	40-60	40-60		
Brown and Schueler 1997	77	47		24	73	51	57	65	83
Shapiro 1998 (wet vault) (1)	50	20	14	-11	41	48	39		
Wanielista 1988					37	50	49		
Yousef and Wanielista 1985						47-97	47-97		
Average	68	48	64	42	52	50	52	47	83
Wet Pond Standard Values Used in this Analysis	80	45	65	40	50	50	50	45	75

(1) Wet vault performance for Lakemont facilities in Bellevue, Washington; water passing through the vault only. A wet vault is a subterranean chamber sized to contain a typical wet pond's volume, but lacks sunlight. Therefore, vaults lack biological uptake performance capability, which reduces nutrient removal performance, especially nitrogen. For that reason, nutrient removal rates for the wet vault were removed from averaging.

Sand Filters

Water quality enhancement in sand filters accrues from filtration, which removes particulates and associated contaminants; adherence (known as sorption) of contaminants to sand particles within the filter; and mineralization, the binding and immobilization of phosphorus to aluminum or iron in sand (Table 3-3). Turf-capped sand filters reduce clogging because the grass roots facilitate water flow in the sand matrix while the turf traps fines. This occurs even where the grass is dormant. Turf covers on sand filters also simplify and improve by improving the accuracy of the visual inspections. Lab tests evaluated the clogging potential of sand filters by applying a one time exposure of 2,000 to 3,500 NTU (AESI unpublished data [1998]). Turbidities this high are essentially silt slurry in excess of any possible developed runoff condition. All of the sand filters tested were clogged to 10 percent of their initial hydraulic conductivity by this treatment. Turf-covered sand filters recovered to 50 percent within a month, whereas the no-turf control remained at about 10 to 25 percent of its original hydraulic conductivity.

Table 3-3
Standard Sand Filter Removal Efficiencies (%) Reported in the Literature

Reference	TSS	TP	NH3-N	Pb	Zn	Cu	Coliform Bacteria
City of Austin (1990)	87	60	76	80	80	59	36
King County (1998)	80	<50					
Shapiro (1998) (1)	93	68	47	61	91	53	
Urbonas (1999) (2)	80-94	50-75			80-90	20-40	
AESI (unpublished 1998) (3)	75	35			77		
AESI (1999b) (4)	80	65	45	60	80	50	35
Values Used in this Analysis	85	65	45	60	80	50	35

(1) Data for water passing through the sand filter after wet vault treatment, data from 1996-1997.
(2) Most common data range for field measured performance ranges.
(3) Data from Trossachs plot Tract R sand filter with grass turf cover, January 1998. Outlet TP concentrations were very low, ranging from 0.008 mg/L to 0.030 mg/L.
(4) Data from Trossachs plat Tract R sand filter with grass turf cover, performance monitoring from January 15 through February 25, 1999. Mean TP inflow was 0.064 mg/L and mean (AESI 1998).

Combined Wet Pond and Sand Filter Performance

To evaluate the combined facilities, performance calculated from independent studies on sand filters and wet ponds was compared to field measures from a combined wet vault and sand filter in Lakemont, Washington. This system is a regional facility that serves a mixed use suburban development in the Puget Sound lowlands, and has 3 years of flow-proportionate field data.

For the calculated efficiency, contaminant removal efficiency of the second facility (the sand filter) was reduced by 25 percent to forecast the combined efficiency of the two-facility system. These values are shown in the first column in Table 3-4. This reduction is meant to account for the drop in removal efficiency that typically occurs in the second facility, because inflow concentrations are lower as a result of treatment in the first facility. For the wet pond plus sand filter combination proposed in Management Zones A and B, the combined efficiency calculations for wet ponds and sand filter performance in tandem were compared to 3 years of measured data from the Lakemont, Washington wet vault plus sand filter system (Shapiro 1998). The calculation method for combined facilities was reasonably accurate for most parameters, but underestimated heavy metals removal by approximately the 25 percent reduction factor applied to the sand filter efficiency (Table 3-4). Without that factor included, the calculated value was very close to the performance measured for the combined Lakemont facility. Nitrate-nitrogen and ammonia-nitrogen removals for the wet vault are lower in the Lakemont data because wet vaults do not allow for sunlight, and therefore afford little opportunity for biological uptake as compared to a wet pond (note: phosphorus is removed via sorption to and settling of particles even in a wet vault). The Lakemont data were not used to estimate nitrogen removal, but were weighted heavily for the remaining parameters since they provided field flow-proportionate data collected over 3 years. The efficiency values in this analysis are shown in Table 3-4.

Table 3-4
Comparison of Calculated Wet Pond Plus Sand Filter Combined Efficiencies

Parameter	Calculated Efficiency (%) (1)	Lakemont (wet vault) Combined Efficiency (%) (2)	Efficiency used in this Analysis for Combined Facilities (%)
Total copper	60	71	70
Total zinc	78	95	95
Total lead	67	77	75
Ammonia-nitrogen	67	54	67
Nitrate-nitrogen	52	6	52
Total phosphorus	69	75	75
Turbidity (3)	85	Not measured	85
Total suspended solids	92	96	95
Fecal coliforms	45	Not measured	45
Oil and grease (4)	75	Not measured	75

(1) Standard wet pond plus sand filter series, the latter discounted 25 percent. Facility efficiencies for wet pond and sand filter from Tables 3-1 and 3-2.
(2) Data for combined wet vault and sand filter at Lakemont, for the treated volume only (Shapiro 1998).
(3) Turbidity estimated at slightly less than TSS removal.
(4) Includes oil/water separator function in addition to wet pond and sand filter.

Stormwater Wetlands (Management Zone D)

Water purification in wetlands occurs through activity in their four principal components: vegetation; water column; substrates; and microbial populations (Hammer 1996). Stormwater wetlands have potential to function better than wet ponds, because pollutant removal is enhanced by a much greater level of microbial uptake and degradation as a result of the association between microbial growth and aquatic vegetation; by pollutant removal through binding to humic and organic acids; and by the sediment-water interface that these systems promote. The large areas providing contaminant-binding sites on sediments and plant structures, and the oxygenated/low oxygenated interface typically present at shallow depths in wetland soils, contribute to the biochemical transformation and immobilization of heavy metals, removal of nitrogen, and the decomposition of organic compounds (James 1995, Reuter et al. 1992, Hammer 1996). The wetland plants themselves also play a role in pollutant removal, although that function is highly variable between species. Roots and rhizomes of aquatic plants have been shown to accumulate metals to a greater degree than in the leafy structures, and plants such as rushes show a particularly good ability to take up organic contaminants, such as oils (Kulzer 1990). Most importantly, decomposing biomass provides a readily available source of organic carbon with a large surface area that sustains microbial populations. Microbial bacteria degrade organics and consume nutrients. Stormwater wetland performance expectations for the proposed system are summarized in Table 3-5.

The stormwater wetlands would be constructed per requirements of BMP T10.30 in the 2001 Ecology Manual (Ecology 2001c).

Table 3-5
Stormwater Wetland Removal Efficiencies (%) Reported in the Literature

Reference	TSS	TP	NH3-N	NO3-N	Pb	Zn	Cu	FC	Oil and Grease
Reed et al. 1981 Hickok et al. 1980 Lynard et al. 1980 Chan et al. 1982 ABAG et al. 1986 [(1)]	64	99			31-96	31-96	31-96		
Martin 1989b [(2)]	66	19	54-60	13-30	54*	75*			
Martin 1989b (pond and wetland)	88	41	0-61	30-43	70*	65*			
Field et al., 1993 [(3)]	54-84	25-40		-27-20	-16-50	47-55			
Schueler 1994 [(4)]	78	51		67	63	54	39	77	90
Reinelt and Horner, 1995 [(5)]	13.6	8				31		49	
Reinelt and Horner 1995	57	82				23		29	
Scholes et al. 1999 [(6)]	24				62	55			
Average Removal Efficiency	53	48	45	30	54	52	40	52	90

* Dissolved metals
(1) Literature review of stormwater treatment in artificial wetlands (Stockdale 1991).
(2) Detention pond and wetland system in Orlando, Florida. The system receives runoff from a 41.6-acre basin. The surface area of the permanent pool in the detention pond is 8,600 ft^2 and the wetlands cover an area of about 32,000 ft^2 and range in depth from 0 to 5 feet.
(3) A vegetative buffer strip with a length between 21 and 45 meters. The grass type is Kentucky 31 covering an area of 3,770 m^2 as part of a level spreader and buffer strip system.
(4) Median pollutant removal efficiencies for wetlands (Center for Watershed Protection 1999).
(5) Two freshwater wetlands Bellevue 3I (B3I) and Patterson Creek 12 (PC12) located in Bellevue, Washington. Wetland B3I is 2 ha in size and a drainage area of 187 ha. The storage volume is 400 to 5,000 m^3 and a residence time of 3.3 hours. Wetland PC12 is 1.5 ha in size and has a drainage area of 87 ha. The storage volume is 600 to 7,000 m^3 and has an average residence time of 20 hours.
(6) Brentwood wetland is a constructed wetland with a surface area of 360 m^2 and was sampled over 2 years. The average residence time for water in the wetland is 50 minutes.

Stormwater Facility Maintenance

To be effective, water quality facilities must be maintained. The 2001 Ecology Manual (Ecology 2001c) lists maintenance requirements for each BMP facility type in Volume V. Wet pond maintenance is described on page 10-12 (BMP T10.10). Pond maintenance measures include inspection and control of erosion on side slopes, removal of debris, maintenance of the spillway, and cleanout of sediments in the forebay.

Sand filter maintenance criteria is on page 8-22 in Volume V of the 2001 Ecology Manual (Ecology 2001c), and these measures would be employed on the Cle Elum UGA. Turf-covered sand filters maintain approximately 25 percent of their initial hydraulic conductivity even under adversely silty conditions, the 25 percent capacity equals 4 inches per hour, or four times the approximate rate at which sand filter design standards assume maintenance is required. Maintenance intervals for sand filters can be extended for many years with turf caps, and the maintenance can be conducted as needed. Inspection at any time of the year will show where the grass will not grow because of clogging. If the inspection warrants, the affected grass area would be replaced, instead of the wholesale sand bed replacement that would be required with a sand filter lacking a turf cover. Thus, capping sand filters with turf is highly recommended for maintenance purposes.

Stormwater wetland maintenance requirements are the same as for conventional wet ponds in the 2001 Ecology Manual (Ecology 2001c).

3.2.4 Alternative 5 Stormwater Quality Analysis Methods

Stormwater quality was forecast for each of the management zones (A, B and D) that would contain development under preferred Alternative 5. The catchments making up each water quality management zone are described in Table 3-6 (Figure 3-3).

