

Scott L. Campbell, ISB No. 2251
MOFFATT, THOMAS, BARRETT, ROCK &
FIELDS, CHARTERED
101 S. Capitol Blvd., 10th Floor
Post Office Box 829
Boise, Idaho 83701
Telephone (208) 345-2000
Facsimile (208) 385-5384
slc@moffatt.com
24810.0000

Attorneys for Protestant
Portneuf Watershed Users Association, Inc.

BEFORE THE DEPARTMENT OF WATER RESOURCES
OF THE STATE OF IDAHO

IN THE MATTER OF APPLICATION FOR
TRANSFER NO. 76779 IN THE NAME OF
THE CITY OF POCA TELLO

**NOTICE OF SERVICE OF PROTESTANT
PORTNEUF WATERSHED USERS
ASSOCIATION'S SECOND SUPPLEMENTAL
ANSWERS AND RESPONSES TO APPLICANT
CITY OF POCA TELLO'S FIRST DISCOVERY
REQUESTS**

NOTICE IS HEREBY GIVEN that on this 21st day of June, 2011, a true and correct copy of **PROTESTANT PORTNEUF WATERSHED USERS ASSOCIATION'S SECOND SUPPLEMENTAL ANSWERS AND RESPONSES TO APPLICANT CITY OF POCA TELLO'S FIRST DISCOVERY REQUESTS** and the foregoing **NOTICE OF SERVICE** was served by the method indicated below, and addressed to the following:

James Cefalo, Hearing Officer
IDAHO DEPARTMENT OF WATER RESOURCES
900 N. Skyline Drive, Suite A
Idaho Falls, ID 83402
Fax: (208) 525-7177

☒ U.S. Mail, Postage Prepaid
☐ Hand Delivered
☐ Overnight Mail
☐ Facsimile

Michael C. Creamer
Jeffrey C. Fereday
GIVENS PURSLEY, LLP
601 W. Bannock Street
P.O. Box 2720
Boise, ID 83701-2720
Fax: (208) 388-1300

☐ U.S. Mail, Postage Prepaid
☒ Hand Delivered
☐ Overnight Mail
☐ Facsimile

Lynden and Candice Symons
SYMONS FARM AND RANCH LLC
P.O. Box 604
Lava Hot Springs, ID 83246

☒ U.S. Mail, Postage Prepaid
☐ Hand Delivered
☐ Overnight Mail
☐ Facsimile

John Schwarz
Box 240
McCammon, ID 83250

☒ U.S. Mail, Postage Prepaid
☐ Hand Delivered
☐ Overnight Mail
☐ Facsimile

James A. Spinner
ATTORNEY AT LAW
P.O. Box 6009
Pocatello, ID 83205-6009
Fax: (208) 232-1808

☒ U.S. Mail, Postage Prepaid
☐ Hand Delivered
☐ Overnight Mail
☐ Facsimile

Travis L. Thompson
BARKER ROSHOLT & SIMPSON, LLP
1010 W. Jefferson, Suite 102
P.O. Box 2139
Boise, ID 83701-2139

☒ U.S. Mail, Postage Prepaid
☐ Hand Delivered
☐ Overnight Mail
☐ Facsimile

Randall C. Budge
RACINE OLSON NYE BUDGE & BAILEY, CHTD.
201 E. Center
P.O. Box 1391
Pocatello, ID 83204-1391

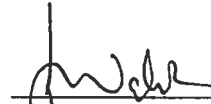
☒ U.S. Mail, Postage Prepaid
☐ Hand Delivered
☐ Overnight Mail
☐ Facsimile

Tom Barnes, President
PORTNEUF-MARSH VALLEY CANAL COMPANY
22465 S. Cambridge Road
Downey, ID 83234

☒ U.S. Mail, Postage Prepaid
☐ Hand Delivered
☐ Overnight Mail
☐ Facsimile

Mark Bennett, Secretary
PORTNEUF-MARSH VALLEY CANAL COMPANY
3670 E. Richards Road
Downey, ID 83234

☒ U.S. Mail, Postage Prepaid
☐ Hand Delivered
☐ Overnight Mail
☐ Facsimile



Andrew J. Waldera

Andy Waldera

From: Andy Waldera
Sent: Thursday, May 05, 2011 12:54 PM
To: Andy Waldera
Subject: Portneuf Litigation Memo (Updated/Supplemented from 5/3/11 Version)

Scott,

The purpose of this e-mail is to document several of my observations in the Portneuf matter now that I have reviewed the documents produced by both sides to date, and the City of Pocatello's discovery responses. I am reducing the following thoughts to writing so that I do not lose them to the ether. The following are things that caught my eye and want you to be aware of.

The e-mail is typed in 14-point font so that you do not go blind reading it.

1. Rapidly approaching (and other) deadlines:

(a) May 2, 2011--written discovery service deadline

Pocatello used all of its ROGS, so we will not see any more of those. Unless we see some new RFPs (which I do not expect to see) we will be in "supplement" mode as necessary.

On the other hand, we do have one remaining ROG. We used that last ROG and a corresponding, follow-up RFP to seek information regarding the advent of the PMVCC storage water shrink free delivery status. That ROG and RFP went out the door on May 2.

(b) May 16, 2011--initial disclosure of "potential" witnesses, both fact and expert

For purposes of this deadline, we will re-package the witnesses we already identified in our prior discovery responses, including Christian Petrich.

(c) May 2011--deposition of Watermaster Steve Hebdon

The scheduling order requires the parties to coordinate the deposition of Steve Hebdon so that it takes place by the end of May. I think that it would be worth our while to interview him at a bare minimum, and we likely want to depose him regarding operations and measurement of the Portneuf, and any concerns he might have regarding the transfer (I know that he has some concerns regarding accounting, measurement, and administration based upon some notes I have reviewed from Pocatello expert Greg Sullivan...though Sullivan's notes do not fully explain the nature of Hebdon's concerns. See Pocatello 81).

(d) June 3, 2011--expert witness report production

Self explanatory.

(e) June 24, 2011--expert witness deposition deadline

5/9/2011

CONFIDENTIAL

PET00020

Remains to be seen if a deposition of Pocatello expert (Greg Sullivan) is necessary. Have to review forthcoming report, and defer to Christian in large part. In the meantime, Christian does have some difference of opinion regarding river channel conveyance losses.

(f) July 1, 2011--pre-hearing motions/motions in limine filing deadline

Self explanatory.

(g) July 13, 2011--rebuttal expert report disclosure deadline

Interesting that this deadline is after the expert witness deposition cut-off, but expert deposition may be necessary in order to better draft any rebuttal report. Again, Christian's input will be necessary in determining whether rebuttal report is necessary.

2. Pocatello's positions/view of the appropriateness of its proposed transfer:

(a) Pocatello is entitled to shrink free/without conveyance loss delivery of the 3,100 AF for two reasons: (1) the partial decrees for water right nos. 29-53 and 29-589 provide for such delivery down to the PMVCC canal heading; and (2) because the reach of the Portneuf River downstream of the PMVCC canal heading to Pocatello is a gaining reach. To be clear, Pocatello is not seeking decree-based shrink free delivery all the way to Pocatello. Instead, Pocatello is seeking decree-based shrink free delivery only to the PMVCC canal heading. See Response to Interrogatory Nos. 1 and 8. The requested shrink free delivery status from the PMVCC canal heading to Pocatello is based solely upon Pocatello's contention that that reach of river is a gaining reach.

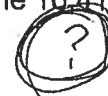
PWUA's position is that: (1) the decree-based shrink free status applies only to protect/foster the beneficial use of irrigation, not other uses of the water; (2) the shrink free status extends only to the irrigation of 5,103 acres of land located within the PMVCC service area; and (3) the operation and administration of the pertinent partial decrees are subject to the provisions of the Portneuf River Settlement Agreement, which contemplates only the "irrigation" use of the applicable water rights, and applies only for the benefit of the parties to the agreement. Pocatello was not a party to the Agreement and, therefore, does not benefit from the agreement. Further, Pocatello is bound by the partial decrees and by the Agreement having been a party to the SRBA (i.e., the irrigation nature of use within the PMVCC service area, and limitation that benefits of the Agreement extend to Agreement signatories only).

(b) The transfer will not cause any additional or atypical flooding because it does not seek to alter historic operations of the river or Chesterfield Reservoir. The storage component under water right nos. 29-53 and 29-589 provides for the storage of 16,410 AF in Chesterfield Reservoir, and the transfer does nothing to change the storage and release of that volume of water. See Response to Interrogatory Nos. 2, 3, and 7.

PWUA's position is that the transfer will require the alteration of historic river and reservoir operations. This is largely because the 3,100 AF of storage water purchased by Pocatello is "new" water. I say "new" because it is a quantity of water above and beyond the 16,410

5/9/2011

CONFIDENTIAL



PET00021

AF stored in the reservoir that has been recaptured through the lining of the PMVCC canal. Historically, Pocatello's 3,100 AF was lost through the bottom of the canal. The canal lining will (or is anticipated to) negate this former loss...which makes sense because PMVCC should be leery of selling any of its water given that its supplies are tight. However, PMVCC probably figures that the sale of the water to Pocatello is a wash. While PMVCC is losing 3,100 AF of Chesterfield storage water, it is picking up at least that much water (3,100 AF) through the canal lining. Thus, PMVCC really is not losing any water though the sale at all (if all goes as the NRCS estimates...which is an interesting aside in and of itself because NRCS estimates potential water savings from the lining to exceed 7,000 AF). PMVCC still gets 16,410 AF of water under the rights, and Pocatello gets the "new" 3,100 AF of water resulting from the canal lining. In my mind, the true arithmetic in this case under the water rights is 19,510 AF of water under the subject water rights (16,410 from the reservoir + 3,100 AF of "new" water developed out of reservoir and off river channel via the canal lining). This math should, in my opinion, provide a compelling case for expansion of the water rights, at least in volume, if not flow rate. To date, we have no evidence that PMVCC will be drying up 3,100 AF of consumptive use within its boundaries to off-set the 3,100 AF recaptured from canal lining and conveyed to Pocatello. ** Was crop delivery history included in investigation?*

In putting a finer point on the concept that there is no evidence suggesting that PMVCC is going to reduce its consumptive use of water by 3,100 AF to off-set Pocatello's downstream use is Pocatello's Application for Transfer itself. Part 1, Section 5 (d) of the application asks Pocatello to: "Describe the effect on the land now irrigated if the place or purpose of use is changed pursuant to this transfer?" In addition to referring to the Sullivan/Spronk Report, Pocatello responded: "No effects are anticipated." To me, this statement confirms that PMVCC will continue to operate and irrigate "business as usual," and that PMVCC will not be reducing its consumptive use to off-set that of Pocatello.

A related point to PMVCC's recapture of a "new" block of water, as hinted above, is the fact that the NRCS estimates that the "Realistic water savings" from reconstructing and lining the PMVCC canal is "7,000 ac-ft." Assuming this estimate is correct, this transfer proceeding arguably implicates a greater enlargement than just 3,100 AF. If PMVCC recaptures (really creates or frees up) 7,000 AF annually from canal lining within its system, that water has to go somewhere. 3,100 AF will be used by Pocatello, leaving 3,900 AF of additional water per year for PMVCC use. This extra water raises a few of possibilities: (1) PMVCC can use the water to irrigate additional acres that do not currently have water; (2) PMVCC can supplement existing per-acre water allotments with the extra water; and/or (3) PMVCC can effectively extend its irrigation season because the 3,900 AF of extra off-channel/off-reservoir water will likely allow the company to delay (and, therefore, extend) storage water releases. Extending the storage release season would extend/exacerbate problems attributed to storage releases (flooding, mossing, sedimentation, shrink-free delivery/natural flow carriage water demand). If the NRCS estimates prove correct (and, at this point, Christian is somewhat skeptical due to lack of data), then PMVCC's consumptive use of water will expand by 7,000 AF annually (3,100 AF used by Pocatello + 3,900 AF used by PMVCC within its delivery system).

Pocatello's view that the transfer is not seeking to alter historic storage and river flow regimes does make sense, at least initially. Pocatello argues that its 3,100 AF of water is

5/9/2011

CONFIDENTIAL

PET00022

part of the 16,410 AF authorized for storage under the subject water rights...which it is because the 3,100 AF of "new" water is recaptured off stream down in the PMVCC canal. Consequently, Chesterfield will still only store 16,410 AF of water under the water rights, and only 16,410 AF of water under the rights will be conveyed/delivered down the Portneuf River. PMVCC will pick up the extra 3,100 AF off channel and in its canal for use within its service area (i.e., the 3,100 AF of "new" water will never be stored in the reservoir or conveyed through the river; therefore, historic storage and conveyance burdens in the reservoir and river, respectively, will remain unchanged). However, the timing and flow rate of Pocatello stored water releases are where reservoir and river operations will likely diverge from historical patterns. *Concept of "new" water:*

(1) more carryover (2) same release

First, Pocatello's storage releases will be year-round if the transfer is approved, as opposed to irrigation season only (based upon the current irrigation use configuration of the subject water rights). This alone is a difference in reservoir/river operations. Year-round delivery of water to Pocatello raises at least two valid and interrelated concerns: (1) the added expense and administration that year-round operation of Water District 29, and its watermaster, would bring from a financial standpoint (burden); and (2) the necessary creation and implementation of a complicated water measurement/allocation/administration framework that does not already exist. It is well known that Water District 29 water users grouse annually about the financial expense they bear in supporting district operations during the irrigation season. Year-round administration will bring only more expense and financial burden. Further, winter (or out of irrigation season) storage releases for Pocatello will require careful measurement and accounting of reservoir storage operations so that storage limitations within the reservoir are not exceeded. For example, water right nos. 29-53 and 29-589 authorize the one-time fill of 16,410 AF in any given year. If Pocatello requests a storage release during storage season, that release must be debited against that season's storage allotment. Put differently, Pocatello releases cannot be allowed to create a storage "hole" that is subsequently filled in and masked by ongoing storage during the storage season. This potential outcome of the transfer will require additional measurement/accounting oversight than presently exists in Water District 29.