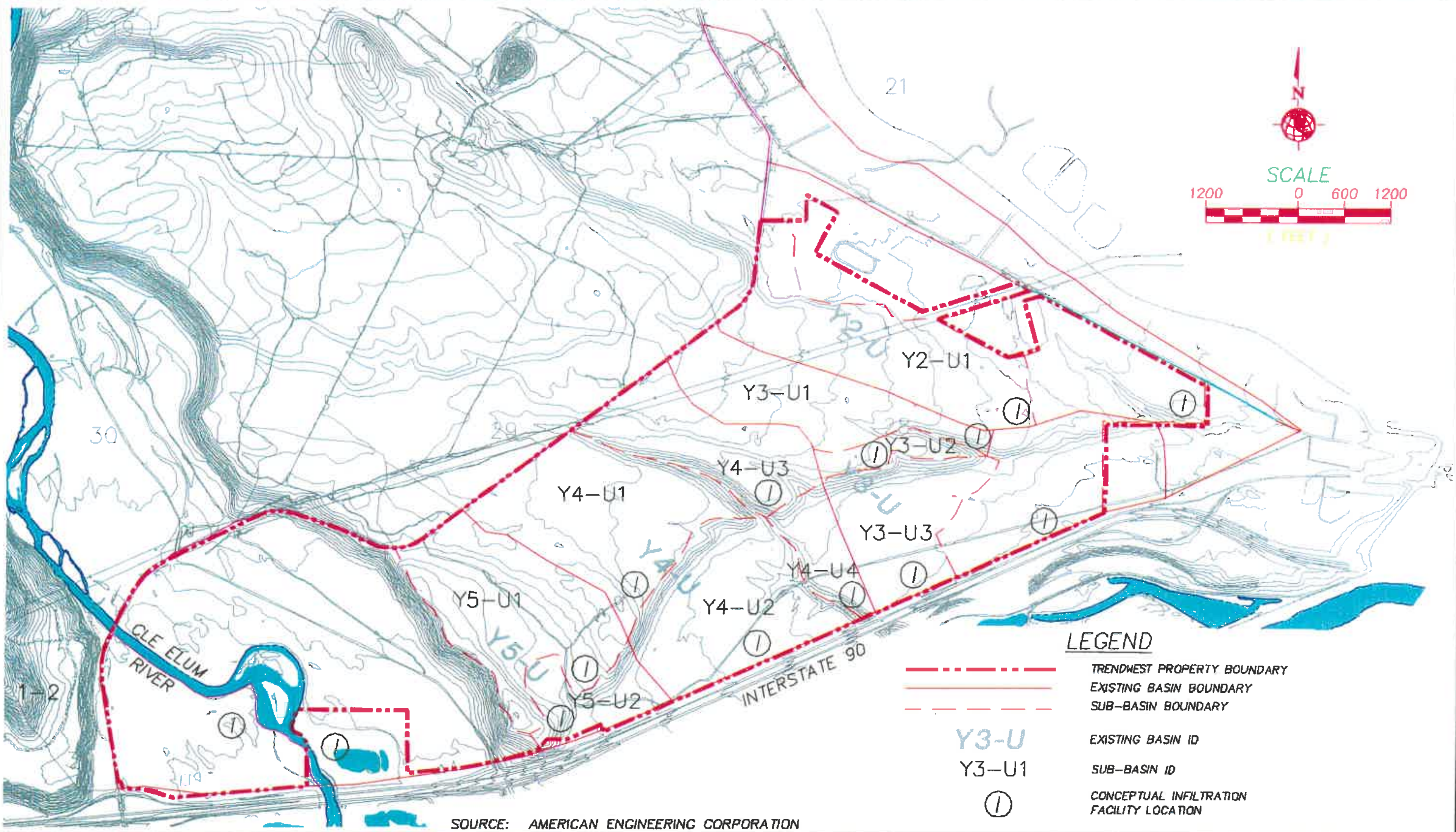
Table 3-6
Catchments Breakdown within the Four Water Quality Management Zones

Water Quality Management Zone	Catchments
A	Y3-U3
B	Y2-U1, Y2-U2, Y3-U1, and Y3-U2
D	Y4-U1, Y4-U2, Y4-U3, Y4-U4, Y5-U1, and Y5-U2

Untreated stormwater quality was forecast by dividing the site into land use categories, and using literature-based data for each category (Table 3-7).

Table 3-7
Untreated Water Quality References for Each Land Use Category

Land Use Category	References
Single-Family Residential *Alternative 5 proposes densities ranging from 2 to 7 DU/acre*	Single-family untreated runoff quality was forecast using residential runoff literature data from housing densities in the approximate range of 4 to 7 DU/acre, including associated roadways and plat infrastructure: AESI 1999b, King County 1995 and 1997, Shapiro and Associates 1998
Multi-Family Residential *Alternative 5 proposes densities ranging from 8 to 15 DU/acre*	Multi-family untreated runoff quality was forecast using runoff literature data for densities ranging from 10 to 14 DU/ac: City of Austin 1990, AESI 1999b, and City of Bellevue 1995
Commercial	Commercial land use untreated runoff quality was forecast using commercial and office park runoff literature data: Booth et al. 1997, City of Bellevue 1995, King County 1993, and Copp et al. 1997



A.C. Kindig & Co.	Water Quality Technical Report Cle Elum UGA FEIS	Figure 3-3 Developed Condition Basin Boundaries

The Cle Elum UGA single-family residential land use under Alternative 5 would range between approximately 2 to 7 dwelling units (DU)/acre of the developed footprint, including the supporting roadways and infrastructure. Residential untreated runoff quality was forecast using residential runoff data from housing densities in the approximate range from 4 to 7 DU/acre, including their associated roadway and parking areas. The higher density multi-family or condominium land use would range from 8 to 15 DU/acre. Therefore, the untreated 10 to 14 DU/acre land use category was conservatively applied to all condominium runoff entering the storm drainage system. Because the traffic volumes expected for the project would be comparable to roadways supporting 2 to 7 DU/acre single-family housing or 8 to 15 DU/acre multi-family housing, this same runoff category was applied to project roadways. Parking lots for parks, commercial impervious, and other (minor miscellaneous) impervious areas were all conservatively assumed to have use similar to parking for commercial or retail use.

Runoff from each land use category was proportionately mixed in each management zone on the basis of impervious surface (Table 3-8). Runoff from the approximately 1.1 acres of arterial roadway accessing the UGA from Ranger Station Road and passing through Management Zone A was assigned the same untreated water quality as the composition of land uses in Management Zone B, which this roadway would serve. Untreated water quality was improved by transit through either a stormwater wetland (Zone D) or wet pond and sand filter facility (enhanced treatment, in Zone B). The untreated water quality estimated for Alternative 5 is presented in Table 3-9. Using removal efficiencies shown in Tables 3-2, 3-4, and 3-5, the resulting discharge quality was then compared to background groundwater concentrations, water quality standards, and fisheries sublethal limits. Non-point influences on water quality that are not collected by the storm drainage system, were analyzed separately (Section 3.2.6).

Table 3-8
Total Impervious Acreage per Management Zone, Alternative 5

Land Use by Management Zone	Impervious Acreage Without Roofs	Roofs	Undeveloped [(1)] Land
A	**1.10**	**0.00**	**63.9**
Access Roadway	1.10		
B	**66.7**	**31.7**	**63.7**
Single-Family Residential	5.80		
Single-Family Residential Roofs		3.80	
Multi-Family Residential	2.60		
Multi-Family Residential Roofs		4.90	
Commercial	56.8		
Commercial Roofs		23.0	
Open Space Parks	1.50		
C	**1.40**	**0.00**	**244.6**
Existing Gravel Road	1.40	0	

(1) School expansion, cemetery expansion, buffer areas, and other undeveloped open space.

Table 3-8
Continued

Land Use by Management Zone	Impervious Acreage Without Roofs	Roofs	Undeveloped [(1)] Land
D	**75.2**	**59.9**	**285.7**
Single-Family Residential	61.1		
Single-Family Residential Roofs		38.9	
Multi-Family Residential	9.30		
Multi-Family Residential Roofs		20.1	
Commercial	4.80		
Commercial Roofs		0.90	

(1) School expansion, cemetery expansion, buffer areas, and other undeveloped open space.

Table 3-9
Alternative 5 Untreated Stormwater Runoff Quality (Excludes Infiltrated Roof Runoff)

Parameter	Untreated Runoff Concentrations from Management Zones A and B	Untreated Runoff Concentrations from Management Zone D
Copper (µg/L)		
Total	10	8.0
Dissolved	4.0	3.2
Lead (µg/L)		
Total	9.0	4.0
Dissolved	4.8	2.1
Zinc (µg/L)		
Total	127	39
Dissolved	51	15
Nitrate+Nitrite-Nitrogen (mg/L)	0.45	0.40
Ammonia-Nitrogen		
Total (µg/L)	130	140
Unionized (µg/L)	0.30	0.30
Total Phosphorus (mg/L)	0.09	0.14
TSS (mg/L)	16	11
Fecal Coliforms (colonies/100 mL)	695	643
Oil and Grease (mg/L)	2.9	2.1

3.2.5 Alternative 5 Stormwater Evaluation Criteria

Three criteria were used to evaluate discharge from the stormwater treatment facilities. These were (1) the sublethal criteria for salmonids derived from the literature, (2) state water quality standards, and (3) the background groundwater and surface water conditions.

Sublethal effects include water chemistry-induced changes in physiology and/or behavior that affect the competitive vitality or reproductive potential of a fish population without direct lethal effect. State water quality standards include a factor of safety to avoid lethal effects, but are not necessarily tied to sublethal effects data. The treated discharge in this updated analysis is compared to state standards, background conditions, and sublethal effects information (Section 3.2.4).

Fish sensitivity to water quality changes was assessed by evaluating literature data for behavioral and physiological fish responses to sublethal concentrations of typical storm runoff contaminants. The following information summarizes the literature review on water quality and fish health. For each parameter, the more restrictive of either the literature review or the state water quality standards was used as the "desirable limit" (Table 3-10).

Table 3-10
Yakima River Background, Surface Water Quality Standards, and Guidelines for Sublethal Fish Effect Avoidance

Parameter	Existing Concentration in the Yakima River [1]	Acute and Chronic State Water Quality Standards Class A	Literature-Based Fish Sublethal Limits
Temperature (°C)	6.8	Less than 18	--
Dissolved Copper (µg/L)	0.19	3.02/2.37	2.4 [2]
Dissolved Lead (µg/L)	0.01	8.38/0.33	11 to 16 [3]
Dissolved Zinc (µg/L)	0.75	24.2/22.1	12 [3]
Ammonia-Nitrogen, Total (µg/L) Unionized (µg/L)	 10 0.05	 24,000 [6] 66 [6]	 7,300 [6] 20 [5]
Nitrate+Nitrite-Nitrogen (mg/L)	0.04	None	Less than 250 [7]
Total Phosphorus (mg/L)	0.07	None	No direct adverse effects
pH	7.2	6.5-8.5	6.0-9.0 [4,5]
Dissolved Oxygen (mg/L)	11	Greater than 8.0 mg/L	--
Oxygen Saturation (percent)	96	Less than 110	75% saturation at 5°C (41°F); 90% saturation at 25°C (77°F) [5]
Hardness (mg/L as $CaCo_3$)	27	None	At least 20 [4]
Turbidity (NTU)	3.4	Less than 5 over background	--
Total Suspended Solids (mg/L)	5.6	None	Less than 80 [4]
Fecal Coliforms (CFU/100 mL)	11	100 [8]	--

Note: Water quality forecasts and baseline conditions were reported in total metals and converted to dissolved metals to compare to state standards. To obtain dissolved concentrations, approximately 40 percent of total copper and total zinc and 53 percent of total lead were assumed to be in dissolved form (King County 1995).