Second, the Storage Water Agreement ("Agreement") between Pocatello and the PMVCC (Transfer Attachment Nos. 2a and 3) provides that the PMVCC "shall convey . . . a share of the outlet capacity and conveyance capacity of the Company's Water System from Chesterfield Reservoir to the point of delivery to the City on the Portneuf River sufficient to deliver up to a maximum flow rate of 50 cubic feet per second," and to "have the 3,100 acre-feet of water released . . . [and] delivered at a rate of up to 50 cubic feet per second . . . in the amounts and at the times as requested by the City." Agreement at Section 1. The Agreement further provides that "at the City's request" the water "shall be released . . . up to a maximum rate of 50 cubic feet per second for purposes designated by the City in its sole discretion." Agreement at Section 10.C. Consequently, river operations under the Agreement will require one of two scenarios: either (1) the displacement of PMVCC water conveyance within the river channel during the height of irrigation season in order to make room (free up river channel capacity) for the delivery of Pocatello water; or (2) the simultaneous delivery of both PMVCC and Pocatello water which would likely quickly exceed river channel capacity and promote flooding.

CONFIDENTIAL

5/9/2011

PET00023

Whether PMVCC realizes it or not, I contend that the Agreement requires PMVCC to "make space" in the river channel by subordinating its own stored water conveyances within the river in favor of Pocatello water delivery up to 50 cfs at a time. Pocatello likely interprets the Agreement in a similar fashion given its discovery responses claiming that historic reservoir and river operations will not be altered. Typical storage release flows in the summer (based upon watermaster records) appear to range between 105 and 125 cfs. These flow rates already cause some localized flooding. If Pocatello demanded delivery of 50 cfs for a couple of days, or even a week, that means PMVCC would only be able to convey between 55 and 75 cfs of its water in-channel...a subordination of between 40-48% of PMVCC water delivery to make enough space in the river channel to accommodate Pocatello water. This is significant because Pocatello could take delivery of water at a rate of 50 cfs for over 31 days before exhausting its 3,100 AF storage allotment. If Pocatello were to demand delivery in one 31 consecutive-day block, PMVCC would likely have difficulty meeting its own storage water delivery needs if the Agreement requires delivery subordination like I think it does. Maybe PMVCC does not care because it figures that the canal lining savings within its system will cover times of delivery subordination, or maybe PMVCC does not care because it does not believe that it needs to subordinate its deliveries to implement the Agreement. Regardless, delivery of water to Pocatello under the Agreement will alter historic reservoir and storage operations, and approval of the transfer will require sufficient conditions to guard against PWUA injury and property damage.

Greg Sullivan's notes also include references to flooding:

--Pocatello 19 ("Flood prone area between Topaz Dempsey and Lava caused by travertine build up in the channel has raised channel [and] moss in river");

--Pocatello 84 ("Flood problem mostly between Topaz Ditch/Dempsey Ditch and Lava Canal; Interferes with haying; running reservoir water at time when they are haying").

(c) Suitable accounting, measurement, and administration procedures are already in place to protect existing, senior water rights. The Chesterfield Reservoir outlet is monitored and adjusted daily to account for storage releases and the pass-through of reservoir in-flows during the irrigation season. Also, adequate measuring devices already exist both above and below the reservoir for accounting purposes. See Response to Interrogatory Nos. 5 and 6.

PWUA's position is that current measurement and accounting procedures are not sufficient to properly/confidently administer water supplies on the Portneuf River and its tributaries. As a threshold matter, there is no permanent measuring device installed above the reservoir measuring reservoir in-flows. This fact is confirmed via: (1) telephone conversations between Christian Petrich and watermaster Steve Hebdon; (2) from the notes of Pocatello expert Greg Sullivan (the reservoir "inflow gage was removed by someone from the tribe" (see Pocatello 80) and "Steve H eyeballs" river and tributary flows into the reservoir (see Pocatello 15)); and (3) Pocatello's concession that the Portneuf River channel upstream of the reservoir is "wide and braided and is generally unsuited for installation of a permanent measuring device. Further, the area upstream of Chesterfield Reservoir is located on the Fort Hall Indian Reservation making construction access to the site difficult" (see Response to Interrogatory No. 3).

5/9/2011

CONFIDENTIAL

PET00024

Overall lack of administration and detailed water measurement within Water District 29 is well documented. In addition to the lack of a measuring device above the reservoir, water users and the IDWR have long questioned the accuracy and adequacy of the Chesterfield Reservoir elevation gage and other measuring devices throughout the system which suffer from mossing, sedimentation, scaling, and general poor condition. See, e.g., PWUA 67-71; 410-427; 1340-41; 1342-43; 1344-45; 1875-78; 1899-1903; and 2391-92. Greg Sullivan field notes also raise accuracy concerns over the Topaz gage. See Pocatello 19 ("If the gage was accurate . . .").

PWUA is also very concerned with the lack of measurement provided at Pebble, above the Topaz Gage. The USGS used to maintain an active gaging station at Pebble where the the 8-mile drainage canal (the channelized section of the river immediately downstream of the reservoir) discharged back into the original river channel. The Pebble gage is an important measurement location that used to capture spring flows and shallow groundwater flows tributary to the river, but that where intercepted by the 8-mile (i.e., Downey) canal after its construction in conjunction with Chesterfield Reservoir. The Pebble gaging station operated from 1911 to 1913, and then again from 1969 to 1977. PWUA members are convinced that these flows, captured upon the construction of the Downey Canal (channelization of the river), is what led to shrink-free delivery status of the PMVCC storage rights, and amount to natural background flows that otherwise belong to natural flow water rights...not to PMVCC or Pocatello.

(d) Approval of Pocatello's proposed transfer is in the public interest because the transfer will maximize the beneficial use of water, augment necessary municipal water supply, and improve instream flows during those times when Pocatello is calling for storage water releases. See Response to Interrogatory No. 11.

PWUA's position is that the proposed transfer is not in the local public interest due to increased flooding risks, enlargement of the underlying water rights, risk of injury to senior, natural flow-based water rights, and negative impacts upon the local agricultural base due to injury to senior natural flow-based water rights. The proposed transfer also threatens to reduce the local agricultural base through the change in use of the water from ag-based irrigation to municipal-based irrigation, as well as the overall change in nature of use from irrigation to other non-irrigation-based uses (municipal, aesthetics, wildlife, aquifer recharge, etc.).

PWUA is also concerned that increased releases of storage water from Chesterfield Reservoir will exacerbate moss and sediment conditions in the river channel during irrigation season. These conditions further decrease channel capacity and threaten additional flooding through lost capacity and through the injection of greater quantities of storage water discharged into the river channel as a means of compensating for, and overcoming, decreased in-channel conveyance efficiencies. Decreased river channel conveyance efficiency also places greater carriage water demand upon natural river flows in order to satisfy the shrink-free delivery of water to the PMVCC canal. Finally, lining of the PMVCC canal in order to recapture the 3,100 AF of water sold to Pocatello has the potential to increase conveyance losses within the river channel by decreasing canal seepage contributions to local groundwater flows tributary to the river. The diminishment/negation of potential groundwater and/or spring water-fed river reach gains

5/9/2011

CONFIDENTIAL

PET00025

via canal lining would require the use of even more natural river flow for carriage purposes in order to maintain shrink-free water deliveries.

Documents demonstrating mossing and seasonal flooding include PWUA 67-71, 410-27, 454-59, 1152-53, and 3122-133. Though Pocatello's discovery responses state that it "has no direct knowledge of the sources or causes of alleged physical impairment or interference with Portneuf River channel capacity in the reaches adjacent to the lands of the PWUA members" (see Response to Interrogatory No. 10), the field and interview notes of Pocatello expert Greg Sullivan provide otherwise. For example, Sullivan's notes state that "stored water makes the moss worse" (see Pocatello 81). Unfortunately, however, Sullivan's notes do not elaborate any further on the statement.

3. Miscellaneous:

(a) According to Greg Sullivan's notes, Eric Olson is the Superintendent and ditch rider for the PMVCC. Eric relayed to Greg that PMVCC canal losses are estimated to be as high as 50% ("divert 120 cfs to deliver 60 cfs"; "Eric knows where the losses occur"). See Pocatello 28.

(b) At this year's Water District 29 meeting, a PMVCC member told PWUA members that if they wanted to know where all of their water was going that they should go up above the old Pebble Gage site and watch how river/spring flows diminish once groundwater users start operating their irrigation wells. This premise is seconded by Dan, though out of priority groundwater pumping is largely a matter for a separate administrative proceeding for another day. However, the negative implications of groundwater pumping are also contained within Greg Sullivan's notes where he scribbled that most irrigation between the reservoir and Topaz/Dempsey canal heading is from "wells in the alluvial aquifer." Sullivan further noted that Eric Olson told him that "they notice" when the wells start pumping. See Pocatello 18.

Delivery call by PWUA on Portneuf Valley irrigation (?)

(c) Greg Sullivan's notes also relay that the Portneuf River Valley "pinches down near [the] intersection of Kelly Toponce Road and Highway; aquifer ends and springs come up and contribute 80-100 cfs to river; -as low as 50 cfs." See Pocatello 18.

(d) Sullivan's notes state that water travel time from the reservoir to the PMVCC Canal headgate on river equals between 30 and 42 hours.

— where?

(e) Sullivan's notes refer to a section on the river known as "the falls." He states that the river dries up at the falls, but that, according to Steve Hebdon, the river has never been dry below the falls. See Pocatello 82-84.

(f) On Sullivan's field visit on 8/19/10, PMVCC personnel expressed concern that Pocatello had yet to file its transfer application (which was subsequently filed on 8/30/10) because without Pocatello's money from the storage right purchase, PMVCC would not have the necessary matching funds for the canal lining project to secure the federal NRCS dollars. See Pocatello 14.

(g) The Sullivan report (Attachment No. 9 to transfer application) ("Report") states that Chesterfield Reservoir storage volumes are measured using staff gage elevation measurements via a stage/capacity relationship, last updated by IDWR in 1997. Report at

5/9/2011

CONFIDENTIAL

PET00026

*What has value?
Inflow below Pelicle!*

*Condition is updated
summary*

Page 8 of 9

4-5. As discussed above, IDWR, among others, has questioned the accuracy of the reservoir elevation gage as a threshold matter. Moreover, the fact that the stage/capacity relationship was last updated in 1997 (over 13 years ago) suggests that the PWUA's measurement concerns are valid.

(h) The Sullivan Report acknowledges that the transfer proposes to change "the times when the Acquired Storage Water Rights may be released from storage for the City's use." Report at 6. As discussed above, this raises concern over the need for more detailed, and now year-round, storage and release accounting to prevent injury. This additional administration will also bring additional expense that will have to be borne by the Water District 29 water users, unless Pocatello agrees to cover non-irrigation watermaster/administration expenses on its own and without irrigation water user contribution.

(i) The Sullivan Report states: "The proposed Pocatello place of use . . . is subject to change in the future, and Pocatello will submit GIS shape files to IDWR reflecting any changes to the place of use as they occur." Report at 6-7. Though it is not entirely clear to me, I presume that Pocatello's transfer seeks an overall permissible place of use, with actual locations of use contained within the larger PPU. If its does not, and the POU proposed does actually change in the future, I would contend that Pocatello would have to complete a subsequent transfer to track the changes.

(j) The Sullivan Report details that "Pocatello proposes to use and reuse return flows from initial municipal use of the Acquired Storage Water Rights to extinction." Report at 8-9. Translation, whatever municipal use Pocatello makes of any portion of the 3,100 AF will be 100% consumptive. Consequently, and because one of the transfer's proposed uses is "municipal" use, IDWR needs to evaluate this transfer under the premise that Pocatello's use of the 3,100 AF will be entirely consumptive. *Issue: "shrink free before res. release and after 9/15"*

(k) Though a contested issue in this transfer proceeding, Sullivan's Report notes that: "Pocatello's legal counsel has advised that the City should be entitled to the benefit of [the shrink-free] delivery of the Acquired Storage Water Rights to the vicinity of the Portneuf-Marsh Valley Canal headgate." Report at 15-16.

(l) With respect to the stretch of river downstream from the PMVCC canal headgate to Pocatello's proposed points of diversion, Sullivan opines that: "the Portneuf River from the Topaz gage to the Pocatello gage is a gaining stream, therefore, it is expected that there would be little or no transit loss in delivering the Acquired Storage Water Rights to Pocatello except for temporary bank storage." Report at 17. However, Sullivan does concede that this same stretch of river did LOSE water (832 AF) in 1992. Report at 16-17. This supposedly gaining reach, coupled with the decreed shrink-free delivery status to the PMVCC canal headgate, is why/how Pocatello seeks full delivery of its 3,100 AF of water free from conveyance loss. However, in the Storage Water Agreement between Pocatello and PMVCC, Pocatello agrees/acknowledges that it "shall bear any conveyance losses that might be imposed by the District 29 Watermaster in delivering the Acquired Storage Water Rights downstream of the [PMVCC canal heading]." Agreement at Section 10.E.

I have not shared this e-mail with Christian yet because I wanted you to see it first in case there are things that you would like to add or omit.

5/9/2011

CONFIDENTIAL

PET00027

--Andy

CONFIDENTIAL

5/9/2011

PET00028

Moffatt Thomas

MOFFATT THOMAS BARRETT ROCK & FIELDS, CHTD.

101 S. Capitol Blvd., 10th Floor
Boise, ID 83702
208 345 2000 ~ 208 385 5384 Fax

June 21, 2011

MTBR&F File
No. 24810.0000

CD-R

Disk 1 of 2

In The Matter Of
Application For
Transfer No. 76779
In The Name Of
The City Of Pocatello

**SPF WATER ENGINEERS
PRODUCTION**

PET00001

Moffatt Thomas

MOFFATT THOMAS BARRETT ROCK & FIELDS, CHTD.

101 S. Capitol Blvd., 10th Floor
Boise, ID 83702
208 345 2000 ~ 208 385 5384 Fax

June 21, 2011

MTBR&F File
No. 24810.0000

CD-R

Disk 2 of 2

In The Matter Of
Application For
Transfer No. 76779
In The Name Of
The City Of Pocatello

**SPF WATER ENGINEERS
PRODUCTION**

PET00014

Directory of D:*.*

Page 1 06/21/2011 1:13pm Sorted by Long Name

Levels: All Disk Label: 110616_1246

D:\

File 272.0170.pdf	Adobe Acrobat Docume	18,000,637	R
-------------------	----------------------	------------	---

1 file; 18,000,636 bytes

D:\Background

Seepage Loss Estimate (NRCS).pdf	Adobe Acrobat Docume	88,011	R
Spronk Expert Report w attachments.pdf	Adobe Acrobat Docume	3,863,294	R
Sullivan notes dated 8-18-10 and 8-19-10.pdf	Adobe Acrobat Docume	14,074,265	R
Transfer No 76779.pdf	Adobe Acrobat Docume	14,287,404	R
Various PWUA letters and docs.pdf	Adobe Acrobat Docume	2,854,738	R

5 files; 35,167,712 bytes

D:\Background\JACK BARNETT CONSULTING

0 files; 0 bytes

D:\Background\JACK BARNETT CONSULTING\INCOMING

01 update.htm	HTML Document	3,290	R
Hearing Outline.doc	Microsoft Word Docume	88,576	R
Portneuf River 1-24-2007.htm	HTML Document	2,382	R
RE Hearing Next Tuesday 4-27-07.htm	HTML Document	20,345	R
RE Hearing Next Tuesday 4-28-07 7_26 pm.htm	HTML Document	20,328	R
RE Portneuf Water Settlement 1-8-2007.htm	HTML Document	19,156	R
Update 4-6-07.htm	HTML Document	1,452	R
Update on efforts 1-29-2007.htm	HTML Document	2,766	R

8 files; 158,295 bytes

D:\Background\JACK BARNETT CONSULTING\INCOMING\Graphical Drop Analysis

Graphical Drop Analysis 4-25-07.htm	HTML Document	14,140	R
Graphical Drop Analysis 4-25-07.xls	Microsoft Excel Worksh	728,576	R
Graphical Drop Analysis 4-27-07.xls	Microsoft Excel Worksh	728,576	R
Ideas 4-27-07.htm	HTML Document	17,203	R

4 files; 1,488,495 bytes

D:\Background\JACK BARNETT CONSULTING\INCOMING\Graphical Drop Analysis\Topaz Guage Data Analysis

FW Topaz Gage Data Analysis.xls.htm	HTML Document	7,896	R
Topaz Gage Data Analysis 4-27-07.xls	Microsoft Excel Worksh	6,026,752	R

2 files; 6,034,648 bytes

D:\Background\JACK BARNETT CONSULTING\INCOMING\Water Out of Priority

FW WATER OUT OF PRIORITY 40 OF 80 YEARS4-30-07 6_12 pm.htm	HTML Document	24,250	R
OUT OF PRIORITY WATER 40 OF 80 YEARS 4-30-07.xls	Microsoft Excel Worksh	481,792	R

2 files; 506,042 bytes

PET00002

Directory of D:*.*

Page 2 06/21/2011 1:13pm Sorted by Long Name

Levels: All Disk Label: 110616_1246

D:\Background\JACK BARNETT CONSULTING\OUT GOING

01 Jack B 1-8-07.doc	Microsoft Word Docume	30,208	R
01 Jack.doc	Microsoft Word Docume	37,376	R
39 cfs junior users and Tuthills memo on the juniors p roblem.pdf	Adobe Acrobat Docume	95,744	R
Basin 29 Settlement Agreement Options- Lava & De mpsey Irriga.doc	Microsoft Word Docume	25,600	R
Data for 20 day average.doc	Microsoft Word Docume	29,184	R
Data for 20 Day Comparison.doc	Microsoft Word Docume	29,696	R
FW Portneuf River and Topaz Gage 3-2-07.htm	HTML Document	13,563	R
Jack 1-24-07.doc	Microsoft Word Docume	31,744	R
Jack B 1-5-07.doc	Microsoft Word Docume	30,208	R
MASTER to 17.4 cfs Drop 4-18-07 For Jack & Dan.xl s	Microsoft Excel Worksh	715,264	R
OUT OF PRIORITY WATER 40 OF 80 YEARS.xls	Microsoft Excel Worksh	479,744	R
Portneuf River and Topaz Gage 2-23-07 2nd time.ht m	HTML Document	1,788	R
Portneuf River and Topaz Gage 2-23-07.htm	HTML Document	1,788	R
Prior to 9-15 water letter rev1.doc	Microsoft Word Docume	28,672	R
SUMMARY OF DATA WORKSHEETS 4-26-07 DON. xls	Microsoft Excel Worksh	1,753,088	R
15 files; 3,303,667 bytes			

D:\Data

0 files; 0 bytes

D:\Data\Flow Data

Portneuf flow data.xlsx	Microsoft Office Excel 2	3,331,292	R
1 file; 3,331,292 bytes			

D:\Data\Water Right

Export_Output_pou.txt	Text Document	22,662	R
Portneuf_PODs_(upstream_McCammon)2.xls	Microsoft Excel Worksh	52,736	R
PWUA Water Rights.xlsx	Microsoft Office Excel 2	14,784	R
schema.ini	Configuration Settings	1,008	R
4 files; 91,190 bytes			

D:\Data\Water Right\McCammon Water Right pdfs

29-13311.pdf	Adobe Acrobat Docume	21,953	R
29-13317.pdf	Adobe Acrobat Docume	21,899	R
29-57.pdf	Adobe Acrobat Docume	22,517	R
29-58.pdf	Adobe Acrobat Docume	22,531	R
4 files; 88,900 bytes			

Directory of D:*.*

Page 3 06/21/2011 1:13pm Sorted by Long Name

Levels: All Disk Label: 110616_1246

D:\Data\Water Right\PMVCC Water Right pdfs

29-53.pdf	Adobe Acrobat Docume	24,134	R
29-589.pdf	Adobe Acrobat Docume	23,170	R

2 files; 47,304 bytes

D:\Data\Water Right\PWUA Water Right asp files

29-147.asp.htm	HTML Document	5,100	R
29-180.asp.htm	HTML Document	6,600	R
29-36.asp.htm	HTML Document	6,453	R
29-37.asp.htm	HTML Document	6,507	R
29-47A.asp.htm	HTML Document	5,307	R
29-48.asp.htm	HTML Document	5,081	R
29-53.asp.htm	HTML Document	7,459	R

7 files; 42,507 bytes

D:\Data\Water Right\PWUA Water Right asp files\29-147.asp_files

pm.js	JScript Script File	6,581	R
-------	---------------------	-------	---

1 file; 6,581 bytes

D:\Data\Water Right\PWUA Water Right asp files\29-180.asp_files

pm.js	JScript Script File	6,581	R
-------	---------------------	-------	---

1 file; 6,581 bytes

D:\Data\Water Right\PWUA Water Right asp files\29-36.asp_files

pm.js	JScript Script File	6,581	R
-------	---------------------	-------	---

1 file; 6,581 bytes

D:\Data\Water Right\PWUA Water Right asp files\29-37.asp_files

pm.js	JScript Script File	6,581	R
-------	---------------------	-------	---

1 file; 6,581 bytes

D:\Data\Water Right\PWUA Water Right asp files\29-47A.asp_files

pm.js	JScript Script File	6,581	R
-------	---------------------	-------	---

1 file; 6,581 bytes

D:\Data\Water Right\PWUA Water Right asp files\29-48.asp_files

pm.js	JScript Script File	6,581	R
-------	---------------------	-------	---

1 file; 6,581 bytes

D:\Data\Water Right\PWUA Water Right asp files\29-53.asp_files

pm.js	JScript Script File	6,581	R
-------	---------------------	-------	---

1 file; 6,581 bytes

D:\Data\Water Right\PWUA Water Right pdfs

PET00004

Directory of D:*.*

Page 4 06/21/2011 1:13pm Sorted by Long Name

Levels: All Disk Label: 110616_1246

29-11773.pdf	Adobe Acrobat Docume	22,168	R
29-147.pdf	Adobe Acrobat Docume	21,188	R
29-180.pdf	Adobe Acrobat Docume	23,433	R
29-36.pdf	Adobe Acrobat Docume	21,400	R
29-37.pdf	Adobe Acrobat Docume	22,853	R
29-4467.pdf	Adobe Acrobat Docume	21,903	R
29-47A.pdf	Adobe Acrobat Docume	19,476	R
29-48.pdf	Adobe Acrobat Docume	19,209	R

8 files; 171,630 bytes**D:\Data\Water Use Projections**

Pocatello Water Demand.xlsx	Microsoft Office Excel 2	10,861	R
-----------------------------	--------------------------	--------	---

1 file; 10,861 bytes**D:\Docs****0 files; 0 bytes****D:\Docs\Correspondence**

PWUA Concerns memo (orig w notes).docx	Microsoft Office Word 2	2,032,521	R
PWUA Concerns memo.docx	Microsoft Office Word 2	46,281	R
S Campbell (1-6-2011).docx	Microsoft Office Word 2	27,615	R
TC Greg Sullivan 2-7-2011.docx	Microsoft Office Word 2	26,880	R

4 files; 2,133,297 bytes**D:\Docs\Figures**

Downey Canal & Portneuf Valley (report).jpg Size: 1700x2200; DPI: 200	JPEG Image	1,766,995	R
PMVCC POU (report).jpg Size: 1700x2200; DPI: 200	JPEG Image	2,075,789	R
PMVCC WRs (report).jpg Size: 1700x2200; DPI: 200	JPEG Image	793,168	R
PMVCC WRs (report).pdf	Adobe Acrobat Docume	704,647	R
Portneuf Water Users PODs-POUs.jpg Size: 1700x2200; DPI: 200	JPEG Image	607,131	R
PWUA Water Rights(report).jpg Size: 1700x2200; DPI: 200	JPEG Image	2,243,649	R
Tenmile area water levels.jpg Size: 3193x2006; DPI: 200	JPEG Image	4,514,694	R

7 files; 12,706,073 bytes**D:\Docs\Notes**

Pre-hearing conference notes.docx	Microsoft Office Word 2	28,207	R
Task List.docx	Microsoft Office Word 2	29,037	R

2 files; 57,244 bytes

PET00005

Directory of D:*.*

Page 5 06/21/2011 1:13pm Sorted by Long Name

Levels: All Disk Label: 110616_1246

D:\Docs\Reports

0 files; 0 bytes

D:\Docs\Reports\Final

Attach A PWUA Water Rights.pdf	Adobe Acrobat Docume	70,067	R
Petrich PWUA Report (Final).docx	Microsoft Office Word 2	4,559,672	R
Petrich PWUA Report (Final).pdf	Adobe Acrobat Docume	6,179,059	R

3 files; 10,808,798 bytes

D:\Docs\Reports\Final\Figures_report

0 files; 0 bytes

D:\Docs\Telephone_Logs

TC Chuck Brockway 2-11-11.docx	Microsoft Office Word 2	27,235	R
TC Steve Hebdon.docx	Microsoft Office Word 2	29,439	R

2 files; 56,674 bytes

D:\Email

Portneuf.pst	Office Data File	190,989,312	R
--------------	------------------	-------------	---

1 file; 190,989,312 bytes

D:\GIS

0 files; 0 bytes

D:\GIS\ArcMap_Projects

Chesterfield-Pebble CP.mxd	MXD File	712,704	R
Inkom - Pocatello CP.mxd	MXD File	1,226,240	R
Lava HS - McCammon CP.mxd	MXD File	1,229,312	R
McCammon - Inkom CP.mxd	MXD File	709,120	R
Misc small scale.mxd	MXD File	1,392,640	R
Overview CP.mxd	MXD File	713,216	R
Overview LG.mxd	MXD File	1,467,392	R
Pebble - Lava HS CP.mxd	MXD File	1,400,832	R
PMVCC POU (report).mxd	MXD File	1,312,256	R
PMVCC WRs (report).mxd	MXD File	1,440,256	R
PMVCC WRs.mxd	MXD File	1,548,288	R
PMVCC WRs2.mxd	MXD File	1,635,840	R
Portneuf River water rights.mxd	MXD File	1,346,048	R
Portneuf Valley area (report).mxd	MXD File	1,350,656	R
Portneuf Water Users GIS - CP.mxd	MXD File	1,271,296	R
Portneuf Water Users GIS.mxd	MXD File	5,643,776	R
Portneuf Water Users PODs-POUsD.mxd	MXD File	1,283,584	R
PWUA Water Rights(report).mxd	MXD File	1,374,208	R
PWUA Water Rights.mxd	MXD File	1,563,648	R

PET00006

Directory of D:*.*

Page 6 06/21/2011 1:13pm Sorted by Long Name

Levels: All Disk Label: 110616_1246

Report POD Map (lg)mxd.mxd	MXD File	842,240	R
----------------------------	----------	---------	---

20 files; 29,463,552 bytes

D:\GIS\Figures_nonreport

Portneuf Water Users GIS.jpg	JPEG Image	724,173	R
------------------------------	------------	---------	---

Size: 2125x2750; DPI: 250
1 file; 724,173 bytes

D:\GIS\GPS

CP May 2011 Trackpoints.xlsx	Microsoft Office Excel 2	123,266	R
Waypoints 2011_05_23 Portneuf.dbf	DBF File	26,340	R
Waypoints 2011_05_23 Portneuf.prj	PRJ File	145	R
Waypoints 2011_05_23 Portneuf.shp	SHP File	1,164	R
Waypoints 2011_05_23 Portneuf.shx	SHX File	404	R

5 files; 151,319 bytes

D:\GIS\Property

0 files; 0 bytes

D:\GIS\Shapes

0 files; 0 bytes

D:\GIS\Shapes\Downey Canal

Downey Canal.dbf	DBF File	73	R
Downey Canal.prj	PRJ File	513	R
Downey Canal.sbn	SBN File	132	R
Downey Canal.sbx	SBX File	116	R
Downey Canal.shp	SHP File	956	R
Downey Canal.shx	SHX File	108	R