(1) Yakima River water quality data collected at Ecology's ambient monitoring station located at RM 191. Samples were collected monthly from October 1994 through September 2001. Yakima River hardness is 27 mg/L $CaCO_3$.
(2) Marr et al. 1995a.
(3) Birge et al. 1993; Wedemeyer 1977.
(4) Wedemeyer 1977.
(5) Piper et al. 1982.
(6) Unionized ammonia criterion for 15°C and pH of 7.0 and based on the EPA Gold Book (EPA 440/5-86-001).
(7) From Westin (1974).
(8) The standard is for a geometric mean of multiple samples, with not more than 10 percent exceeding 200 CFU/mL (CFU=colony forming units).

Metabolic rates of fish increase rapidly as temperatures rise. Many biological processes such as spawning and egg hatching are geared to annual temperature changes in the natural environment. Each fish species has a temperature range that it can tolerate, and within that range it has optimal temperatures for growth and reproduction (Piper et al. 1982). Temperatures of 18°C, or 10°C for spawning and egg development, are the upper continuous limits conducive to fish health in hatcheries (Wedemeyer 1977), but temperature requirements vary by species and life stage (EPA Region 10 2001). Temperatures that are too high or too low impart stresses that can dramatically affect production and render fish more susceptible to disease. Most chemical substances dissolve more readily as temperature increases and gases such as oxygen and carbon dioxide become less soluble as temperature rises (Piper et al. 1982). In most cases in the Pacific Northwest, stormwater discharge is not coincident with warmer weather conditions.

The literature indicates that sublethal concentrations of total copper (below those shown to induce mortality in bioassays) can induce physiologic and immune alterations, reduce growth, interfere with reproduction, and lead to behavioral changes in salmonids. Rainbow trout (*Oncorhynchus mykiss*) acclimation to chronic low levels of copper (20 μg/L) has been demonstrated, which is theorized to involve some specific cell types for local metal detoxification in addition to organs such as the liver, which provide whole-body detoxification (Julliard et al. 1993, 1995). A total copper concentration of 20 μg/L is just above that considered to be a non-effect value on fish olfactory systems (Julliard et al. 1995) and fish exposed to 20 μg/L cupric ion for as long as 40 weeks continued to distinguish olfactory clues (Saucier and Astic 1995). Under long-term exposure to low doses of cupric ion, a new steady state of olfactory cell growth and death was hypothesized to account for the olfactory recovery (Julliard et al. 1995).

Acclimation of rainbow trout to copper has also been reported with regard to its effect on gills and lipid metabolism.

Rainbow trout have been shown to be more tolerant than brown trout (*Salvelinus trutta*) to heavy metal concentration increases (Marr et al. 1995b). However, when metal concentrations are increased in conjunction with decreases in hardness and pH (down to 5), rainbow trout showed less acclimation potential than brown trout (Marr et al. 1995a). The Marr et al. (1995a) study was designed to examine the metal response of native populations adapted to high ambient metals in the Clark Fork River, Montana to populations adapted to low background metals. Pre-exposure resulted in increased physiological resiliency to metals; however, that acclimation came at the cost of reduced growth, which could reduce overall fitness (Marr et al. 1995a). Growth inhibition of rainbow trout has been shown at total copper concentrations as low as 4.6 μg/L (Marr et al. 1995a).

Increased susceptibility of rainbow trout to infectious diseases was examined by sublethal exposure to copper (25 μg/L), ammonia (0.5 mg/L), and nitrite (0.24 mg/L) for 24 hours, following exposure to the pathogen *Saprolegnia parasitica* (Carballo et al. 1995). Susceptibility to infection by this pathogen was observed to increase with exposure to ammonia (71 percent), copper (57 percent), and nitrite (50 percent). The reasons hypothesized for increased susceptibility varied by toxicant. Ammonia was hypothesized to be related to interference with mucus production and, therefore, natural mucus removal of the pathogen. Copper was hypothesized to interfere with cellular immune response to the pathogen. Stress response was hypothesized as the cause of increased susceptibility to nitrite (Carballo et al. 1995). An upper limit of 5.0 μg/L total copper was recommended by Wedemeyer (1977) as the upper limit for continuous exposure in hatcheries to avoid suppressing gill ATPase production and to avoid compromising smoltification in anadromous salmonids.

An upper limit of 30 μg/L total lead was recommended by Wedemeyer (1977) for continuous exposure in soft water for hatcheries, where stress and crowding can make fish susceptible to disease and impaired physiological processes. The same limit was recommended by Wedemeyer (1977) for continuous total zinc exposure. This limit closely matches the 10 to 50 μg/L total zinc concentration (median 30 μg/L) shown to trigger rainbow trout behavioral avoidance (Birge et al. 1993).

Ammonia and nitrite can be toxic to fish (Russo and Thurston 1991). Nitrite does not occur naturally or in stormwater runoff at concentrations that are toxic. Nitrate can be high in hatcheries when ammonia is also high due to fish crowding (Russo and Thurston 1991). Nitrate-nitrogen is rarely reported in the hatchery literature, reflecting the fact that conventional flows through fish hatcheries do not have nitrate- or nitrite-nitrogen problems.

Nitrate is relatively non-toxic to fish, although concentrations in excess of 2 mg/L nitrate-nitrogen may have potential to affect salmonid egg development, based on one study by Kincheloe et al. (1979). However, the Kincheloe et al. (1979) study was severely compromised by poor control survival due to a fungal disease that complicated interpretation of these results. Generally, nitrate is more likely to be involved in eutrophication problems before being involved in toxic problems for fish. A maximum allowable concentration of 370 mg/L nitrate-nitrogen was recommended for hatcheries based on bioassay studies for chinook salmon (*Oncorhynchus tshawytscha*) (250 mg/L nitrate-nitrogen for trout) by Westin (1974).

Ammonia toxicity rises with pH and temperature due to the increasing fraction of unionized ammonia that occurs with increasing pH and increasing temperature. Ammonia toxicity is also reported to increase when DO concentrations fall (Russo and Thurston 1991). A 5-year life-cycle test on rainbow trout was performed over three generations at five ammonia exposure concentrations in the range of 0.01 to 0.07 mg/L unionized ammonia. Lesions were common in fish at 0.02 mg/L ammonia and higher concentrations; however, no other major effects were observed, including egg viability, numbers of eggs spawned and produced, growth of progeny, or mortality of parents or progeny of any of the tested generations (Thurston et al. 1984).

Other sublethal tests lasting from 1 week to 3 months were summarized by Russo and Thurston (1991) for both salmonid and non-salmonid species. These tests showed reduced food uptake and assimilation associated with growth inhibition at unionized ammonia concentrations as low as 0.002 to 0.15 mg/L. Pathological and disease-resistance effects under a wide range of water test conditions were reported between unionized ammonia concentrations of 0.04 to 0.4 mg/L.

However, some studies testing ammonia derived from domestic wastewater have attributed toxicity to ammonia that is actually caused by chlorine added to water supplies, based on re-testing. When chlorine was excluded, ammonia concentrations up to 0.4 mg/L for 90 days failed to cause pathologic damage cited as a characteristic in previous studies (Daoust and Ferguson 1984). Wedemeyer (1977) recommended 0.02 mg/L unionized ammonia be used as the limit for continuous exposure, which is lower than the toxicity-based state standards under Chapter 173-201A-040 WAC (Ecology 2000).

There have been numerous studies to determine the effects of turbidity on fish. High turbidity can change fish behavior, growth, physiology, and egg development, or indirectly affect fish by decreasing food supply and cover. Species responses to increased turbidity differ. Some fish stop feeding and seek cover, whereas others migrate into tributaries or up or downstream to clearer water (Chapman and Bjornn 1969, Hartman 1965, Olson et al. 1973, and Ruggles 1966). Physiological reactions to turbidity can include excessive mucus secretion, excretory interference, and respiration interference that can lead to suffocation (Ellis 1944, Trautman 1933). One year and older fish can survive high-suspended sediment for considerable periods (Sorenson et al. 1977). However, suspended sediment ranging from 100 to 6,000 mg/L were found to be limiting to rearing fish, and concentrations greater than 20,000 mg/L are considered acute lethal concentrations for salmonids (Cordone and Kelly 1961, Sorenson et al. 1977). Sigler et al. (1984) exposed coho (*Oncorhynchus kisutch*) and steelhead salmons (*O. mykiss*) to varying turbidity over 14 days. After being exposed to turbidity greater than 165 NTU, no study fish remained in the channel; 35 percent remained at 143 NTU, and 100 percent remained at turbidity ranging from 57 to 77 NTU.

The suggested criterion for salmonid fisheries for pH is between 6 and 9 (Piper at al. 1982, Wedemeyer 1977). Many fish species can live in waters having a pH greater than 9, even for extended periods, but at the cost of reduced growth and reproduction. Fish species have less tolerance of pH extremes at higher temperatures, and as discussed previously, ammonia toxicity becomes an important consideration at high pH (Piper et al. 1982). Trout species could experience mortality at pH greater than 9 (Piper et al. 1982).

The DO content of natural waters varies with temperature, salinity, turbulence, the photosynthetic activity of algae and plants, and atmospheric pressure. Determination of DO concentrations is a fundamental part of water quality analyses since oxygen is involved in, or influences, nearly all chemical and biological processes within water bodies. The lowest safe level of DO for trout is approximately 5 mg/L (Piper et al. 1982). Reduced food consumption by fingerling coho salmon occurs at oxygen concentrations near 4 to 5 mg/L (Piper et al. 1982). Fish death most often occurs if DO drops below 3 mg/L (Piper et al. 1982). Oxygen should be at or near 100 percent saturation and a continual concentration of 80 percent saturation or more provides a desirable oxygen supply (Piper et al. 1982). Oxygen is suggested to be at least 75 percent of saturation at 5°C or 90 percent at 25°C (Wedemeyer 1977). Saturation over 100 percent poses a threat to fish, and any levels over 110 percent are suggested by the literature as lethal (Piper et al. 1982). Gas supersaturation can occur when air is introduced into water under high pressure, which is subsequently lowered, or when water is heated. Water that is plunged over waterfalls or dams, water drawn from deep wells, or water heated from snowmelt is potentially supersaturated (Piper et al. 1982).

At very low alkalinities, water loses its ability to buffer against changes in acidity, and pH may fluctuate quickly and widely to the detriment of fish; therefore, at least 20 mg/L as $CaCO_3$ is the suggested upper limit set for continuous exposure to salmonids (Wedemeyer 1977).

Turbidity and TSS can indirectly affect fish through their food supply, which is frequently a limiting factor (Leonard 1948). Trout feed on aquatic invertebrates year-round and exhibit an annual cycle of growth that closely corresponds to invertebrate numbers (Brown 1946, Maciolek and Needham 1952). The existing Washington State standard of 5 NTU above background provides a relatively high level of protection based on a review of studies on habitat alterations due to reductions in light penetration and direct effects of sediment and turbidity on aquatic life (Lloyd 1987). The literature-based criteria for TSS is 80 mg/L or less (Wedemeyer 1977), or 100 mg/L or less (Cordone and Kelly 1961, Sorenson et al. 1977).