6 files; 1,898 bytes

D:\GIS\Shapes\Hydrography

Portneuf_River.dbf	DBF File	48,468	R
Portneuf_River.prj	PRJ File	418	R
Portneuf_River.sbn	SBN File	1,012	R
Portneuf_River.sbx	SBX File	180	R
Portneuf_River.shp	SHP File	52,076	R
Portneuf_River.shp.xml	XML Document	159,654	R
Portneuf_River.shx	SHX File	844	R
Selected_Hydrography.dbf	DBF File	23,674	R
Selected_Hydrography.prj	PRJ File	418	R
Selected_Hydrography.sbn	SBN File	564	R
Selected_Hydrography.sbx	SBX File	156	R
Selected_Hydrography.shp	SHP File	31,876	R

PET00007

Directory of D:*.*

Page 7 06/21/2011 1:13pm Sorted by Long Name

Levels: All Disk Label: 110616_1246

Selected_Hydrography.shp.xml	XML Document	159,654	R
Selected_Hydrography.shx	SHX File	452	R
Selected_Hydrography_Lakes.dbf	DBF File	1,018	R
Selected_Hydrography_Lakes.prj	PRJ File	418	R
Selected_Hydrography_Lakes.sbn	SBN File	164	R
Selected_Hydrography_Lakes.sbx	SBX File	124	R
Selected_Hydrography_Lakes.shp	SHP File	51,488	R
Selected_Hydrography_Lakes.shp.xml	XML Document	23,193	R
Selected_Hydrography_Lakes.shx	SHX File	116	R

21 files; 555,967 bytes

D:\GIS\Shapes\Incorporated Areas 2000

Pocatello.dbf	DBF File	482	R
Pocatello.prj	PRJ File	418	R
Pocatello.sbn	SBN File	132	R
Pocatello.sbx	SBX File	116	R
Pocatello.shp	SHP File	2,524	R
Pocatello.shp.xml	XML Document	124,634	R
Pocatello.shx	SHX File	108	R

7 files; 128,414 bytes

D:\GIS\Shapes\PLSS

0 files; 0 bytes

D:\GIS\Shapes\Roads

0 files; 0 bytes

D:\GIS\Shapes\Water Rights

29-147_POU.dbf	DBF File	598	R
29-147_POU.prj	PRJ File	418	R
29-147_POU.sbn	SBN File	132	R
29-147_POU.sbx	SBX File	116	R
29-147_POU.shp	SHP File	1,772	R
29-147_POU.shp.xml	XML Document	133,683	R
29-147_POU.shx	SHX File	108	R
29-180_POU.dbf	DBF File	598	R
29-180_POU.prj	PRJ File	418	R
29-180_POU.sbn	SBN File	132	R
29-180_POU.sbx	SBX File	116	R
29-180_POU.shp	SHP File	2,640	R
29-180_POU.shp.xml	XML Document	133,683	R
29-180_POU.shx	SHX File	108	R
29-36 and_29-37_POU.dbf	DBF File	598	R
29-36 and_29-37_POU.prj	PRJ File	418	R

Directory of D:*.*

Page 8 06/21/2011 1:13pm Sorted by Long Name

Levels: All Disk Label: 110616_1246

29-36 and_29-37_POU.sbn	SBN File	132	R
29-36 and_29-37_POU.sbx	SBX File	116	R
29-36 and_29-37_POU.shp	SHP File	5,644	R
29-36 and_29-37_POU.shp.xml	XML Document	133,683	R
29-36 and_29-37_POU.shx	SHX File	108	R
29-47A_and_29-48_POU.dbf	DBF File	598	R
29-47A_and_29-48_POU.prj	PRJ File	418	R
29-47A_and_29-48_POU.sbn	SBN File	132	R
29-47A_and_29-48_POU.sbx	SBX File	116	R
29-47A_and_29-48_POU.shp	SHP File	4,344	R
29-47A_and_29-48_POU.shp.xml	XML Document	133,683	R
29-47A_and_29-48_POU.shx	SHX File	108	R
29-53_and_29-589_POU.dbf	DBF File	598	R
29-53_and_29-589_POU.prj	PRJ File	418	R
29-53_and_29-589_POU.sbn	SBN File	132	R
29-53_and_29-589_POU.sbx	SBX File	116	R
29-53_and_29-589_POU.shp	SHP File	2,412	R
29-53_and_29-589_POU.shp.xml	XML Document	133,683	R
29-53_and_29-589_POU.shx	SHX File	108	R
29-57,_29-58,_29-13317_29-13317_POD.dbf	DBF File	2,730	R
29-57,_29-58,_29-13317_29-13317_POD.prj	PRJ File	418	R
29-57,_29-58,_29-13317_29-13317_POD.sbn	SBN File	172	R
29-57,_29-58,_29-13317_29-13317_POD.sbx	SBX File	116	R
29-57,_29-58,_29-13317_29-13317_POD.shp	SHP File	212	R
29-57,_29-58,_29-13317_29-13317_POD.shx	SHX File	132	R
29-57,_29-58,_29-13317_29-13317_POU.dbf	DBF File	4,046	R
29-57,_29-58,_29-13317_29-13317_POU.prj	PRJ File	418	R
29-57,_29-58,_29-13317_29-13317_POU.sbn	SBN File	156	R
29-57,_29-58,_29-13317_29-13317_POU.sbx	SBX File	116	R
29-57,_29-58,_29-13317_29-13317_POU.shp	SHP File	16,468	R
29-57,_29-58,_29-13317_29-13317_POU.shx	SHX File	132	R
Dempsey_&_Topaz_POD.dbf	DBF File	1,606	R
Dempsey_&_Topaz_POD.prj	PRJ File	418	R
Dempsey_&_Topaz_POD.sbn	SBN File	156	R
Dempsey_&_Topaz_POD.sbx	SBX File	116	R
Dempsey_&_Topaz_POD.shp	SHP File	156	R
Dempsey_&_Topaz_POD.shx	SHX File	116	R
Portneuf_PODs_(upstream_McCammon).dbf	DBF File	67,922	R
Portneuf_PODs_(upstream_McCammon).prj	PRJ File	418	R
Portneuf_PODs_(upstream_McCammon).sbn	SBN File	1,212	R
Portneuf_PODs_(upstream_McCammon).sbx	SBX File	132	R
Portneuf_PODs_(upstream_McCammon).shp	SHP File	3,460	R
Portneuf_PODs_(upstream_McCammon).shx	SHX File	1,060	R
Portneuf_PODs_(upstream_McCammon).xls	Microsoft Excel Worksh	43,008	R

Directory of D:*.*

Page 9 06/21/2011 1:13pm Sorted by Long Name

Levels: All Disk Label: 110616_1246

Portneuf_PODs_(upstream_McCammon)2.xls Microsoft Excel Worksh 33,792 R
61 files; 870,775 bytes

D:\GIS\Water_Rights

_DocslImages_2401_.ZIP	WinZip File	2,466	R
29_53_pod.dbf	DBF File	2,159	R
29_53_pod.prj	PRJ File	418	R
29_53_pod.sbn	SBN File	172	R
29_53_pod.sbx	SBX File	124	R
29_53_pod.shp	SHP File	184	R
29_53_pod.shx	SHX File	124	R
dempseyic.dbf	DBF File	570	R
dempseyic.prj	PRJ File	429	R
dempseyic.sbn	SBN File	132	R
dempseyic.sbx	SBX File	116	R
dempseyic.shp	SHP File	3,780	R
dempseyic.shx	SHX File	108	R
Export_Output.txt	Text Document	38,851	R
McCammon_Ditch_to_Pocatello_wrpod.dbf	DBF File	128,493	R
McCammon_Ditch_to_Pocatello_wrpod.prj	PRJ File	418	R
McCammon_Ditch_to_Pocatello_wrpod.sbn	SBN File	2,212	R
McCammon_Ditch_to_Pocatello_wrpod.sbx	SBX File	164	R
McCammon_Ditch_to_Pocatello_wrpod.shp	SHP File	6,512	R
McCammon_Ditch_to_Pocatello_wrpod.shx	SHX File	1,932	R
McCammon_Ditch_to_Pocatello_wrpou.dbf	DBF File	95,626	R
McCammon_Ditch_to_Pocatello_wrpou.prj	PRJ File	418	R
McCammon_Ditch_to_Pocatello_wrpou.sbn	SBN File	1,172	R
McCammon_Ditch_to_Pocatello_wrpou.sbx	SBX File	180	R
McCammon_Ditch_to_Pocatello_wrpou.shp	SHP File	105,288	R
McCammon_Ditch_to_Pocatello_wrpou.shx	SHX File	988	R
PMVCC_pod_users.dbf	DBF File	4,978	R
PMVCC_pod_users.prj	PRJ File	418	R
PMVCC_pod_users.sbn	SBN File	204	R
PMVCC_pod_users.sbx	SBX File	116	R
PMVCC_pod_users.shp	SHP File	324	R
PMVCC_pod_users.shx	SHX File	164	R
PMVCC_pod_users.xls	Microsoft Excel Worksh	13,824	R
schema.ini	Configuration Settings	451	R
Selected_PODs.dbf	DBF File	8,350	R
Selected_PODs.prj	PRJ File	418	R
Selected_PODs.sbn	SBN File	260	R
Selected_PODs.sbx	SBX File	124	R
Selected_PODs.shp	SHP File	492	R
Selected_PODs.shx	SHX File	212	R

Directory of D:*.***Page 10 06/21/2011 1:13pm Sorted by Long Name****Levels: All Disk Label: 110616_1246**

Selected_POUs.dbf	DBF File	7,482	R
Selected_POUs.prj	PRJ File	418	R
Selected_POUs.sbn	SBN File	220	R
Selected_POUs.sbx	SBX File	132	R
Selected_POUs.shp	SHP File	28,512	R
Selected_POUs.shx	SHX File	164	R
Topaz_&_Dempsey_Co_Boundaries.dbf	DBF File	906	R
Topaz_&_Dempsey_Co_Boundaries.prj	PRJ File	418	R
Topaz_&_Dempsey_Co_Boundaries.sbn	SBN File	140	R
Topaz_&_Dempsey_Co_Boundaries.sbx	SBX File	116	R
Topaz_&_Dempsey_Co_Boundaries.shp	SHP File	8,184	R
Topaz_&_Dempsey_Co_Boundaries.shp.xml	XML Document	133,683	R
Topaz_&_Dempsey_Co_Boundaries.shx	SHX File	116	R

53 files; 603,862 bytes**D:\GIS\Wells****0 files; 0 bytes****D:\Legal**

2011-02-16 Affidavit of Brent Orton.pdf	Adobe Acrobat Docume	1,499,539	R
IC Section 42-222.htm	HTML Document	30,508	R
IC Section 42-222.pdf	Adobe Acrobat Docume	60,145	R
Pocatello response to PWUA 2nd discovery.pdf	Adobe Acrobat Docume	921,745	R
Pocatello response to PWUA first discovery.pdf	Adobe Acrobat Docume	324,127	R

5 files; 2,836,064 bytes**D:\Resources**

Addendum agreement to Portneuf River Settlement.pdf	Adobe Acrobat Docume	83,229	R
Geochemistry Portneuf Aquifer – recharge and contamination (Meehan).pdf	Adobe Acrobat Docume	5,639,179	R
JackABarnettAffidavit.pdf	Adobe Acrobat Docume	906,781	R
Loss Estimates of PMVCC Canal (signed) 2011 Feb .pdf	Adobe Acrobat Docume	48,030	R
NRCS approved project (includes Portnefu) 8-14-09.pdf	Adobe Acrobat Docume	56,893	R
NRCS funding news release 7-31-2009.html	HTML Document	15,132	R
Orientation of stream feds to Portneuf RiverbyIASCD.doc	Microsoft Word Docume	452,608	R
Portneuf Wtr Resource Recon (Norvitch Larsen 1970).pdf	Adobe Acrobat Docume	9,093,812	R
Surface ground water interaction in Portnefu River Valley (USGS).pdf	Adobe Acrobat Docume	2,977,753	R
Water Balance in Pumping Capacity (Welhan - 2001).pdf	Adobe Acrobat Docume	499,013	R

10 files; 19,772,428 bytes**PET00011**

Directory of D:*.*

Page 11 06/21/2011 1:13pm Sorted by Long Name

Levels: All Disk Label: 110616_1246

D:\Resources\NRCS funding news release 7-31-2009_files

button_go.gif	GIF Image	813	R
Size: 32x19; DPI: 96			
footer_uparrow.gif	GIF Image	53	R
Size: 11x8; DPI: 96			
header_nracs.gif	GIF Image	2,727	R
Size: 314x47; DPI: 96			
header_photo.jpg	JPEG Image	4,981	R
Size: 202x70; DPI: 100			
header_sitename.gif	GIF Image	634	R
Size: 152x70; DPI: 96			
header_usda.gif	GIF Image	880	R
Size: 314x13; DPI: 96			
irrigation.jpg	JPEG Image	81,014	R
Size: 504x360; DPI: 300			
largetext.css	Cascading Style Sheet D	1,043	R
mediumtext.css	Cascading Style Sheet D	1,043	R
nav_largetext.gif	GIF Image	73	R
Size: 17x17; DPI: 96			
nav_medtext.gif	GIF Image	70	R
Size: 17x17; DPI: 96			
nav_sep.gif	GIF Image	88	R
Size: 1x17; DPI: 96			
nav_smalltext.gif	GIF Image	113	R
Size: 17x17; DPI: 96			
nracs.js	JScript Script File	2,339	R
spacer.gif	GIF Image	43	R
Size: 1x1; DPI: 96			
style.css	Cascading Style Sheet D	6,319	R

16 files; 102,233 bytes**D:\Resources\Pocatello Water Master Plan**

Master Plan Cover - back.pdf	Adobe Acrobat Docume	17,483	R
Master Plan Cover.pdf	Adobe Acrobat Docume	237,022	R
Master Plan Section 01 - Introduction .pdf	Adobe Acrobat Docume	39,735	R
Master Plan Section 02 - Existing Water System Description.pdf	Adobe Acrobat Docume	3,252,993	R
Master Plan Section 03 - Service Area Char.pdf	Adobe Acrobat Docume	12,105,104	R
Master Plan Section 04 - Groundwater Evaluation.pdf	Adobe Acrobat Docume	571,195	R
Master Plan Section 05 - Planning & Criteria.pdf	Adobe Acrobat Docume	84,819	R
Master Plan Section 06 - Water Requirements.pdf	Adobe Acrobat Docume	132,138	R
Master Plan Section 07 - Regulatory Evaluation.pdf	Adobe Acrobat Docume	133,470	R
Master Plan Section 08 - Model Development.pdf	Adobe Acrobat Docume	84,001	R
Master Plan Section 09 - Existing System Evaluation.pdf	Adobe Acrobat Docume	1,828,879	R

PET00012

Directory of D:*.*

Page 12 06/21/2011 1:13pm Sorted by Long Name

Levels: All Disk Label: 110616_1246

Master Plan Section 10 - Future System Analysis.pdf	Adobe Acrobat Docume	1,242,512	R
Master Plan Section 11 - Development and Screening of Water Supply Alternatives.pdf	Adobe Acrobat Docume	105,772	R
Master Plan Section 12 - Recommended Plan.pdf	Adobe Acrobat Docume	11,535,854	R
Master Plan Water Distribution System - Schematic Diagram - 6_2005 Rev 2.pdf	Adobe Acrobat Docume	7,228,981	R
Master Plan Water Distribution System Map - Fire Flow Demands (Revised).pdf	Adobe Acrobat Docume	1,247,422	R
Master Plan Water Distribution System Map - Max Daily-Max Headloss.pdf	Adobe Acrobat Docume	421,186	R
Master Plan Water Distribution System Map - Max Daily-Max Pressure.pdf	Adobe Acrobat Docume	791,921	R
Master Plan Water Distribution System Map - Max Daily-Max Velocities.pdf	Adobe Acrobat Docume	421,810	R
Master Plan Water Distribution System Map - Max Daily-Min Pressure.pdf	Adobe Acrobat Docume	783,755	R

20 files; 42,266,052 bytes

Total: 50 directories; 315 files; 382,717,408 bytes

PET00013

Directory of D:*.*

Page 1 06/21/2011 3:44pm Sorted by Long Name

Levels: All Disk Label: 110616_1253

D:\

0 files; 0 bytes

D:\Photos

Greg Sullivan photos.pdf	Adobe Acrobat Docume	26,166,661
PWUA 1988 flooding pic.jpg	JPEG Image	159,672
PWUA photos.pdf	Adobe Acrobat Docume	3,224,259

3 files; 29,550,592 bytes

D:\Photos\Petrich

P5170001.JPG	JPEG Image	3,297,213
P5190002.JPG	JPEG Image	2,942,436
P5190003.JPG	JPEG Image	3,273,188
P5190004.JPG	JPEG Image	3,388,357
P5190005.JPG	JPEG Image	3,316,282
P5190006.JPG	JPEG Image	3,297,510
P5190007.JPG	JPEG Image	3,299,991
P5190008.JPG	JPEG Image	3,287,694
P5190009.JPG	JPEG Image	3,247,715
P5190010.JPG	JPEG Image	2,868,685
P5190011.JPG	JPEG Image	2,979,844
P5190012.JPG	JPEG Image	3,231,014
P5190013.JPG	JPEG Image	3,252,538
P5190014.JPG	JPEG Image	3,153,679
P5190015.JPG	JPEG Image	3,008,258
P5190016.JPG	JPEG Image	3,293,076
P5190017.JPG	JPEG Image	3,236,414
P5190018.JPG	JPEG Image	3,270,720
P5190019.JPG	JPEG Image	3,269,910
P5190020.JPG	JPEG Image	3,288,234
P5190021.JPG	JPEG Image	3,410,196
P5190022.JPG	JPEG Image	3,264,259
P5190023.JPG	JPEG Image	5,290,414
P5190024.JPG	JPEG Image	3,238,972
P5190025.JPG	JPEG Image	3,223,840
P5190026.JPG	JPEG Image	3,299,888
P5190027.JPG	JPEG Image	3,322,562
P5190028.JPG	JPEG Image	3,327,918
P5190029.JPG	JPEG Image	3,319,526
P5190030.JPG	JPEG Image	2,505,656
P5190031.JPG	JPEG Image	3,347,672
P5190032.JPG	JPEG Image	3,222,199
P5190033.JPG	JPEG Image	3,249,881
P5190034.JPG	JPEG Image	2,879,276

PET00015

Directory of D:*.*

Page 2 06/21/2011 3:45pm Sorted by Long Name

Levels: All Disk Label: 110616_1253

P5190035.JPG	JPEG Image	3,241,495
P5190036.JPG	JPEG Image	3,154,008
P5190037.JPG	JPEG Image	2,471,611
P5190038.JPG	JPEG Image	2,874,463
P5190039.JPG	JPEG Image	2,321,746
P5190040.JPG	JPEG Image	3,502,381
P5190041.JPG	JPEG Image	3,251,593
P5190042.JPG	JPEG Image	3,104,398
P5190043.JPG	JPEG Image	3,191,548
P5190044.JPG	JPEG Image	3,318,634
P5190045.JPG	JPEG Image	3,259,576
P5190046.JPG	JPEG Image	3,284,748
P5190047.JPG	JPEG Image	3,992,587
P5190048.JPG	JPEG Image	3,249,988
P5190049.JPG	JPEG Image	3,232,179
P5190050.JPG	JPEG Image	3,333,634
P5190051.JPG	JPEG Image	4,986,238
P5190052.JPG	JPEG Image	4,151,321
P5190053.JPG	JPEG Image	5,332,587
P5190054.JPG	JPEG Image	4,148,636
P5190055.JPG	JPEG Image	4,675,213
P5190056.JPG	JPEG Image	3,542,549
P5190057.JPG	JPEG Image	4,649,877
P5190058.JPG	JPEG Image	3,397,546
P5190059.JPG	JPEG Image	3,245,845
P5190060.JPG	JPEG Image	3,301,766
P5190061.JPG	JPEG Image	3,132,312
P5190062.JPG	JPEG Image	3,118,903
P5190063.JPG	JPEG Image	3,260,137
P5190064.JPG	JPEG Image	2,912,961
P5190065.JPG	JPEG Image	2,944,015
P5190066.JPG	JPEG Image	5,091,396
P5190067.JPG	JPEG Image	3,298,061
P5190068.JPG	JPEG Image	3,151,737
P5190069.JPG	JPEG Image	3,180,935
P5190070.JPG	JPEG Image	3,321,783
P5190071.JPG	JPEG Image	3,287,409
P5190072.JPG	JPEG Image	3,295,949
P5190073.JPG	JPEG Image	3,200,899
P5190074.JPG	JPEG Image	3,246,067
P5190075.JPG	JPEG Image	3,322,523
P5190076.JPG	JPEG Image	4,179,256
P5190077.JPG	JPEG Image	3,451,592
P5190078.JPG	JPEG Image	3,256,244

PET00016

Directory of D:*.*

Page 3 06/21/2011 3:45pm Sorted by Long Name

Levels: All Disk Label: 110616_1253

P5190079.JPG	JPEG Image	2,379,169
P5190080.JPG	JPEG Image	3,197,726
P5190081.JPG	JPEG Image	2,945,822
P5190082.JPG	JPEG Image	3,191,444
P5190083.JPG	JPEG Image	2,759,081
P5190084.JPG	JPEG Image	3,252,786
P5190085.JPG	JPEG Image	3,197,328
P5190086.JPG	JPEG Image	3,316,828
P5190087.JPG	JPEG Image	2,964,520
P5190088.JPG	JPEG Image	3,202,674
P5190089.JPG	JPEG Image	3,222,344
P5190090.JPG	JPEG Image	3,254,189
P5190091.JPG	JPEG Image	3,262,365
P5190092.JPG	JPEG Image	3,359,605
P5190093.JPG	JPEG Image	3,290,081
P5190094.JPG	JPEG Image	3,272,499
P5190095.JPG	JPEG Image	3,277,593
P5190096.JPG	JPEG Image	3,241,698
P5190097.JPG	JPEG Image	3,228,738
P5190098.JPG	JPEG Image	3,223,463
P5190099.JPG	JPEG Image	3,363,599
P5190100.JPG	JPEG Image	3,429,993
P5190101.JPG	JPEG Image	3,693,921
P5190102.JPG	JPEG Image	3,460,545
P5200103.JPG	JPEG Image	3,299,245
P5200104.JPG	JPEG Image	3,257,917
P5200105.JPG	JPEG Image	3,749,043
P5200106.JPG	JPEG Image	4,003,427
P5200107.JPG	JPEG Image	3,248,658
P5200108.JPG	JPEG Image	3,215,400
P5200109.JPG	JPEG Image	3,212,550
P5200110.JPG	JPEG Image	3,240,234
P5200111.JPG	JPEG Image	3,201,497
P5200112.JPG	JPEG Image	3,261,948
P5200113.JPG	JPEG Image	3,271,768
P5200114.JPG	JPEG Image	3,340,138
P5200115.JPG	JPEG Image	3,272,118
P5200116.JPG	JPEG Image	3,276,077
P5200117.JPG	JPEG Image	3,230,527
P5200118.JPG	JPEG Image	3,276,161
P5200119.JPG	JPEG Image	3,256,220
P5200120.JPG	JPEG Image	3,418,692
P5200121.JPG	JPEG Image	3,259,806
P5200122.JPG	JPEG Image	3,281,986

PET00017

Directory of D:*.*

Page 4 06/21/2011 3:45pm Sorted by Long Name

Levels: All Disk Label: 110616_1253

P5200123.JPG	JPEG Image	3,319,232
P5200124.JPG	JPEG Image	3,331,706
P5200125.JPG	JPEG Image	3,292,531
Thumbs.db	Data Base File	415,744

126 files; 417,893,984 bytes

Total: 3 directories; 129 files; 447,444,576 bytes

Moffatt Thomas

MOFFATT THOMAS BARRETT ROCK & FIELDS, CHTD.

Boise
Idaho Falls
Pocatello

John W. Barrett
Richard C. Fields
John S. Simko
John C. Ward
D. James Manning
David B. Lincoln
Gary T. Dance
Larry C. Hunter
Randall A. Peterman
Mark S. Prusynski
Stephen R. Thomas
Glenna M. Christensen
Gerald T. Husch
Scott L. Campbell
Robert B. Burns
Michael E. Thomas

Patricia M. Olsson
Christine B. Nicholas
Bradley J. Williams
Lee Radford
Michael O. Roe
Nancy J. Garrett
David S. Jensen
James L. Martin
C. Clayton Gill
Michael W. McGreaham
David P. Gardner
Julian E. Gabiola
Kimberly D. Evans Ross
Jon A. Stenquist
Mark C. Peterson
Tyler J. Anderson

Andrew J. Waldera
Dylan B. Lawrence
Paul D. McFarlane
C. Edward Cather III
Benjamin C. Ritchie
Noah G. Hillen
Matthew J. McGee
David J. Dance
Mindy M. Willman

Robert E. Bakes, of counsel

Willis C. Moffatt, 1907-1980
Eugene C. Thomas, 1931-2010
Kirk R. Helvie, 1936-2003

May 9, 2011

US Bank Plaza Building
101 S Capitol Blvd 10th Fl
PO Box 829
Boise Idaho 83701 0829

208 345 2000
800 422 2889
208 385 5384 Fax
www.moffatt.com

PRIVILEGED AND CONFIDENTIAL ATTORNEY WORK PRODUCT ATTORNEY-CLIENT PRIVILEGE

Christian R. Petrich, Ph.D., P.E.
SPF Water Engineering LLC
300 East Mallard Drive, Suite 350
Boise, ID 83706

Re: Portneuf Watershed Users Association / Transfer No. 76779
Confidential Litigation Memo
MTBR&F File No. 24810.0000

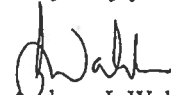
Dear Christian:

Enclosed you will find the litigation memorandum you and I discussed over the phone this afternoon. Again, the memorandum is in e-mail format, but is being sent via U.S. Mail as a means of controlling the ease of its dissemination for privilege purposes.

Please review the memorandum sooner rather than later. The memorandum presents our view of Pocatello's proposed transfer application as it currently stands. If you have any questions or comments regarding the memorandum, please let me know. Unless you raise expert/technical disagreement with the contents of the memorandum, the memorandum will largely serve as a roadmap/plan for challenging the proposed transfer.

Thank you for your attention to this matter.

Very truly yours,



Andrew J. Waldera

AJW/dll
Enclosure

Client:2042126.1

PET00019

Regarding river losses/gains: see page 17, Table 6-2,
Section 6.2.3, per John Welhan e-mail of 2/8/2011

**Geochemistry of the Southern Pocatello Aquifer and its Implications for
Aquifer Recharge and Contamination Potential**

By:

Christopher W. Meehan

A thesis submitted in partial fulfillment of
the requirements for the degree of

Master of Science
in
Geology

Idaho State University
2005

Executive Summary

The Lower Portneuf River Valley (LPRV) aquifer supplies the entire water needs to rural and urban residents of Pocatello and Chubbuck, Idaho. Several localized areas of groundwater have been impacted by nitrate and salts such as sulfate, chloride, sodium, and calcium, which are usually associated with non-point sources like septic leachate, road salting, and runoff from agriculture and ranching operations.

Inorganic groundwater chemistry was examined from more than 80 sites in a 40 km² portion of the southern lower Portneuf River valley in an attempt to identify the source(s) of contamination. Water samples were collected during several sampling events over a year-long period from June of 1993 through July of 1994. Many of these locations were re-sampled in the summer of 2004 to evaluate changes in aquifer chemistry that may be due to anthropogenic sources.

Wells producing water from near the water table tend to have higher concentrations of nitrate and salts, suggesting a near-surface source is responsible for impairing groundwater quality. At a vast majority of sample sites, all major ions in groundwater, including nitrate, show strong positive correlations with chloride, suggesting a mixing relationship with a common source is responsible for adding these salts to groundwater.

Proportions of chloride/nitrate and chloride/sulfate in groundwater fall within a very narrow range, consistent with a single source whose chemical signature is similar to septic drain field effluent. The spatial relationship between septic drain fields and groundwater elevated in salts and nitrate also suggests that septic effluent may be the principal factor impairing groundwater quality in the LPRV.