The fecal coliform water quality standard of 100 colonies/100 mL is a geometric mean that makes allowances for occasional higher peak values. Thus, the presence of peaks in fecal coliforms, as expected, would not necessarily be inconsistent with state water quality standards. Increased concentrations of fecal coliforms are a generalized result of residential development. Improperly disposed pet waste and waste from wildlife deposited on impervious surfaces leads to high numbers of fecal coliforms. However, fecal coliform bacteria do not represent a threat to fish habitat value or habitat impairment unless they are correlated with nutrient or toxic (unionized ammonia) loadings as well. The literature does not indicate this association occurs from residential runoff.

3.2.6 Stormwater Quality Results and Discussion for Preferred Alternative 5

Groundwater Impacts

The predicted runoff quality from the UGA was calculated for the proposed project. The calculation combined the predicted untreated stormwater quality with the forecast contaminant removal efficiencies for the enhanced treatment facility and stormwater wetland. The facilities would discharge to an infiltration facility. Both pollutant filtration and capture occur in native soils via infiltration, which results in additional pollutant removal and treatment. The forecast water quality results described in Table 3-11 below assess both the stormwater discharge quality prior to entering the infiltration pond (this column in Table 3-11 does not account for the additional natural removal of contaminant that would occur in the outwash soils of the vadose zone), and for the predicted stormwater quality after crediting infiltration with additional treatment (this column in Table 3-11 credits infiltration with the contaminant removal that would occur naturally in on-site outwash soils). The additional pollutant removal via infiltration in Table 3-11 is conservative, based on the fact that the minimum treatment efficiency (percent removal) is based on filtration via outwash soils, and excludes the filtration that would naturally occur through the lacustrine aquitard, which is present in Management Zone D.

Table 3-11
Alternative 5 Forecast Stormwater Quality

Parameter	Treated Stormwater Discharged to Infiltration (no treatment credit for infiltration)		Treated Stormwater After Infiltration, Prior to Aquifer Mixing (infiltration credited with treatment)			Existing Groundwater Quality [1]	Groundwater Quality Standards
	Management Zones A and B	Management Zone D	Minimum Treatment by Infiltration (%)	Management Zones A and B	Management Zone D		
Copper (µg/L)							
Total	3.0	3.7	50	1.5	1.9	3.0	1000
Dissolved	1.2	1.5		0.60	0.80	1.3	NA
Lead (µg/L)							
Total	2.3	1.8	60	0.92	0.72	1.8	50
Dissolved	1.2	0.97		0.48	0.39	0.95	NA
Zinc (µg/L)							
Total	6.4	19	80	1.3	3.8	76	5000
Dissolved	2.6	7.2		0.52	1.4	30	NA
Nitrate+Nitrite-Nitrogen (mg/L)	0.22	0.27	5.0	0.21	0.26	0.20	10
Ammonia-Nitrogen							
Total (µg/L)	43	77	45	19	42	4.0	NA
Unionized (µg/L)	0.10	0.20		0.10	0.10	0.0	NA
Total Phosphorus (mg/L)	0.02	0.07	65	0.01	0.02	0.02	NA
TSS (mg/L)	0.80	5.2	50	0.40	2.6	3.0	NA
Fecal Coliforms (colonies/ 100 mL)	382	308	100	<1	<1	13	1
Oil and Grease (mg/L)	0.73	0.53	0.0	0.73	0.53	NA	NA

Note: Water quality forecasts and baseline conditions were reported in total metals and converted to dissolved metals. To obtain dissolved concentrations, approximately 40 percent of total copper and total zinc and 53 percent of total lead were assumed to be in dissolved form (King County 1995).

(1) Groundwater quality results for wells OW-1, OW-4, OW-7, and OW-9, December 1998 through September 1999 (AESI 1999c).

Analyses Results Not Crediting Additional Treatment from Infiltration

Stormwater quality after treatment is forecast to have higher concentrations of most stormwater constituents than the existing background groundwater quality, when considered prior to infiltration but after treatment at the surface. TSS, total and dissolved zinc, and total and dissolved lead are the exceptions, and would have concentrations at or below existing groundwater quality. However, prior to any added treatment from infiltration to the native soils, all treated discharge to the infiltration areas would conform to groundwater quality standards.

Infiltration Credited with Additional Treatment

After crediting additional treatment in native on-site soils via infiltration, all parameters would remain within groundwater quality standards. Nitrate-nitrogen and ammonia-nitrogen concentrations in the recharging stormwater are forecast to be higher than the background groundwater concentration (only 5 percent denitrification and no conversion of ammonia to nitrate-nitrogen was assumed as a very conservative assumption; in actuality, almost all ammonia would convert to nitrate-nitrogen).

All treated stormwater in Management Zones A, B, and D would infiltrate through moraine and outwash soils, which have a silty sand component. This component of the on-site soils would more than add the functional equivalence of sand filtration to all the stormwater catchments. An 18-inch depth of sand is one criterion Ecology references to credit infiltration to native soils with water quality treatment (Ecology 2001c), which is more than exceeded by soils under the UGA. Lesser treatment depths also are credited under the Ecology Manual if sufficient vertical removal and pollutant removal capacity is available in the soil. Ecology considers pollutant removal capacity to occur where there is at least 5 milliequivalents (meq)/100 grams (gm) dry weight soil cation exchange capacity. As reported in the Draft EIS, samples of MPR on-site soils had greater than 15 meq/100 gm dry soil cation exchange capability (Land Profile, Inc. 1998). This amount of cation exchange is well above the minimum of 5 meq/100 grams dry soil recommended by Ecology (2001c) for crediting infiltration as a water quality treatment device. Thus, in addition to the treatments for stormwater quality previously described before discharge to infiltration, storm runoff would receive at least the functional equivalence of sand filtration treatment after discharge. Consequently, the infiltration removal values for metals shown in Table 3-11 were set equal to the expected performance for sand filters (see Table 3-3). Fecal coliform removal from infiltration was given a 100 percent removal in Table 3-11, because of the times of unsaturated transit expected from infiltration. TSS was afforded less removal by infiltration than by sand filter treatment because the treatments before infiltration are expected to remove nearly all of the TSS load.

Fecal coliform bacteria values in Table 3-11 are predicted to average over 300 colonies/100 mL before infiltration. A minimum of 3.9 feet of soil has been shown to provide sufficient filtration to prevent fecal coliform contamination (Brown et al. 1979). Fecal coliforms would not reach the groundwater at a concentration greater than the maximum standard of 1 colony/100 mL because they have limited survival and mobility in soils, and because the average vertical transit times of infiltrated stormwater to the aquifer in the three management zones would vary from about 120 to 200 days (AESI 2001). Particles such as fecal coliforms move much more slowly than water through the vadose zone due to physical interference with movement or by electrical charge attraction to the soil particles; most would be trapped altogether by filtration.

Infiltration is also advantageous in phosphorous removal, because phosphorus binds to iron and aluminum in soils in a mineralized form, which effectively removes soluble phosphorus from groundwater. Phosphorus bound to particulates would be removed through filtration.

Analysis Results from Proportionately Mixed Treated Stormwater Compared to the Existing Groundwater Quality

Groundwater impacts were also evaluated by proportionately mixing the treated stormwater from Management Zones A, B, and D and comparing the results to existing groundwater quality prior to any mixing or dilution in the underlying aquifer (Table 3-12). The management zones were proportionately mixed on the basis of impervious acreage. The percent impervious acreage for each management zone was multiplied by each constituent concentration and the results summed across all management zones to achieve the combined (mixed) concentration of each constituent (Table 3-12). The combined constituent concentrations were compared to Washington State groundwater quality standards and background groundwater quality.

Table 3-12
Alternative 5 Proportionately Mixed Treated Stormwater Compared to Existing Groundwater Quality

Parameter	UGA Treated Stormwater After Infiltration, Prior to Aquifer Mixing (Table 3-11) (infiltration credited with treatment)		Proportionately Mixed Groundwater Quality Resulting from the UGA	Existing Groundwater Quality [1]	Groundwater Quality Standards
	Management Zones A and B	Management Zone D			
Copper (µg/L)					
Total	1.5	1.9	1.8	3.0	1000
Dissolved	0.60	0.80	0.70	1.2	NA
Lead (µg/L)					
Total	0.92	0.72	0.82	1.8	50
Dissolved	0.48	0.39	0.43	0.95	NA
Zinc (µg/L)					
Total	1.3	3.8	2.6	76	5000
Dissolved	0.52	1.4	0.98	30	NA
Nitrate+Nitrite-Nitrogen (mg/L)	0.21	0.26	0.24	0.20	10
Ammonia-Nitrogen					
Total (µg/L)	19	42	31	4.0	NA
Unionized (µg/L)	0.10	0.10	0.10	0.0	NA
Total Phosphorus (mg/L)	0.01	0.02	0.02	0.02	NA
TSS (mg/L)	0.40	2.6	1.5	3.0	NA
Fecal Coliforms (colonies/ 100 mL)	<1	<1	<1	13	1
Oil and Grease (mg/L)	0.73	0.53	0.63	NA	NA

Note: Water quality forecasts and baseline conditions were reported in total metals and converted to dissolved metals. To obtain dissolved concentrations, approximately 40 percent of total copper and total zinc and 53 percent of total lead were assumed to be in dissolved form (King County 1995).

[1] Groundwater quality results for wells OW-1, OW-4, OW-7, and OW-9, December 1998 through September 1999 (AESI 1999c).

The analysis forecast that TSS, TP, fecal coliforms, and all of the total and dissolved metals from infiltrated treated stormwater would not influence existing groundwater concentrations because they would be relatively indistinguishable from background groundwater quality in the descending recharge. The remaining constituents, total and dissolved copper, nitrate-nitrogen, and ammonia-nitrogen concentrations would likely have slightly higher concentrations in the descending recharge than exists in the present groundwater. The forecast for proportionately mixed UGA treated stormwater concentrations after infiltration in Table 3-12 (but prior to any aquifer mixing) is very conservative because they were based on filtration through outwash soils, but exclude the additional filtration that would naturally occur through the lacustrine aquitard, which is present under Management Zone D. The analyses are also conservative in that no conversion of ammonia to nitrate was assumed.

Surface Water Impacts

Groundwater from Management Zones B and D ultimately recharges to the Yakima River. The average annual flow in the Yakima River is 1,279,131 acre-feet. This volume is based on flow records collected by USGS at Station 12479500 near Cle Elum, Washington. The average annual developed condition discharge volume from Alternative 5 was not available for analysis; however, Alternative 2 was evaluated in the Draft EIS and would contribute an approximate stormwater volume of 1,763 acre-feet/year. Therefore, Alternative 2 would contribute approximately 0.14 percent of the Yakima River's volume on an average annual basis after development. Although the developed condition discharge volume is not available for Alternative 5, the preferred Alternative 5 would have a lesser degree of development and impervious surface than Alternative 2. It is reasonable to expect that Alternative 5 would contribute less than 0.14 percent of the volume to the Yakima River on an average annual basis.