On the order of 670 septic drain fields lie on or adjacent the municipal aquifer, the vast majority of them situated above the aquifer's principal recharge areas, potentially discharging more than 50 million gallons of septic effluent annually. Models were created to evaluate the mixing relationships in the southern portion of the municipal aquifer between septic effluent, background recharge water, and potential river leakage from the Portneuf River. The models demonstrated that at concentrations of nitrate typical of septic effluent (30-60 mg/l), the cumulative impact of septic fields could reproduce the observed trends of increasing nitrate and chloride concentrations along the flowpath of the aquifer.

Similar mixing models were also constructed for Mink Creek, a tributary drainage whose principal potential source of ground water contamination is only septic effluent. The Mink Creek models also verified that impacts from septic fields account for the trends observed in groundwater.

This study represents the first systematic, quantitative evaluation of contaminant sources in the LPRV aquifer that are responsible for observed major ion trends. It provides the strongest evidence to date for the cumulative impact that septic sewage is exerting on this aquifer, even where dilution by ground water flow is so effective. With further refinement, the mixing models can be used as a management tool to predict future water quality trends under various development scenarios involving septic-based growth in the southern LPRV and evaluating emerging septic technologies.

In presenting this thesis in partial fulfillment of the requirements for an advanced degree at Idaho State University, I agree that the university library may make it freely available for inspection to the public. I further state that permission for extensive copying of this thesis for scholarly purpose may be granted by the Dean of Graduate Studies and Research, Dean of Arts and Sciences, or by the University Librarian. It is understood that any copying or publication of any portion of this thesis for financial gain shall not be allowed without my written permission.

Signature _____

Date _____

**Geochemistry of the Southern Pocatello Aquifer and its Implications for
Aquifer Recharge and Contamination Potential**

By:

Christopher W. Meehan

A thesis submitted in partial fulfillment of
the requirements for the degree of

Master of Science
in
Geology

Idaho State University
2005

To the Graduate Faculty:

The members of the committee appointed to examine the thesis of Christopher W. Meehan find it satisfactory and recommend that it be accepted.

Dr. John Welhan, Major Advisor

Dr. Paul Link, Advisor

Dr. Robert Van Kirk, Graduate Faculty
Representative

Acknowledgements

I would like to thank Dr. Scott Hughes and the entire ISU Geosciences Department faculty for allowing me the opportunity to return after such a long period of time to complete this thesis. Most importantly, I would like to extend my appreciation to Dr. John Welhan for his confidence in me and his ability to inspire and me along the way. I hope this thesis bears witness to his wisdom. I would also like to thank Dr. Paul Link and Dr. Robert Van Kirk for being on my committee and for their guidance and contributions.

Thank you to my mother and family for their encouragement and faith in me. Special thanks to Todd and Kyla for sending me pictures from home, which motivated me each day to complete my work so that I could be with them as soon as possible.

In addition, I would like to thank Melissa Neiers, it was nice to have a friend here from the past.

Lastly, I would like to dedicate this thesis to my friend Mason Estes. We made a bet ten years ago as to who would finish first. I win. I hope you found peace brother.

Table of Contents

List of Figures.....	vii
List of Tables	ix
List of Acronyms and Abbreviations.....	x
Abstract.....	xi
Executive Summary	xiii
Chapter 1: Introduction.....	1
1.1 Problem Statement.....	1
1.2 Scope and Objectives.....	3
1.3 Location	5
1.4 Geology.....	7
1.4.1 Structural and Topographic Development of the LPRV.....	7
1.4.2 Aquifer Development.....	11
1.5 Hydrogeology	13
1.5.1 Sources of Information	13
1.5.2 Subsystems and the Southern Aquifer	13
1.5.3 Water Balance.....	17
1.6 Water Quality.....	22
1.7 Management Issues.....	25
Chapter 2: Methodology	28
2.1 Selection of Historical Data	28
2.2 Selection of Sample Locations.....	29
2.2.1 1993-1994 Sampling.....	29
2.2.2 2004 Sampling	30
2.3 Sampling Methods	33
2.4 Laboratory Methods and Quality Control.....	33
2.5 Chemical Mass Balance Modeling	38
2.6 Septic Mixing Models.....	39
2.7 Variogram Methods	41
Chapter 3: 1993 – 1994 Water Quality Results	44
3.1 Major Ion Chemical Composition	44
3.2 Water Quality Trends.....	47
3.3 Evidence for Contaminant and Recharge Contributions to the Southern Aquifer	51
3.4 Seasonal Variations in Water Chemistry	54
Chapter 4: 2004 Water Quality Results	60
4.1 2004 Water Quality Trends.....	60
4.2 Evidence for Mixing in the Southern Aquifer	63
4.3 Mink Creek Water Quality.....	65
4.3.1 Autocorrelation of Mink Creek and Main Aquifer Chemistry	68

4.4 Temporal Changes in Water Quality from 1994 – 2004.....	70
4.4.1 Southern Aquifer and Groundwater South of the Portneuf Gap.....	70
4.4.2 West Bench and Mink Creek Groundwater	73
4.4.3 Eastern Aquifer Groundwater.....	74
Chapter 5: Discussion	76
5.1 Temporal Trends in the Southern Aquifer	76
5.1.1 Decreasing Salt Trends	76
5.1.2 Eastern Aquifer Impacts and Trends.....	79
5.2 Spatial Relationships.....	81
5.2.1 Summary of Spatial Distribution of Water Types	81
5.2.2 Spatial Changes Along the Flow Path of the Southern Aquifer	83
5.2.3 Evidence for a Near-Surface Source of Salt Contamination	85
5.3 Sources of Salt Contamination.....	86
5.3.1 Least Likely Sources of Salt Contamination	87
5.3.2 Most Likely Sources of Salt Contamination.....	92
5.4 Summary of Source Evaluation.....	100
Chapter 6: Modeling	102
6.1 Simple Chemical Mass Balance Modeling.....	102
6.2 Segmented Mixing Models.....	105
6.2.1 Model A: Incremental Increase in Aquifer Flux.....	108
6.2.2 Model B: West Bench Groundwater Contributions.....	110
6.2.3 Model C: Portneuf River Contributions.....	114
6.2.4 Discussion of Main Aquifer Septic Mixing Models.....	120
6.3 Mink Creek Septic Mixing Models.....	121
6.3.1 Discussion of Mink Creek Models	126
Chapter 7: Summary and Conclusions.....	128
7.1 Summary.....	128
7.2 Conclusions.....	132
7.3 Recommendations for Further Study.....	133
References.....	135
Appendix A: Database of LPRV Inorganic Groundwater Chemistry	143
Appendix B: Analytical Results for Water Samples Collected in 1993-94 and 2004.....	144
Appendix C: Updated Southern Pocatello Water Balance, 1994 to 2003	157
Appendix D: Impact of Leachable Sulfate on the Quality of Groundwater in the Pocatello Aquifer.....	165
Appendix E: Autocorrelation of Mink Creek and Southern Aquifer Groundwater Chemistry.....	182
Appendix F: Potential Sources of Salt Contamination to Groundwater	189

List of Figures

Figure 1-1: Location Map for the Lower Portneuf River Valley.....	6
Figure 1-2: East-west cross section through the southern Pocatello valley.....	9
Figure 1-3: Geographic boundaries defining the northern, southern and eastern Pocatello aquifers.....	15
Figure 1-4: Sources of recharge to the southern Pocatello aquifer.....	18
Figure 1-5: Temporal trends in chloride concentrations in Pocatello municipal production wells.....	23
Figure 1-6: Correlation of major ionic constituents in municipal well water.....	23
Figure 1-7: Plume of TCE contamination in the Pocatello aquifer.....	26
Figure 2-1: Locations of water samples collected in 1993 and 1994.....	30
Figure 2-2: Locations of water samples collected in 2004.....	31
Figure 2-3: Inferred locations of septic tanks in the lower Portneuf River valley.....	32
Figures 2-4 (a) (b) and (c): Sample pairs of the HACH analyses and IAS lab analyses for Cl, SO ₄ and NO ₃	37
Figure 2-5: Boundaries of cells used for septic mixing models.....	40
Figure 2-6: Flow diagram of third modeling scenario for septic mixing models.....	41
Figure 2-7: General form of spatially correlated semivariogram.....	42
Figure 3-1: Locations of water samples collected in 1993 and 1994.....	45
Figure 3-2: Piper plot showing the composition of major ions in water samples.....	46
Figure 3-3: Stiff diagrams displayed by locations of water types.....	47
Figures 3-4 (a) through (e): Plots showing correlations of hardness (CaCO ₃), Ca, Mg, Na, Si and their relationship to Cl.....	49
Figures 3-5(a) and (b): Plots showing correlations of chloride with sulfate and nitrate....	50

Figure 3-6: Histograms of Cl, SO ₄ , and NO ₃ concentrations for samples collected in the southern aquifer in May through July of 1994.....	52
Figure 3-7(a) and (b): Cross plots of chloride with sulfate and nitrate for all samples collected in the southern aquifer in 1993 and 1994.....	52
Figure 3-8: Map of the southern aquifer showing areas where groundwater samples in 1993 and 1994 indicated impacts from salt contamination.....	53
Figures 3-9(a) through (j): Time series plots depicting variations in concentrations of Cl, SO ₄ , and NO ₃ in aquifer recharge sources in 1993 and 1994.....	56
Figures 3-10(a) through (c): Time series plots depicting variations in concentrations of Cl, SO ₄ , and NO ₃ in the southern aquifer in 1993 and 1994.....	57
Figure 3-11: Depths to groundwater measured in wells located in the southern aquifer, south of the Portneuf Gap and Mink Creek.....	58
Figure 4-1: Locations of water samples collected in 2004.....	61
Figures 4-2(a) and (b): Correlations of chloride with sulfate and nitrate for water samples collected in 2004.....	62
Figures 4-3(a) and (b): Cross plots of chloride with sulfate and nitrate respectively for all samples collected in the southern aquifer in 2004.....	63
Figure 4-4: Locations of the thirty seven groundwater samples and two surface water samples collected in the Mink Creek drainage in 2004.....	66
Figures 4-5(a), (b), and (c): Concentrations of CL, SO ₄ , and NO ₃ in water samples collected in the Mink Creek drainage.....	67
Figures 4-6(a), (b), and (c): Changes in groundwater chemistry from 1994 to 2004 displayed for each location's spatial distance from the Portneuf Gap.....	72
Figures 4-7 (a) through (f): Changes in groundwater chemistry from 1994 to 2004 in West Bench and Mink Creek groundwater.....	75
Figures 5-1 (a) through (f): Temporal trends of Cl, SO ₄ , and NO ₃ in wells in the southern aquifer.....	77
Figure 5-2: Temporal changes in NO ₃ concentrations in well DEQ-83, located in the eastern aquifer.....	80
Figure 5-3: Spatial distribution of groundwater chemistry.....	82

Figures 5-4(a) through (c): Concentrations of Cl, SO ₄ , and NO ₃ in the southern aquifer in 2004.....	84
Figure 6-1: Southern aquifer cell boundaries for septic mixing models.....	106
Figure 6-2: Septic mixing model A flow diagram.....	108
Figures 6-3(a) and (b): Results of Septic Mixing Model A for a total septic flux of 49 million gallons per year.....	109
Figures 6-4(a) and (b): Matching the trend predicted by the model the observed trend.....	109
Figure 6-5: Septic mixing model B flow diagram.....	111
Figures 6-6(a) and (b): Results of mixing calculations incorporating West Bench recharge.....	112
Figure 6-7: Septic mixing model C flow diagram.....	114
Figure 6-8: Locations of stream discharge measurements along the Portneuf River.....	115
Figures 6-9(a) and (b): Relationship between Portneuf River water concentrations of Cl, SO ₄ , and NO ₃ and river discharge in Pocatello.....	119
Figures 6-10(a) through (d): Septic mixing model results considering Portneuf River contributions.....	119
Figures 6-11(a) and (b): Comparison of Cl, SO ₄ , and NO ₃ in model results and observed groundwater concentrations.....	120

List of Tables

Table 2-1: Data precision for HACH analyses.....	35
Table 2-2: Standard deviations (analytical precision) as a function of sample dilution effects.....	35
Table 4-1: Best estimates of autocorrelation ranges for Cl, SO ₄ and NO ₃ concentrations in Mink Creek and the main aquifer.....	70
Table 5-1: Concentrations of NO ₃ in septic effluent as determined in five studies.....	97
Table 5-2: Summary of potential fluxes to groundwater and ratios of chloride and nitrate from sources in the southern LPRV.....	100

Table 6-1: Concentrations of tracers used to define end-members in one matrix model and the resulting end-member contributions to the southern aquifer.....	103
Table 6-2: Portneuf River and tributary discharge measurements in the Pocatello Valley.....	115
Table 6-3: Reach gains and losses for the Portneuf River in the Pocatello Valley.....	117
Table 6-4: Parameters used estimate annual groundwater fluxes in Mink Creek mixing models.....	122
Tables 6-5(a) and (b): Results of septic mixing models for the upper section of the Mink Creek drainage above the poultry farm.....	124
Tables 6-6(a) and (b): Results of septic mixing models for the mouth of the Mink Creek drainage.....	125

List of Acronyms and Abbreviations

Bgal: Billion Gallons
 Bgy: Billion gallons per year
 cfs: Cubic feet per second
 Cr: Creek
 EPA: Environmental Protection Agency
 Gpd: Gallons per day
 IDEQ: Idaho Department of Environmental Quality
 ISU: Idaho Sate University
 LPRV: Lower Portneuf River Valley
 MCL: Maximum contaminant level
 mg/l: Milligrams per liter
 Mgy: Million gallons per year
 TDS: Total dissolved solids
 USGS: United States Geological Survey

Abstract

Inorganic groundwater chemistry was examined from more than 80 sites in a 40 km² portion of the southern lower Portneuf River valley (LPRV) in southeast Idaho. Wells producing water from near the water table tend to have higher concentrations of nitrate and salts, suggesting a near-surface anthropogenic source is responsible for impairing groundwater quality. Proportions of chloride/nitrate and chloride/sulfate in groundwater fall within a very narrow range, consistent with a single source whose chemical signature is similar to septic drain field effluent.

On the order of 670 septic drain fields lie on or adjacent to this portion of the aquifer, the vast majority of them situated above the aquifer's principal recharge areas. Statistically significant increases in nitrate and chloride were observed along a 12 km length of the LPRV aquifer (the increase in sulfate concentration along the flow path was not statistically significant). A five-compartment box model was used to evaluate the mixing relationships between septic effluent, background recharge water, potential river leakage, and aquifer throughflow to determine if septic effluent alone can account for the observed concentration increases along the flow path.

The models demonstrated that at concentrations of nitrate typical of septic effluent (30-60 mg/l) and the magnitude of fluxes calculated independently for the aquifer water balance, the cumulative impact of septic systems could reproduce the observed trend of increasing nitrate, chloride and sulfate concentrations along the flowpath of the aquifer. These models also indicate that Portneuf River losses to groundwater are insignificant.

Similar mixing models were also constructed for Mink Creek, a tributary drainage

whose principal potential source of ground water contamination is only septic effluent. The Mink Creek models also verified that impacts from septic fields account for the trends observed in groundwater.

Septic effluent appears to be the principal factor impairing groundwater quality in the LPRV. Other potential sources such as road salt, chemical fertilizers and animal manure appear to have minor impacts on groundwater quality.

Executive Summary

The Lower Portneuf River Valley (LPRV) aquifer supplies the entire water needs to rural and urban residents of Pocatello and Chubbuck, Idaho. Several localized areas of groundwater have been impacted by nitrate and salts such as sulfate, chloride, sodium, and calcium, which are usually associated with non-point sources like septic leachate, road salting, and runoff from agriculture and ranching operations.

Inorganic groundwater chemistry was examined from more than 80 sites in a 40 km² portion of the southern lower Portneuf River valley in an attempt to identify the source(s) of contamination. Water samples were collected during several sampling events over a year-long period from June of 1993 through July of 1994. Many of these locations were re-sampled in the summer of 2004 to evaluate changes in aquifer chemistry that may be due to anthropogenic sources.

Wells producing water from near the water table tend to have higher concentrations of nitrate and salts, suggesting a near-surface source is responsible for impairing groundwater quality. At a vast majority of sample sites, all major ions in groundwater, including nitrate, show strong positive correlations with chloride, suggesting a mixing relationship with a common source is responsible for adding these salts to groundwater.

Proportions of chloride/nitrate and chloride/sulfate in groundwater fall within a very narrow range, consistent with a single source whose chemical signature is similar to septic drain field effluent. The spatial relationship between septic drain fields and groundwater elevated in salts and nitrate also suggests that septic effluent may be the principal factor impairing groundwater quality in the LPRV.

On the order of 670 septic drain fields lie on or adjacent the municipal aquifer, the vast majority of them situated above the aquifer's principal recharge areas, potentially discharging more than 50 million gallons of septic effluent annually. Models were created to evaluate the mixing relationships in the southern portion of the municipal aquifer between septic effluent, background recharge water, and potential river leakage from the Portneuf River. The models demonstrated that at concentrations of nitrate typical of septic effluent (30-60 mg/l), the cumulative impact of septic fields could reproduce the observed trends of increasing nitrate and chloride concentrations along the flowpath of the aquifer.

Similar mixing models were also constructed for Mink Creek, a tributary drainage whose principal potential source of ground water contamination is only septic effluent. The Mink Creek models also verified that impacts from septic fields account for the trends observed in groundwater.

This study represents the first systematic, quantitative evaluation of contaminant sources in the LPRV aquifer that are responsible for observed major ion trends. It provides the strongest evidence to date for the cumulative impact that septic sewage is exerting on this aquifer, even where dilution by ground water flow is so effective. With further refinement, the mixing models can be used as a management tool to predict future water quality trends under various development scenarios involving septic-based growth in the southern LPRV and evaluating emerging septic technologies.

Chapter 1: Introduction

1.1 Problem Statement

Groundwater is an important natural resource widely used for drinking, irrigation, and industrial uses. Normally groundwater is of suitable quality for drinking unless it has been impaired by various human activities in urban and agricultural environments.

Management and development of groundwater resources requires knowledge of the quality and quantity of water in urban aquifers and the waters recharging those aquifers.

The Lower Portneuf River Valley (LPRV) aquifer supplies the entire water needs to rural and urban residents of Pocatello and Chubbuck, Idaho. It is located in the lower drainage basin of the Portneuf River, one of the major tributaries to the Snake River watershed in southeastern Idaho (Figure 1-1). The LPRV aquifer system is comprised of two very different subsystems: the northern aquifer system and the southern aquifer system (Welhan et al., 1996) (Figure 1-3). Gravity surveys and lithology from well logs indicate the northern aquifer is several thousand feet deep, and is hydraulically confined by two or more aquitards (CH2MHill 1994c; Welhan et al., 1996; Reid, 1997). The southern aquifer is a narrow, relatively thin strip aquifer consisting of very permeable, coarse gravels, and characterized by high linear flow velocities. These two sub-systems are separated by a prominent subsurface bedrock high in the vicinity of Red Hill.

This research focuses on the southern sub-system, hereafter referred to as the southern aquifer. Previous research in the southern aquifer has focused on hydrologic and water-balance models with only limited geochemical evaluation. The southern aquifer is the main source of recharge to the northern aquifer so it is therefore vital to understand mechanisms that may be influencing groundwater chemistry. Recognizing how and where the aquifer's water chemistry is influenced by natural recharge and

contaminant fluxes will help facilitate proper resource management.

Groundwater quality in the LPRV has been degraded by a plume of trichloroethylene (TCE) that extends along most of the southern aquifer and into the northern aquifer (CH2MHill, 1994b). Several localized areas have also been impacted by nitrate and salts such as sulfate, chloride, sodium, and calcium. Groundwater quality issues such as the TCE plume are being addressed by local agencies; however, no proactive programs have been implemented to locate sources or define the nature and extent of the nitrate and salt impacts. Such contaminants are usually associated with non-point sources like septic leachate, road salting, and runoff from agriculture and ranching operations (Panno et al., 2002; Wilhelm et al., 1996).

Of particular concern is septic leachate as a source of nitrate contamination; unincorporated areas with high spatial densities of households using septic systems have the potential to impact groundwater quality. Portions of the southern aquifer and its recharge areas in the Bannock Range and upper Portneuf watershed are overlain by unincorporated areas where most households have septic systems and private wells. While municipal wells are tested frequently for nitrate, there has been no broad assessment of the potential impact of septic tanks on groundwater that serves the unincorporated areas.

This thesis attempts to address this question and to identify the sources of inorganic salts in the southern Pocatello aquifer by summarizing and evaluating the best available information on the inorganic geochemistry of the southern aquifer and its recharge sources.

The spatial distributions and temporal trends of certain inorganic chemical tracers were assessed to determine natural recharge and contaminant fluxes. This provided a

first order assessment of areas that are the most vulnerable to anthropogenic contamination of groundwater. The data were also used to determine the relative contributions of aquifer recharge by chemical mass balance models, the results of which were also used to test an existing water balance model (Welhan et al., 1996).

1.2 Scope and Objectives

This study focuses primarily on the geochemistry of water in the southern aquifer and the adjacent recharge areas of the system. Chemical constituents in groundwater that were examined include calcium, magnesium, sodium, alkalinity, sulfate, chloride, and nitrate. These constituents were used to characterize naturally occurring background water chemistry, as well as impacts from anthropogenic sources like septic leachate, road salting, and agricultural/animal waste leachate.

The southern aquifer was chosen because of its importance as a source of water to the communities in the LPRV, and because this investigation can build upon and test some of the assumptions and conclusions of previous investigations that have focused on the southern aquifer.

This thesis research first began in 1993 as part of a hydrogeologic study of the LPRV aquifer. Groundwater and surface water samples were collected from the southern aquifer and its recharge areas during several sampling events over a year-long period from June of 1993 through July of 1994. These samples were used to characterize groundwater chemistry and to develop a chemical mass balance model to determine the relative contributions of each recharge component. Many of these locations were re-sampled in the summer of 2004 to evaluate changes in aquifer chemistry over the preceding decade that may be attributable to anthropogenic sources. The sampling campaign in 2004 also included many new locations, and historical data from

government agencies was incorporated to form a more complete picture of the spatial and temporal trends observed in groundwater chemistry.

As defined in Welhan et al. (1996), the southern aquifer extends north from the Portneuf Gap to Red Hill (Figure 1-3). This narrow strip aquifer is bounded on the east by the Portneuf lava flow and on the west by the Proterozoic rocks and alluvial fan deposits of the southern Bannock Range. Recharge to the southern aquifer is thought to originate as underflow primarily from the southern Bannock Range watersheds (Mink Creek, Gibson Jack, Johnny/ City/ Cusick Creeks) and from the upper Portneuf watershed through the Portneuf Gap (Welhan et al., 1996) (Figure 1-4). The Portneuf River and an alluvial aquifer east of the Portneuf Lava may also contribute to recharge but only as minor components. For this investigation, I evaluated all of the available groundwater and surface water inorganic chemistry that was available for the southern aquifer, the southern Bannock Range watersheds, the Portneuf River, and the aquifer east of the Portneuf lava flow. To determine the influence of groundwater underflow through the Portneuf Gap from the upper Portneuf watershed, groundwater samples collected in the area of the Portneuf River valley that extends from the Portneuf Gap to the City of Inkom were also examined.

The specific objectives of this study were to:

1. **Characterize the groundwater chemistry in the southern aquifer and its tributary areas of recharge.** Prior to 1993, the only water chemistry data available in the southern aquifer was from five City of Pocatello municipal wells. Groundwater samples collected in 1993 and 1994 for this study represented the first comprehensive look at aquifer chemistry and its recharge systems.

2. **Analyze the spatial distribution and seasonal variations in groundwater chemistry to characterize sources that affect groundwater quality.** Water sampling was conducted in four separate time periods from the summer of 1993 through the summer of 1994. This provided information on the relationship between seasonal fluctuations in the southern aquifer chemistry and fluxes from recharge sources. The data identified areas affected by contaminants and helped identify potential sources, and how hydrologic and geologic conditions may affect contaminant transport.
3. **Conduct a water sampling event a decade later to evaluate temporal changes in groundwater chemistry and to identify areas where land use (such as high density of septic tanks) may be affecting groundwater quality.** A fifth water sampling event was conducted in the summer of 2004 at most of the previously sampled locations as well as many new locations to determine the impacts of septic effluent.
4. **Apply a chemical mass balance model to evaluate the relative contributions of chemically distinct recharge components to groundwater.** The chemical mass balance approach utilized chemical tracers in the components of aquifer recharge in an attempt to determine their relative contributions to portions of the southern aquifer. The mixing models incorporated fluxes of septic effluent that mix with natural recharge fluxes to match the observed chemistry in the southern aquifer.

1.3 Location

The LPRV is an elongate, northwest-trending valley located in Bannock County, southeastern Idaho (Figure 1-1). It is bounded to the southwest by the Bannock Range and to the northeast by the Pocatello Range. The valley opens to the north to merge with

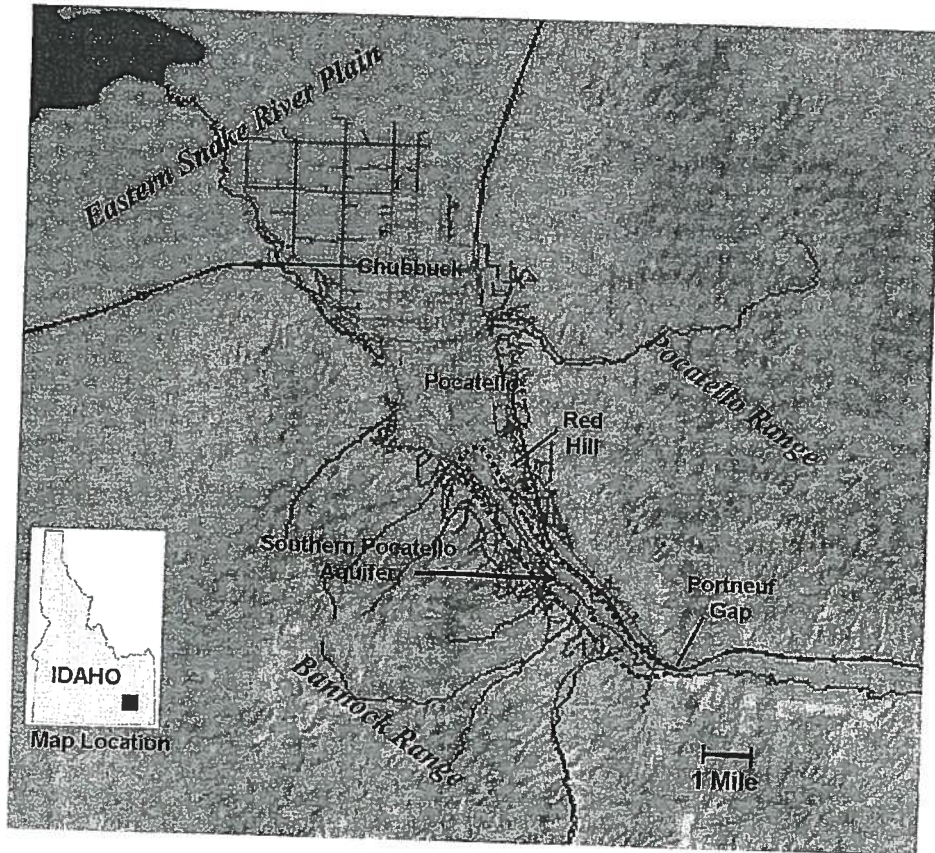


Figure 1-1: Location Map for the Lower Portneuf River Valley. The southern Pocatello aquifer is outlined with a dashed line.

the volcanic province of the eastern Snake River Plain (ESRP), and is separated from the upper Portneuf Valley on its southern end by a narrow bedrock gorge, known as the Portneuf Gap. The primary drainage in the valley is the Portneuf River which originates from the Gem Valley and Marsh Creek watersheds of the upper Portneuf basin. The river enters the LPRV through the Portneuf Gap and flows through the valley, entering American Falls reservoir and the Snake River drainage a few miles northwest of Pocatello.