Surface water impacts from groundwater recharge to surface waters were evaluated conservatively by comparing the proportionately mixed and treated stormwater resulting from the UGA (Management Zones B and D) directly to the existing average Yakima River water quality (Table 3-13). Under natural conditions, the treated stormwater from the UGA would mix with the groundwater and travel approximately 121 to 204 days prior to entering the Yakima River, however data on the rate of groundwater flow under various portions of the UGA site are not available. Therefore, by comparing recharge quality to the river quality and standards directly with no dilution is the most conservative and feasible approach. Analyzed under this conservative approach, all of the water quality constituents in the UGA stormwater recharging the aquifer would meet the state surface water quality standards, except dissolved lead. The predicted dissolved lead concentration of 0.43 μg/L would be slightly elevated above the chronic surface water quality standard of 0.33 μg/L, but unlikely to be distinguishable from the existing background groundwater quality of 0.95 μg/L. All stormwater constituents are interpreted from the analysis to approach background groundwater quality conditions. Dissolved copper, lead and zinc, as well as nitrate+nitrite-nitrogen concentrations in the treated stormwater recharging the aquifer, would be higher than the background concentrations in the Yakima River. However, given that (1) transit and mixing through the underlying aquifer, (2) the UGA development under Alternative 5 would contribute a minimal volume to the Yakima River (less than 0.14 percent of average annual flow), and (3) the overall good quality of the treated stormwater after infiltration, there is no reasonable expectation of adverse change to the water chemistry of the Yakima River.

Table 3-13
Yakima River Water Quality Resulting from the UGA Analysis

Parameter	Proportionately Mixed Stormwater Quality Resulting from the UGA Prior to Aquifer Mixing (Table 3-12)	Average Yakima River Water Quality (1)	Acute and Chronic State Water Quality Standards Class A	Literature Based Sublethal Limits (Table 3-10)
Copper (µg/L) Dissolved	0.71	0.19	3.02/2.37	2.4
Lead (µg/L) Dissolved	0.43	0.01	8.38/0.33	11 to 16
Zinc (µg/L) Dissolved	0.98	0.75	24.2/22.1	12
Ammonia-N (mg/L) Total Unionized	31 0.0	10 0.0	24,000 (2) 66	7,300 20
Nitrate+Nitrite-N (mg/L)	0.24	0.04	None	Less than 250
Total Phosphorus (mg/L)	0.02	0.07	None	--
TSS (mg/L)	1.5	5.6	None	Less than 80
Fecal Coliforms (CFU/100 mL)	<1	11	100 (3)	--

(1) Average Yakima River water quality sampled at RM 191 by Ecology from October 1994 through September 2001.
(2) Unionized ammonia criterion for 15°C and pH of 7.0 and based on the EPA Gold Book (EPA 440/5-86-001).
(3) The standard is for a geometric mean of multiple samples with no more than 10 percent exceeding 100 CFU/100 mL.

3.2.7 Alternative 5 Non-Point Water Quality: Landscaping

The Draft EIS evaluated impacts of pesticides and fertilizers under Alternatives 2, 3, and 4 in Section 3.4.3. This section supplements that discussion for Alternative 5. Non-point influences to groundwater quality would include nutrients and possibly heavy metals from fertilizers applied to landscaping; pesticides and herbicides, which could be applied to landscaping; and fecal coliforms from pet waste.

Commercial pesticides and herbicides can be transported in stormwater runoff. The mobility and persistence of pesticides varies greatly. Organic pesticides used in residential gardens are not reported as a significant problem in surface runoff treatment facilities, but do not occur unpredictability in measurable quantities. Where measured, their appearance tends to be sporadic and has not been associated with toxic effect to surface waters. Metro (1982) reported tentative identification of seven pesticides in five of twenty-one samples collected during its survey of residential and urban areas in the early 1980s. Of the seven pesticides found, all had concentrations in untreated runoff above chronic standards at least once; however, no violations of standards in receiving waters were noted and the report concluded *"due to dilution, flushing, adsorption, and sediment deposition, no acute toxicity problems were discovered in the sites studied"* (Metro 1982). More recently, USGS and Ecology conducted a survey of pesticides in 13 small streams in the Puget Sound Basin, using data collected between 1987 and 1995 (Bortleson and Davis 1997). None of the pesticides detected exceeded existing state or federal freshwater aquatic life criteria. Although no violations of state toxicity standards were found, four pesticides (diazinon, mevinphos, malathion [all insecticides], and diuron [an herbicide]) were found at levels exceeding maximum concentrations recommended by the National Academy of Sciences and National Academy of Engineering for the protection of aquatic life (National Academy of Sciences and National Academy of Engineering 1973). As a result, these products have come under increasing scrutiny. Although there was no definite conclusion of impact in the Bortleson and Davis (1997) study, it did highlight the importance of homeowner education as a source control measure for pesticides.

In a recent study concluded in 1998, USGS, Ecology, and King County tested 10 streams in King County for pesticides. Diazinon was the only pesticide shown to be a problem, but this product was found at levels considered toxic to aquatic life in 9 of the 10 streams. Diazinon was frequently used by homeowners to control European cranefly (*Tipula paludosa*) larvae in lawns. On May 19, 2000, EPA's Office of Pesticide Programs published a Federal Register notice announcing a preliminary human health risk assessment for diazinon and is classified as "restricted use" by the EPA. On December 5, 2000 EPA announced the elimination of all indoor uses of diazinon, and the phase-out of lawn and garden uses. For all indoor household use, retail sales will stop by December 2002. For all lawn, garden, and turf uses, manufacturing diazinon would cease in June 2003, and all sales and distribution would stop in August 2003.

Other pesticides have also come under scrutiny and are being restricted. In June 2000, the EPA released a revised risk assessment and announced an agreement with registrants to eliminate or phase out the pesticide dursban (also known as chlorpyrifos). Chlorpyrifos is commonly found in many home and garden bug sprays and is used in some treatments of termites, as well as on some agricultural crops. The provisions of the agreement and associated EPA actions become effective for the following uses on the following dates: December 31, 2000 dursban uses became prohibited for food and crop uses; December 31, 2001 retails stop the sale of dursban; and dursban would be eliminated from use as a new-home and building construction termiticide by December 31, 2005.

All the UGA stormwater would be infiltrated to soils. The relative mobility of 82 pesticides in soils was assessed, and a minority (17 percent) rated as having moderate or greater mobility (Erickson 1987, Kerle et al. 1996). Of the four pesticides listed above diazinon and diuron have a moderate mobility rating in soils, and mevinphos and malathion have a low mobility rating in soil (Kerle et al. 1996). Even if a pesticide is highly mobile, factors such as application rate, rainfall patterns, soil type, organic matter content, and the adsorption and decomposition characteristics of the soil type determine whether a pesticide is likely to reach the groundwater (Erickson 1987). Pesticide movement within the soil and in the groundwater is usually low because pesticides are often adsorbed and broken down by soil organic matter and microbial biodegradation, therefore transport through the unsaturated zone is reduced (Erickson 1987, Gold 1988).

A.C. Kindig and Company has monitored the water quality at the Snoqualmie Ridge Golf Course over two years for all pesticides proposed for use in their Golf Course Management Plan. Golf course pesticides were monitored at four locations (1 groundwater well, 2 aquifer discharge stations, and a surface water station). No pesticides have been detected at either of the stations. Fertilizers are sources of nutrients, particularly nitrogen and phosphorus. Traces of heavy metals may also be included in some fertilizer blends available to homeowners. A review of the literature on nitrate-nitrogen leaching through turf to reach groundwater shows that, while variable, nitrogen losses rarely exceed 5 percent of the applied nitrogen fertilizer, which generally translates to about 1.6 mg/L nitrate-nitrogen concentration immediately below the turf rooted zone in well-fertilized lawns, and prior to any nitrogen losses by denitrification (Balogh and Walker 1992). Ammonia products in fertilizers are rapidly converted to nitrate-nitrogen within the soil. Nitrate-nitrogen is very mobile in groundwater because it is negatively charged, and is thus not attracted to the negatively charged soils. Some nitrate-nitrogen would be eliminated by denitrification, by which process nitrate-nitrogen is converted to gaseous nitrogen and/or volatile nitrous oxide by anaerobic bacteria during transit through the soil. Losses from denitrification typically range between 10 and 25 percent (Broadbent and Clark 1967) and are independent of the nitrate concentration over a broad range.

Homeowner practices, however, can be unpredictable, and over-fertilization, particularly when associated with overwatering, are the key triggers to nitrate-nitrogen leaching. Consequently, nitrogen loadings from homeowner fertilizer use could be much greater than for professionally managed turf, not only because homeowners can over-apply fertilizer in excess of turf or plant ability to take up nutrients, but they can also compound the problem by overwatering and thus leaching of the over-applied amount. Fertilizer application from residences and other landscaping would be dispersed in space and time over the developed lots, and will generally occur during the drier growing and irrigation season, when leaching from precipitation is less likely. Assuming (1) that the highest average turf-zone concentration for fertilized landscaping would be 1.6 mg/L (Balogh and Walker 1992), (2) 75 percent of landscaping is well fertilized in the longer-term, (3) the remainder of the areas are not fertilized but receive some rainfall with 0.4 mg/L nitrate-nitrogen (from the National Atmospheric Deposition Program station at LaGrande, Washington, in Pierce County; winter and spring 1998 maximum), and (4) 10 percent denitrification occurs, then landscaped areas could locally raise infiltrating groundwater by approximately 1.0 mg/L in nitrate-nitrogen concentration. This concentration is much greater than the concentration from infiltrated stormwater collected by the storm drainage and treatment system (Table 3-11), but nonetheless is well under the 10 mg/L groundwater quality standard (Table 3-11) and literature-based sublethal limits for fisheries (Table 3-10), even prior to any further mixing with other sources of recharge or the underlying aquifer. There are no surface water quality standards for nitrate-nitrogen (Chapter 173-201A-030 WAC), because it is virtually non-toxic. Consequently, there is no reasonable expectation of adverse impact from nitrogen fertilizers to groundwater or the Cle Elum or Yakima Rivers.

There are no drinking water maximum contaminant levels (Chapter 246-290 WAC) for phosphorus, or groundwater maximum standards for phosphorus (Chapter 173-200 WAC). There are no maximum water quality standards for phosphorus in surface waters (Chapter 173-201A WAC). Phosphorus binds readily to iron and aluminum and calcium to form mineralized compounds in soils, and thus is not readily transported subsurface through soils. A number of investigations on the movement of phosphorous applied to soil in effluent from wastewater treatment plants have shown that phosphorous binds to sorption sites in the soil. Even under high phosphorous loading to infiltration basins, for example wastewater treatment plant discharges, the soluble phosphorous front advances only very slowly as sorption sites are occupied. The distribution of phosphorus from effluent at various depths below a six-year-old septic drainfield trench has been measured (Sawhney and Starr 1977). Phosphorus in effluent 15 centimeters (about 6 inches) immediately below the drain trench rapidly reached a concentration equivalent to the septic effluent in the trench, as phosphorus-sorption sites in this horizon were occupied and saturated. Phosphorus concentrations at a 30 centimeter depth (about one foot) throughout the 13 months of monitoring showed a very large fraction was removed between 15 and 30 centimeters, even though the septic system had been in operation for six years. At a 60 centimeter depth below the infiltration trench (about two feet), phosphorus was reduced from an average 13.2 mg/L in the effluent to 0.5 mg/L, which is an approximate 96 percent reduction. Intermittant drainfield flow allowing for alternate drying and wetting of drainfield soils increased sorption capacity for phosphorus (Sawhney and Starr 1977).

Wilhelm et al. (1994) studied an operating 13-year-old single-family septic system to evaluate the geochemical processes occurring below the drainfield, including processes affecting phosphorus transport in the effluent. This drainfield included evaluations in both unsaturated and saturated zones within the underlying shallow aquifer. While phosphorus was removed in both zones, Wilhelm et al. (1994) found that phosphorus was reduced to near-background concentrations quickly in the saturated zone within the shallow aquifer, which was attributed to sorption onto calcium carbonate and precipitation with calcium ions. In 5-day laboratory experiments, Wilhelm et al. (1994) measured a soil sorption capacity of 5 mg/L of soluble phosphorus per 100 grams of soil. However, longer-term experiments and field studies show soils typically sorb with twice this capacity, pointing to the importance of regeneration, likely by re-coating of soils with calcium providing additional binding sites Weiskel and Howes (1992). Wilhelm et al. (1994) found that relative to the initial septic-tank effluent, the groundwater plume that traveled from the domestic septic system was locally and quickly depleted of phosphorus, even after 13 years of operation.

The conclusion that phosphorus is locally reduced to background levels because of its rapid adsorption and precipitation to iron, aluminum, and calcium is supported by Reneau and Pettry (1976), Gilliom and Patmont (1983), and Weiskel and Howes 1992. Given that very high loading sources of phosphorus do not contribute phosphorus above background to the receiving groundwater basin, there is no reasonable expectation that phosphorus from landscaping activity would have any influence on groundwater quality.

Metals in trace amounts could be contained in some fertilizers, however they are also largely immobilized in soils. Metals diffusion in ponds used to treat interstate freeway runoff was measured at less than 0.1 cm^2 per year, and over 95 percent o the total metals were contained in the pond sediments (Yousef et al. 1984). That situation had metals concentrations orders of magnitude higher than could occur with landscaping fertilizers. Therefore, any metals in fertilizers are not expected to migrate with groundwater and result in a measurable change to groundwater quality.

3.2.8 Alternative 5 Artificial Water Bodies

The conceptual site plan for Alternative 5 includes a number of artificial water bodies (small lakes or ponds) to provide landscape and recreational amenities for the community. None of the lakes would discharge to or be connected with a natural surface water. A total surface area of 15 acres is proposed for 8 to 10 ponds or lakes. The largest lake would be approximately 10 acres in size, and located as a part of the neighborhood clubhouse complex. The remaining lakes and ponds would average around three-quarters of an acre in size.

For the large lake, uses would include swimming, non-motorized boating and canoeing, fishing, and waterfowl viewing. A pedestrian trail system would border the lake's perimeter to allow for recessional walking and nature viewing. The lake would be 10 to 15 feet in depth and lined to prevent leakage into the groundwater system. To protect the water quality of the lake, stormwater would not discharge to the lake. Grading adjacent to the lake would be designed to direct shoreline runoff away from the lake.

The water in the large lake would be exchanged at least twice per year with untreated water for irrigation supply to the golf courses on the MPR to assist in maintaining water quality and minimizing undesirable aquatic vegetation growth. The exchange water would lower the temperature in the lake water and would be beneficial for a put and take fishery in the lake. Aeration of the lake at one or more locations could be provided if warranted to maintain circulation, depending upon final design.

For the smaller lakes, perimeter trail walking and nature viewing would be the principal intended use Some of the smaller lakes could have wetland components to attract waterfowl and other bird life. These lakes would also be lined. Water exchange in these smaller lakes is desirable, and could be accomplished by the method described above for the larger lake. Alternatively, treated water to be used for the UGA common area irrigation could be routed through the smaller lakes and then pumped to the points of irrigation.

All the lakes would have controls to maintain water surfaces at their functional design levels. Water needed to make up for evaporative losses from the lakes was included in the water demands calculations for Alternative 5 (Grimm, personal communication 2001).

A lake management and maintenance program is proposed for all artificial lakes, none of which would discharge to a natural water body. The applicable community or property owner association would operate the program for individual lakes. The program(s) would be modeled after existing volunteer lake management and monitoring programs, such as presented in the *EPA's Volunteer Lake Monitoring: A Methods Manual.* For the large lake, coliform bacteria monitoring would be required to assure safe water quality conditions for swimming (Grimm, personal communication 2001).

3.2.9 Alternative 5 Mitigation

- Infiltration is feasible and proposed after stormwater treatment for all areas. Rooftop runoff would be infiltrated without needing water quality treatment.

- Stormwater infiltrated in Management Zones A and B would be treated with a wet pond and a sand filter to give enhanced treatment per the 2001 Ecology Manual, exceeding the treatment required in the manual.

- Stormwater infiltrated in Management Zone D would be treated with a stormwater wetland meeting the basic treatment requirements in the 2001 Ecology Manual.

- Avoidance of external copper (unsealed) and galvanized metal in structures in Management Zones A and B would be a significant source control measure for zinc and copper.

- Native vegetation could be encouraged for common landscaping for commercial and multi-family landscaping. This would minimize the need for landscape chemicals. If chemicals must be used, slow-release fertilizers low in phosphorus, and herbicide or pesticide usage on a minimal "as-needed" basis, selected on the basis of minimal transport and persistence potential, is recommended.

- Educational materials for water quality and habitat/resource protection could be provided to new homeowners to minimize the use of pesticides and lawn and landscape fertilizers.

- Installing covered parking, parking garages, or carports in multi-family and office areas could reduce vehicular contaminants in storm runoff from commercial development.

- To lower the potential for peak fecal coliform concentrations, dumpster areas could be roofed.

3.3 Cumulative Impacts with the MPR

3.3.1 On-Site Operational Impacts

With regard to long-term operational impacts, the MPR DEIS (Appendix C) provided a detailed analysis of impacts from a higher-density MPR plan than is currently proposed. Consequently, the MPR DEIS evaluation conservatively overestimates impacts of the current proposal. The MPR DEIS (Appendix C) provided quantified results for stormwater-generated fecal coliforms, heavy metals, turbidity and TSS, nutrients, oil and grease, landscaping chemical use, golf course pesticides and fertilizer use, and equestrian center impacts to surface and groundwater quality. Cumulative impacts within the UGA were also briefly evaluated prior to publication, though not for the preferred UGA Alternative 5.

Briefly, the MPR DEIS concluded the following for the then-Proposed MPR and Alternative 1:

1. Even with TESC measures in place, some amount of sediment and petroleum hydrocarbon products from construction equipment could reach shallow groundwater in continuity with the Cle Elum River as a result of construction within the geomorphic floodplain of the Cle Elum River. This impact would not occur under the Reduced-Density MPR, because no residential or commercial development is proposed within the geomorphic floodplain.

2. Some large though rare storms and some rain-on-snow events could activate stormwater bypasses which could convey water with some contaminants directly to the Cle Elum River, rather than to on-site infiltration. The magnitude of these events would be greater than the volume of the 100-year storm runoff, however, and that would reduce contaminant concentrations in bypassed flows to very low levels.

3. Infiltration of treated stormwater and other non-point landscaping infiltration was determined to be sufficient to meet criteria for the protection of surface and groundwater beneficial uses.

4. Golf courses on the MPR site would be a source of fertilizers and pesticides that did not previously exist. A golf course management plan was prepared to avoid transport potential and impact, but some amount of pesticides could be leached into groundwater and some could be conveyed in surface flows when bypasses would be activated during large storms. It is not anticipated that under normal use these compounds would be detectable in surface or groundwater any distance from the golf courses. However, accidental spills could occur with localized effects if they were sufficiently large.

5. An equestrian center would increase equine manure loadings at the center and on the equestrian trails. Some localized release of manure and urine to the Cle Elum River could result, but it is not expected these loadings would adversely affect the river due to high irrigation releases by the USBR during nearly all of the warmer and drier seasons when horse riding would typically be enjoyed at the resort.

Since the MPR DEIS, 2 years of data have been collected from runoff from The TPC at Snoqualmie Ridge Golf Course, in Snoqualmie Washington (AESI 1999a). A golf course management plan very similar to that proposed for MountainStar is being used at Snoqualmie Ridge, with allowances for the difference in climate between the east and west sides of the

Cascade Mountains. All pesticides used on the golf course have been tested in streams and in golf course drainage facilities, and there have been no analytical results above detection limits for any pesticide.

MPR operational impacts described in the Draft EIS and summarized above were all negligible with regard to water quality, and this conclusion was reached through evaluation of a proposal with greater impact potential than the current Reduced-Density MPR proposal. Similarly, operational impacts described in this analysis for the Preferred UGA Alternative 5 are negligible. For both, the reasons for negligible change to background conditions resulted from (1) treating stormwater to current BMPs or better, (2) proposing source control measures (most notably a golf course management plan on the MPR), and (3) proposing to infiltrate nearly all stormwater (excluding surface discharge to Stream C for a small portion of the MPR). With the single exception of Stream C on the MPR, there would be no surface discharge from either the MPR or the UGA outside of emergency overflows in excess of the 100-year storm runoff event. Consequently, treated and infiltrated water from both sides would reach and mix with underlying groundwater, then recharge the Cle Elum and Yakima Rivers on a time-attenuated basis (AESI 2001). While the MPR and the UGA projects would represent independent loading sources to the underlying aquifer, on a per-acre basis the quality of the infiltrated water reaching the aquifer was shown for both projects to be well within groundwater beneficial use standards. In addition, the quality of the infiltrated stormwater was also shown to be well within surface water beneficial use standards and fish sublethal evaluation criteria on a concentration basis for each project independently, even under a very conservative assumption in that it directly reached the Yakima or Cle Elum Rivers without any prior groundwater mixing. When actual diffusion, mixing with groundwater, time-attenuated recharge, and mixing with Yakima River or Cle Elum River waters is accounted for, no adverse change to water quality is reasonably forecast.

3.3.2 Suburban 1 Zoned Portion of the UGA

The risk for cumulative operational water quality impacts from potential residential development on the property zoned Suburban 1 in conjunction with development under Alternative 5 and the MPR alone would increase above that described for the MPR and Alternative 5 of the UGA alone. This property is located where the Cle Elum River is in direct continuity with an underlying shallow alluvial aquifer. For this reason, this zone was labeled as the highest risk area of the UGA for development (see Section 3.2.3). Stormwater contaminants would contribute stormwater runoff and possibly treated effluent from on-site septic systems to the underlying shallow alluvial aquifer in direct continuity with the Cle Elum River.

Residential development would add nutrients from fertilizers and increase the risk of pesticide introduction to storm runoff from landscaping chemicals. Along with nutrients, a rural residential land use designation could result in increases in contaminants such as fecal coliforms and BOD, if livestock are held on the property. Very low automobile use would be expected in association with any future project at this density, and effective impervious surfaces generating runoff would also be very low on an area basis. Low level automobile traffic would likely mean that heavy metals and combustible products in stormwater would be inconsequential, certainly on a regional scale. Stormwater treatment for the lots would need to comply with the 1992 Ecology Manual adopted by Kittitas County.

The MPR, Trendwest's UGA Alternative 5, and the property zoned Suburban 1 would all drain stormwater via subsurface routes to the Cle Elum or Yakima Rivers. The MPR and UGA proposals offer treatment to 2001 Ecology Manual standards, and infiltrate through soils suitable for further water quality treatment. It is less likely that development on the Suburban 1 zoned property, if it occurred, would treat to that standard, because the minimum Kittitas County and Cle Elum have yet to adopt the more stringent 2001 Ecology Manual. In addition, infiltration to the alluvial soils is not likely to afford additional treatment after infiltration because they are coarser materials and likely lack the cation exchange capacity that exists elsewhere on the UGA site, which Ecology has determined provides additional water quality treatment after infiltration. As such, potential risks to the Cle Elum or Yakima Rivers from cumulative development would increase. Specific surface and groundwater impacts resulting from future development in this area would be subject to environmental review prior to permits and approvals.

If residential development on the Suburban 1 zoned property were served by on-site septic systems, they would be required to comply with the Kittitas County Department of Health regulations for septic drainfield density and septic installation. If future environmental or Department of Health review resulted in a need for mitigation beyond standard septic installation, various enhanced septic systems or connection to the UGA sewer system would be feasible. The latter would require approval and construction of the UGA Trendwest proposal for connection to the sewer. Potential cumulative impacts to area drainage basins from septic systems associated with cumulative indirect growth are discussed in Section 3.3.2. Additional septic systems on the scale of the Suburban 1 zoned property were not found to have adverse consequences.

3.3.3 Off-Site Impacts

Water Supply

The proposed transfer of consumptive portions of tributary water rights to the MPR and UGA, leaving previous irrigation diversions instream, could only be beneficial to off-site tributary waters, though the amount of benefit to aquatic habitat and biota is not quantified.

Employment-Induced Septic Systems

The combined Reduced-Density MPR and UGA Alternative 5 would result in off-site rural employment-induced housing. To the extent that housing occurred in rural Kittitas County, it would employ septic systems. The potential for adverse impact to surface or groundwater from these added septic systems was evaluated (Exhibit 1) (A.C. Kindig & Co. 2002a).

Septic induced nitrate-nitrogen loadings to groundwater were analyzed by review of the literature, and by quantified analysis using the method of Hantzsche and Finnemore (1992) for 5-acre and 20-acre rural lots. With conventional septic treatment, and under the simplifying and conservative assumption that the septic effluent plus precipitation recharge comprised the total groundwater volume under each lot, a nitrate-nitrogen concentration of 1.28 mg/L and 0.62 mg/L were predicted for 5-acre and 20-acre parcels with single households, respectively (See Appendix A). This result, which assumes no mixing with any underlying aquifer, is well within the 10 mg/L nitrate-nitrogen groundwater and drinking water standards. As expected at these densities, no adverse impacts to beneficial uses of groundwater would be expected. No adverse impacts from septic-origin phosphorus were reasonably expected, given the binding potential of phosphorus to native soils (Exhibit 1).

The evaluation nitrate-nitrogen on a regional subbasin (e.g., tributary basin) scale is provided for the combined Reduced Density MPR and Alternative 5 in Appendix A, reflecting the Master Site Plan for the UGA. Households from MPR and UGA-induced employment were distributed throughout subbasins in Kittitas County as described in the *Employment Induced Water Demand Analysis* (Shapiro and Associates, et. al. 2002). The results show that the maximum change in any of the subbasins would be 35 μg/L nitrate-nitrogen. This change would not have any adverse implications to beneficial uses of surface or groundwater in any basin.

3.3.4 Groundwater Quality

The MPR DEIS evaluated the cumulative impacts to the groundwater quality from stormwater infiltration by the MPR and UGA. A re-analysis of the most intensively built MPR subbasin was performed using alternative methods of analysis requested by Ecology. Two additional analyses, each increasingly conservative in terms of tending to over-estimate impacts, were requested by Ecology to verify the conclusions drawn in the MPR Draft EIS. Results of the two additional analyses confirmed no adverse impacts from the MPR (A.C. Kindig & Co. 2002b). These analyses are contained in the MPR FEIS Addendum (Kittitas County 2002).

3.3.5 Wastewater Treatment Plant Loadings to the Yakima River

Sewage treatment for the MPR and UGA could be accommodated by expansion of the Cle Elum wastewater treatment plant, which discharges to the Yakima River. The City of Cle Elum is planning to construct a regional wastewater treatment plant at the location of the existing plant at the east end of the City. Identification of potential effluent flows and pollutant loads to the Yakima River from Trendwest residential development and the regional wastewater treatment plant is provided in Section 3.16, Utilities, of the Final EIS.

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EXHIBIT 1

Off-Site Rural Employment-Induced Septic System Analysis

EXHIBIT 1
OFF-SITE RURAL EMPLOYMENT-INDUCED SEPTIC SYSTEM ANALYSIS

1.0 Reason for Analysis

The Washington Department of Ecology (Ecology) raised off-site rural area groundwater quality impacts from added septic systems from employment-induced population growth arising from the Reduced Density MountainStar Master Planned Resort (MPR) and the Cle Elum Urban Growth Area (UGA). The issue was raised in a February 16, 2001 letter from Pacific Groundwater Group (PGG) to Ecology (*Re: Initial Evaluation of Issues for Determination of Environmental Benefit from Trend West MPR Water Right Transfer Applications*). Ecology requested Trendwest evaluate whether nitrate and phosphorus concentrations in groundwater would rise as a result of increases in rural Kittitas Co. populations and household septic tanks. The evaluation was requested on individual lot (e.g. 5-acre and 20-acre minimum rural parcel) and regional subbasin (e.g. tributary basin) scales. This analysis is provided for the combined Reduced Density MPR and Alternative 5, reflecting the Master Site Plan for the UGA. The number of employment-induced households in unincorporated Kittitas County, their distribution to tributary subbasins, and mean annual baseflow volumes for each subbasin, are reported in the *Employment-Induced Water Demand Analysis (Shapiro and Associates, et. al. 2002)*. This analysis is Appendix A of the Water Supply Technical Report Supplement in the Trendwest Properties: Cle Elum UGA Final EIS (City of Cle Elum 2001).

2.0 Phosphorus from Employment Induced Septic Drainfields

Phosphorus is often not regulated. There are no regulatory limits specific to phosphorus under the water quality standards. Phosphorus does not have a drinking water maximum contaminant level (Chapter 246-290 WAC), a groundwater maximum standard (Chapter 173-200 WAC), or a maximum surface water quality standard (Chapter 173-201A WAC). For some lakes and fewer rivers, phosphorus controls nuisance algal growth, and limits are placed on it through basin plans or Total Maximum Daily Load [TMDL] plans to prevent or correct human-induced eutrophication (excessive nutrients and algal growth). However, there are no reported eutrophication problems with the surface waters evaluated in this report. Phosphorus is not an issue for groundwater because it phosphorus binds readily to iron and aluminum and calcium to form mineralized compounds in soils, and thus does not readily migrate through soils.

Assuming that septic systems are legally built to minimum requirements of the Washington Department of Health, literature references were evaluated to evaluate the mobility of phosphorus away from a septic drainfield, and thus the risk to surface waters through added septic systems distributed on 5- to 20-acre or larger lots in rural Kittitas County.

A number of investigations on the movement of fertilizer phosphorous and of phosphorous applied to soil in effluent from waste water treatment plants have shown that phosphorous binds to sorption sites in the soil. Even under high phosphorous loading to infiltration basins, for example wastewater treatment plant discharges, the soluble phosphorous front advances only very slowly as sorption sites are occupied. Phosphorus in effluent at various depths below a six-year old septic drainfield trench was measured to see how phosphorus was distributed in the unsaturated soils immediately under the drainfield (Sawhney and Starr 1977). Soluble inorganic phosphorus in effluent 15 centimeters (about 6 inches) immediately below the drain trench rapidly reached a concentration equivalent to the septic effluent in the trench, as phosphorus-sorption sites in this horizon were occupied and saturated. Soluble inorganic phosphorus concentrations at a 30 centimeter depth (about one foot) throughout the 13 months of monitoring showed a very large fraction of the phosphorus was removed between 15 and 30 centimeters, even though the septic system had been in operation for six years. At a 60 centimeter depth below the infiltration trench (about two feet), soluble inorganic phosphorus was reduced from an average 13.2 mg/L in the effluent to 0.5 mg/L, which is an approximate 96 percent reduction. Sawhney and Starr (1977) also examined sorption regeneration, and showed that where septic systems flow to drainfields on an intermittent basis, allowing alternate drying and wetting of the drainfield soils, sorption capacity of the unsaturated soils was increased.

Wilhelm et al. (1994) studied an operating 13-year old single family septic system to evaluate the geochemical processes occurring below the drainfield, including processes affecting soluble phosphorus transport in the effluent. This drainfield included evaluations in both unsaturated and saturated zones within the underlying shallow aquifer. While soluble phosphorus was removed in both zones, Wilhelm et al. (1994) found that phosphate decreased to near-background concentrations quickly in the saturated zone within the shallow aquifer, which was attributed to sorption onto $CaCO_3$ and precipitation with Ca^{++}. In five-day laboratory experiments, Wilhelm et al. (1994) measured a soil sorption capacity of 5 mg/L of soluble phosphorus per 100 grams of soil. However, longer-term experiments and field studies show soils typically sorb with twice this capacity, pointing to the importance of regeneration, likely by re-coating of soils with calcium providing additional binding sites (Weiskel and Howes, 1992). Wilhelm et al. (1994) found that relative to the initial septic-tank effluent, the groundwater plume that traveled from the domestic septic system was locally and quickly depleted of soluble phosphorus, even after 13 years of operation.

The conclusion that phosphorus is locally reduced to background levels is supported by results even where septic systems are placed in medium to coarse sands (Weiskel and Howes 1992). Soluble phosphorus was removed to background levels within 2.4 meters (less than 8 feet) of the drainfield discharge. The rapid immobilization of phosphorus was attributed to inorganic adsorption and precipitation to iron, aluminum, and calcium. Similar results are reported by Reneau and Pettry (1976) and Gilliom and Patmont (1983) for movement of septic-generated phosphorus through saturated soils.

Where water tables are at least a few feet below the drainfield, or where groundwater is shallow and soils are even medium-coarse saturated sands or finer-grained, septic system contributions of phosphorus to groundwater will be minimal. However, if septic systems were to be placed over shallow groundwater with very coarse soils, phosphorus concentrations could be much higher.

The Kittitas County Department of Health (DOH) has the responsibility to regulate septic drainfield density, and regulate septic installation only where soil and depth to groundwater conditions are suitable for adequate removal of all septic constituents. DOH generally prohibits septic drainfield placement in coarser grained soils, except where densities and rainfall are very low and beneficial uses of groundwater resources are not at risk. The DOH requires a depth of greater than 3 feet for vertical separation of any septic drainfield to the seasonally high groundwater table, for conventional gravity systems in fine and medium-grained soils (1B-6 soil types, WAC 246-272-02001) (DOH 2001).

The conclusion is that legally placed septic systems resulting from employment-induced growth from the Cle Elum UGA and MountainStar MPR would not result in export of phosphorus above background to groundwater, or via groundwater to surface waters. There is no reasonably expected adverse impact for any of the tributary subbasins affected by employment-induced population growth.

3.0 Nitrate-nitrogen from Employment Induced Septic Drainfields

Unlike phosphorus, nitrate-nitrogen readily travels sub-surface, and has a 10 mg/L maximum contaminant level health-based drinking water standard (Chapter 246-290 WAC) and a 10 mg/L maximum groundwater quality standard (Chapter 173-200 WAC). There is no maximum standard for nitrate-nitrogen in surface waters (Chapter 173-201A WAC), because nitrate-nitrogen is virtually non-toxic to aquatic organisms. Nitrate loadings from induced septic system construction in rural Kittitas County was evaluated at the 5-acre and 20-acre lot size, and cumulatively in each affected subbasin identified in the *Employment-Induced Water Demand Analysis* (Shapiro and Associates, et. al. 2002)

3.1 Rural Lot Nitrate-Nitrogen Analysis Methods

Septic induced nitrate-nitrogen loadings to ground water was analyzed by review of the literature, and by quantified analysis using the method of Hantzsche and Finnemore (1992) for 5-acre and 20-acre rural lots. This model calculates nitrate-nitrogen loadings and concentrations prior to dilution or mixing in the underlying aquifer. The model mixes percolating recharge waters at steady state over the period of time simulated; in this case annually. Mass balance is inherent in the Hantzsche and Finnemore model, which includes inputs from waste water and recharge from rainfall, and losses from denitrification in the soil column. This approach includes a simplifying assumption that there is uniform mixing of waste water and percolating rainfall over the area being evaluated, which is not an unreasonable assumption at the 5-acre, 20-acre, or subbasin-wide spatial scales in this analysis. The model conservatively assumes full conversion of nitrogen to nitrate. This is usually a reasonable assumption, and at worst would tend to slightly over-estimate the nitrate-nitrogen concentrations where conversion is not 100 percent. The model ignores finer-scale dispersion and lateral flow, and mixing with ground water below the drainfields or from upgradient areas.

Three reference sources were used to estimate daily wastewater flows, the EPA Design Manual for On-site Wastewater Treatment and Disposal (U.S. EPA 1980), Metcalf and Eddy (1991), and measured flow data from the Holmes Harbor Wastewater Reclamation Facility in Freeland, Washington (T. Cleverdon, pers. comm. February 17, 1999). The Holmes Harbor system was selected because the collection system uses a septic tank effluent pump (STEP) and low pressure sewer system that is not susceptible to infiltration and inflow, and because it generated average data from a number of homes.

Average daily flow from a typical residential dwelling is cited by the U.S. EPA (1980) as approximately 45 gal/capita/day (gpcd), with peaks usually no greater than 60 gpcd and seldom exceeding 75 gpcd. Metcalf and Eddy (1991) estimated that typical wastewater flow rates from medium income and luxury homes is 60 gpcd. Flow data from the Holmes Harbor facility for 6 months in late 1998 averaged 15,000 gallons/day. There were approximately 100 residences discharging to the Holmes Harbor system, giving an average flow rate per contributing residence of 150 gpd. Population data from the Washington State Office of Financial Management indicates average dwelling occupancy is 2.5

persons/residence. Applying this occupancy rate to Holmes Harbor yields an average flow rate of 60 gpcd. To add a safety factor to the forecast flow rates, the average number of persons per residence was increased to three. The Kittitas County Health Department estimates a 240 gpd peak flow rate based on 2 bedroom occupancy (Nelson 2001). The average flow rate per residential dwelling of 180 gpd per residence was assumed for the model, which calculates an average annual case.

Nitrate loading is a function of wastewater flow and nitrate concentration of the wastewater. Metcalf and Eddy (1991) estimated average total nitrogen for medium strength wastewater (typical residence) as 40 mg/L. The EPA (1978) publication Management of Small Waste Flows reports two studies in which total nitrogen (TN) concentration in septic tank influent (inflow to the tank) ranged from 36 to 94 mg/L. Approximately 24 to 28 percent TN was removed through settling of solids in the septic tank. As a result, the TN concentration in the effluent ranged from 26 to 76 mg/L , and the mean value of effluent (outflow from the tank) samples was 45 mg/L. These data are very similar to those of Metcalf and Eddy (1991). Averaged, they yield a TN concentration of 53.5 mg/L TN in influent to the septic tank, and 42.5 mg/L in septic tank effluent. This effluent is then dispersed to the soil in the drainfield.

Long (1995) reported nitrogen removal performance for four soil types based on a literature review (course sands to clays). The more porous the media, the less nitrate removal expected. Long (1995) anticipated medium grain sands have an approximate 40 percent denitrification rate. If the 40 percent denitrification rate is used, the effluent concentration of nitrogen (as nitrate) after 3 to 4 feet of travel through the soil column would be approximately 26 mg/L, given a 42.5 mg/L TN effluent from the septic tank. This estimate compares very favorably with data presented in the U.S. EPA Design Manual (1980). Performance measurements of systems in medium sand treating typical household wastewater showed that TN in the effluent ranged from 18 to 33 mg/L. (U.S. EPA Design Manual, 1980). Nonetheless, as an additional conservative measure, denitrification was estimated (county-wide) to average 10 percent and not the 40 percent rate assumed by Long. Hanztsche and Finnemore (1992) suggested a 10 to 25 percent denitrification range was reasonably assumed, based on fertilization studies that should be applicable to septic effluent evaluations.

In summary, the anticipated TN removal for conventional on-site systems occurs through sedimentation of solids (24 to 28 percent reduction), and through a variety of processes (biological removal, immobilization, mineralization) once the effluent reaches the soil drainfield (10 percent reduction). Almost complete conversion of nitrogen compounds to nitrate-nitrogen would be expected as wastewater is treated in the soil. Nitrogen loading per residence, assuming 180 gal/day and 42.5 mg/L nitrogen (as nitrate-nitrogen) effluent, would be about 9.5 kg per year per residence.

The septic loading volumes were based on an average occupancy of 3 people at 60 gal/day each, a TN septic tank effluent concentration of 42.5 mg/L, and a 10 percent denitrification rate. Recharge to the rural lots evaluated would be variable for the employment-induced distribution through rural Kittitas County. For the purposes of this evaluation, recharge was based on the MPR East-C-Creek Subbasin recharge in the MPR DEIS, which was 20.25 inches per year. Rainfall nitrate-nitrogen concentration was set as 0.40 mg/L, which was taken from the National Atmospheric Deposition Program station at LaGrande, Washington, in Pierce County (winter and spring 1998 maximum).

3.2 Rural Lot Nitrate-Nitrogen Analysis Results

With conventional septic treatment, and under the simplifying and conservative assumption that the septic effluent plus precipitation recharge comprised the total groundwater volume under each lot, a nitrate-nitrogen concentration of 1.28 mg/L and 0.62 mg/L were predicted for 5-acre and 20-acre parcels with single households, respectively (Appendix A). This result, which assumes no mixing with any underlying aquifer, is well within the 10 mg/L nitrate-nitrogen ground water and drinking water standards. As expected at these densities, no adverse impacts to beneficial uses of groundwater would be expected.

3.3 Subbasin-wide Nitrate-Nitrogen Analysis Results

Households from MPR and UGA-induced employment were distributed throughout subbasins in Kittitas County as described in the *Employment Induced Water Demand Analysis* (Shapiro and Associates, et. al. 2002). Average annual baseflow volumes for each subbasin were also provided in that document. Using the nitrate loadings per household established above, the change in nitrate-nitrogen concentration that could accrue subbasin wide was also calculated, assuming the entire induced growth septic nitrate-nitrogen load in each subbasin became a part of the average annual baseflow (Table E-1).

The results show that the maximum change in any of the subbasins would be 35 μg/L nitrate-nitrogen. This change would not have any adverse implications to beneficial uses of surface or groundwater in any basin.

Table E-1
Maximum Subbasin Changes in Nitrate-Nitrogen Concentrations from Employment-Induced New Septic Systems in Rural Kittitas County (Reduced-Density MPR and UGA Alternative 5)

Subbasin Number	Number of In-Migrant Households (MPR Reduced Density and UGA Alternative 5)[1]	Total Annual Waste Load (kg/yr)	Mean Annual Base Flow (ac-ft/yr)[2]	Change in Nitrate-Nitrogen Concentration Over Background (µg/L)
Subbasin No. 1 (Cle Elum)	13.3	126.4	115,076	0.89
Subbasin No. 2 (Easton)	43.3	411.4	143,986	2.32
Subbasin No. 3 (Teanaway)	36	342	66,687	4.16
Subbasin No. 4 (Swauk)	12.1	114.9	23,624	3.94
Subbasin No. 5 (Elk heights)	20.9	198.6	13,116	12.3
Subbasin No. 6 (Taneum)	10	95.0	29,542	2.61
Subbasin No. 7 (Reecer)	40.3	382.9	13,248	23.4
Subbasin No. 8 (Thorp)	21.5	204.3	7,099	23.6
Subbasin No. 9 (Wilson-Naneum)	100.3	952.9	22,144	34.9
Subbasin No. 12 (Shushuskin)	6.4	60.8	2,033	24.2

1) Table 6, Employment-Induced Water Demand Analysis MountainStar MPR and Cle Elum UGA, January 7, 2002 (Shapiro and Associates et. al. 2002).
2) Table 10, Employment-Induced Water Demand Analysis MountainStar MPR and Cle Elum UGA, January 7, 2002 (Shapiro and Associates et. al .2002).

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EXHIBIT 2

Hantzche and Finnemore (1992) Model Results for 5-acre and 20-acre Parcels

Mt. Star Estimated nitrate nitrogen concentration at entry to the shallow aquifer from residential septic use.

Date: December 5, 2001

Hantzsche and Finnemore (1992)

Conventional Septic

5 acre parcel **Equation Inputs:**

a	180	(gpd/du)	Waste Loading Rate.
b	1	(du)	Contributing Residential Units.
c	5	(acres)	Acres effluent will be discharged over.
d	42.5	(mg/l)	Septic tank effluent tot. nitrogen concentration.
e	20.25	(in/year)	Recharge Rate for fraction of year considered as steady state
f	0.4	(mg/l)	Nitrate concentration of the rainfall.
g	0.1	(%)	Percent denitrification
h	1	(year)	Fraction of year considered as steady state
I	180	(gpd)	Waste Load
j	24.06	(ft3/day)	Waste Load
k	8783.42	(ft3/year fraction)	Waste Loading Rate
l	0.04	(ft/yr)	Waste Load Distribution
m	0.48	(in/year)	Waste Load Distribution

Final Concentration NO3 1.28 (mg/l)

= ((m*d*(1-g))+(e*f))/(m+e)

Mt. Star Estimated nitrate nitrogen concentration at entry to the shallow aquifer from residential septic use.

Date: December 5, 2001

Hantzsche and Finnemore (1992)

Conventional Septic

20 acre parcel **Equation Inputs:**

a	180	(gpd/du)	Waste Loading Rate.
b	1	(du)	Contributing Residential Units.
c	20	(acres)	Acres effluent will be discharged over.
d	42.5	(mg/l)	Septic tank effluent tot. nitrogen concentration.
e	20.25	(in/year)	Recharge Rate for fraction of year considered as steady state
f	0.4	(mg/l)	Nitrate concentration of the rainfall.
g	0.1	(%)	Percent denitrification
h	1	(year)	Fraction of year considered as steady state
I	180	(gpd)	Waste Load
j	24.06	(ft3/day)	Waste Load
k	8783.42	(ft3/year fraction)	Waste Loading Rate
l	0.01	(ft/yr)	Waste Load Distribution
m	0.12	(in/year)	Waste Load Distribution

Final Concentration NO3 0.62 (mg/l)

= ((m*d*(1-g))+(e*f))/(m+e)