The valley floor is relatively flat and sits at approximately 4,500 feet above mean sea level (amsl) with mountain ranges to the south rising abruptly to elevations greater than 7,000 feet amsl. The climate is considered to be semi-arid with annual precipitation varying from less than 15 inches in the lower valley to more than 30 inches in the higher elevations of the Bannock Range (> 6,000 feet amsl). Most of the precipitation falls in the form of winter snow with lesser amounts as spring and autumn rains.

The Cities of Pocatello and Chubbuck form the major municipalities in the valley and have a combined population of more than 60,000. Land use in the northern part of the valley is primarily commercial and urban-residential. The southern portion of the valley floor is dominated by rural-residential property with minor agricultural use. Residential development has encroached onto many of the benches and tributary drainages of the Bannock Range on the western flank of the valley. Most of the land in the higher elevations of the Bannock Range is federal property maintained by the U.S. Forest Service and Bureau of Land Management.

1.4 Geology

1.4.1 Structural and Topographic Development of the LPRV

The most recent geologic mapping for the Pocatello area was completed by Link and Stanford (1999) and Rodgers and Othberg (1999). The most recent and comprehensive study of the geologic evolution of the Pocatello Valley can be found in Osier (2004), whose work formed the basis for the following geologic summary of the study area.

The Bannock Range consists predominantly of Late Proterozoic Brigham Group quartzite and younger Cambrian limestone, and the Pocatello Range consists almost

entirely of the older Late Proterozoic Pocatello Formation: diamictite, argillite, shale, and greenstone-metamorphosed basalt (Trimble, 1976; Link et al., 1987; Link and Stanford, 1999; Rodgers and Othberg, 1999; Osier, 2004). These formations were deposited in a west facing, continental shelf-slope setting (Armstrong and Oriel, 1965) following late Proterozoic continental rifting (Stewart, 1972). During the early Mesozoic, they were thrust faulted eastward as a portion of the Paris-Putnam thrust sheet during the Sevier Orogeny (Link et al., 1987; Kellogg, 1992). Following compression, the region was eroded and became an area of subdued topography during much of Cenozoic time (Rodgers and Janecke, 1992; Osier, 2004).

Beginning in the Late Cenozoic, much of the western continental plate was subjected to coeval uplift and east-west extension (Christiansen and McKee, 1978). Extension created horst-graben and tilt-block/half-graben structures that resulted in the development of the Basin and Range Province, an area typified by long, linear ranges separated by broad, flat valleys (Stewart, 1998). As the horsts were typically faulted along north-trending normal faults, the adjacent grabens (basins) that formed between them accumulated detrital material from the rising horsts (Stewart, 1998). Extension in the Pocatello Valley began approximately 7 to 10 million years ago along the basin's major range-bounding normal fault, the Fort Hall Canyon fault, to create a tilt block/half graben structure (Figure 1-2). This fault strikes generally north-south, dips west 60° beneath the valley floor and has an estimated 6 km of offset (Osier, 2004).

Several transverse east-west striking faults were also formed to accommodate north-south extension (Osier, 2004). These include the Gibson Jack and City Creek faults, which dip to the south, and the Trail Creek fault, which dips to the north (Osier, 2004). In addition, several smaller west-dipping normal faults have been recognized as

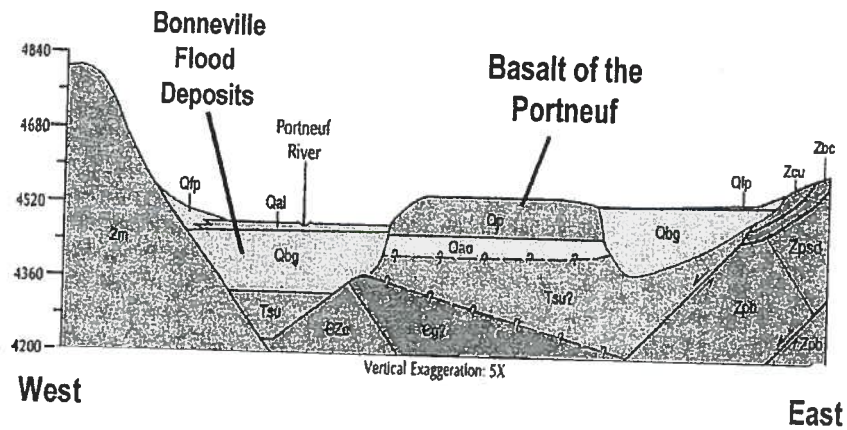


Figure 1-2: East-west cross section through the southern portion of the Pocatello valley (Rodgers and Othberg., 1999). Geologic units represented are: Zpb, Pocatello Formation-Bannock Volcanic Member; Zbc, Blackrock Canyon Limestone; Zpsd, Pocatello Formation-Scout Mountain Member; Zcu, Caddy Canyon Quartzite; Zm, Mutual Formation; Zc, Camelback Mountain Quartzite; Zg, Gibson Jack Formation; Zs, Starlight and Salt Lake Formations; Zao, older Pleistocene alluvium of the Portneuf River; Zp, Basalt of the Portneuf; Zfp, Alluvial fan gravel; Zbg, Bonneville flood deposits; Zal, Alluvium.

having splayed off the northern stretch of the Fort Hall Canyon fault. Osier (2004) hypothesized that these splayed-faults may have accommodated the conflict in directions of motion (transverse) that occurred between the basin's major faults, to create an intrabasinal bedrock-high in the area that is now Red Hill. This initially separated the basin into two sub-basins, north and south.

These basins were filled with clastic sediments and rhyolitic tephra that make up a portion of the Late Miocene upper member of the Starlight Formation. Rhyolitic ash fall tuffs are thought to have originated from the ancestral Yellowstone hotspot just to the north. The differences in sedimentary trends found in the basin fill sequence, with fanglomerate to the north and lacustrine sediments to the south, suggest that basin filling

occurred generally from north to south (Osier, 2004). This style of filling is inferred to have occurred because as the basin was being rifted east-west along the Fort Hall Canyon fault, it was subsequently being tilted southward away from the volcanic edifice forming to the north in response to the migrating hotspot (Osier, 2004).

At the cessation of extensional faulting approximately 7 Ma ago, the Pocatello basin had filled across its footwall and hanging wall, integrating southward drainage with Marsh Valley to the east and Arbon Valley to the west (Osier, 2004). As the ESRP subsided with the migration of the hotspot farther to the north and east, it created a northward-dipping flexure of the bedrock. This initiated a reversal of the ancestral Portneuf River's drainage from southward to northward, and a long period of fluvial degradation (Osier, 2004). With the superposition of the ancestral Portneuf River across the basin's footwall, the down cutting developed the Portneuf Gap and captured the Marsh Valley drainage to the north and eventually Gem Valley drainage further east (Osier, 2004). Capturing these drainages increased both the discharge capacity of the river and the rate at which basin fill was exhumed, with more than 2,500 feet of basin-fill eventually removed. The Mink Creek drainage at the southern end of the LPRV remained protected against the erosive actions of the river, and contains the only significant amount of pre-Pleistocene basin fill in the Pocatello Valley (Osier, 2004).

During the early to middle Pleistocene, the drainage dynamic in the valley changed from one of northward cutting to southward backfilling. This reversal occurred in response to changes in base level on the ESRP and climatically induced effects of glacial to interglacial evolution. Base level changes on the ESRP were controlled by lateral shifts of the Snake River's position, subsidence and uplift in response to the hotspot migration, and impoundment of the Snake River from basalt flows to form

pluvial lakes during glacial times, causing aggradation in the valley by widespread alluvial fan deposition along the valley's relict hanging wall slope on the western flank and the creation of piedmont slopes across the valley (Osier, 2004).

Eventually the valley drainage reversed once again from one of transverse drainage and aggradation of alluvial fans to one of longitudinal through-flow of the Portneuf River. This may have occurred in response to another change in base level on the ESRP due to downcutting of the Snake River, decrease in sediment load available from the already heavily eroded mountain ranges, and increase in drainage discharge from ensuing glaciation. The effect was to incise the piedmont and pediment slopes and to establish the Portneuf River's now dominant fluvial regime. With the cessation of erosion on pediment slopes, these paleo surfaces were buried by several differing loess layers during the last 100,000 years (Osier, 2004).

Following the drainage reversal, two basalt flows originating from Gem Valley, 50 miles to the east, flowed down the Portneuf Valley to a point just south of Red Hill. This Portneuf lava flow has been dated at 430 +/-70ka. The emplacement of the basalt south of Red Hill restricted the Portneuf River to flow through a much narrower area, between the basalt and the west benches, than before. North of Red Hill the Portneuf could meander over a much wider area (Osier, 2004).

1.4.2 Aquifer Development

Municipal wells completed in the northern and southern aquifers derive water from the ancestral American Falls Lake deposits and the Bonneville Flood deposits.

The American Falls Lake formed when the ancestral Snake River was dammed by basalt flows (~ 72ka: Scott et al., 1982), 15 miles west of the City of American Falls, depositing thick sequences (> 500 feet) of lacustrine sediments north of Red Hill in the

LPRV.

In late Pleistocene time (~14,500 ybp), pluvial lake Bonneville in northern Utah overflowed at Red Rock Pass, 40 miles south of Pocatello (O'Conner, 1990). The ensuing flood discharged ~1,500 cubic miles of water over several months' time (O'Connor, 1990, 1993). The event plucked the sides and scoured the Portneuf Basalt and truncated alluvial fans along the western margins of the valley (Osier, 2004). Waning flood waters deposited well-sorted coarse gravels and sands in much of the lower Portneuf River valley. These Bonneville flood gravels have been designated as Michaud gravels by Trimble (1976). South east of Red Hill, much of the valley fill west of the basalt in the southern portion of the valley was scoured to depths of 200 feet (Reid, 1997) (Figure 1-2). North of Red Hill, as the flood debouched onto a wider area and water velocities rapidly decreased, large boulders were deposited on the valley floor (Osier, 2004).

In the northern part of the valley, Michaud gravels comprise a 40-60 foot thick layer that overlies clay units deposited in the ancestral American Falls Lake. Osier (2004) hypothesized that the American Falls Lake was not at full capacity and flood gravels and boulders were deposited into a deeply eroded channel that had been incised by the Portneuf River into the lake sediments.

The extensive gravel deposits from the flooding raised the local base level and inhibited the erosive potential in the valley. The Holocene epoch has been marked by minor deposition in the modern stream channels and upon the small alluvial fans at the mouths of tributaries drainages that debouch onto the valley floor (Osier, 2004).

1.5 Hydrogeology

1.5.1 Sources of Information

Several studies have been conducted in the LPRV related to the hydrogeology of the southern aquifer. Norvitch and Larson (1970) reported on a reconnaissance of the water resources in the Portneuf River Basin. Kindel et al. (1991) and Brown & Caldwell (1994) reported on the hydrogeology of Fort Hall Canyon, a tributary at the southern end of the valley in which a landfill is situated. CH2MHill (1994b) performed an assessment of the hydrogeology and TCE contamination in the southern aquifer. CH2MHill also completed a test well drilling program for the City of Chubbuck in the northern aquifer (CH2MHill, 1994c). Welhan (2001a) discussed groundwater interactions in a portion of the southern aquifer referred to as the Highway Ponds, where pits excavated for gravel expose the water table during times when groundwater elevations rise.

The most comprehensive source of information on the hydrogeology of the LPRV can be found in Welhan and Meehan (1994) and Welhan et al. (1996). These investigations summarized hydrogeologic information in the LPRV, with considerable focus on the southern aquifer, to create a conceptual model of the system and to design a wellhead protection strategy for the municipal well field. Welhan et al. (1996) incorporated a gravity survey later published in Reid (1997), which helped define the base of the Bonneville gravels and depth to bedrock in the LPRV.

1.5.2 Subsystems and the Southern Aquifer

The LPRV aquifer can be geographically divided into three subsystems whose borders reflect changes in subsurface geology (Figure 1-3). Reid's (1997) gravity survey confirmed that a bedrock high does exist in the Red Hill area that separates the basin into two sub basins (southern and northern aquifers). Sediments in the southern basin are

approximately 2,500 feet thick at its deepest points, whereas sediments in the northern basin are at least 4,400 feet thick (Reid, 1997).

Wells in the northern basin are completed to depths of up to 500 feet in an aquifer system comprised of multiple, confined silty gravel and sand units. This stratigraphy is characterized by poorly-sorted sediments (silty clay-rich gravels and sands) and is in sharp contrast to the southern aquifer, which is characterized by coarse, clean Bonneville flood gravels. Wells that tap the southern aquifer gravels have very high water yields at depths ranging from 100-150 feet. Wells in the northern aquifer are drilled deeper to obtain adequate yields from the less permeable strata (Welhan et al., 1996).

The southern aquifer is separated from an eastern Tertiary aquifer (eastern aquifer) by the Portneuf Basalt flow. The sediments that underlie the basalt have been interpreted to be valley fill that predate the Bonneville flood, and lack the high permeability of Michaud gravels. Wells logs indicate that a relatively shallow alluvial/colluvial aquifer exists east of the basalt consisting of poorly sorted clay-rich bouldery gravel and silty sand (Welhan et al., 1996).

The major lithologic units in the southern aquifer are 1) bedrock that is predominantly quartzite, shale, or argillite, predominantly of Proterozoic age; 2) Middle to Late Tertiary indurated sediments and volcanoclastics of the Salt Lake Group and unindurated silty gravels of Pleistocene age; and 3) coarse, clean Holocene gravel in the uppermost part of the valley section, hereafter referred to as the Upper Gravels. These Upper Gravels are considered to be the most prolific water bearing unit in the southern aquifer. Water yields from wells completed in indurated Tertiary sediments and bedrock indicate that these units have hydraulic conductivities that are orders of magnitude lower than the Upper Gravels (Welhan et al., 1996).

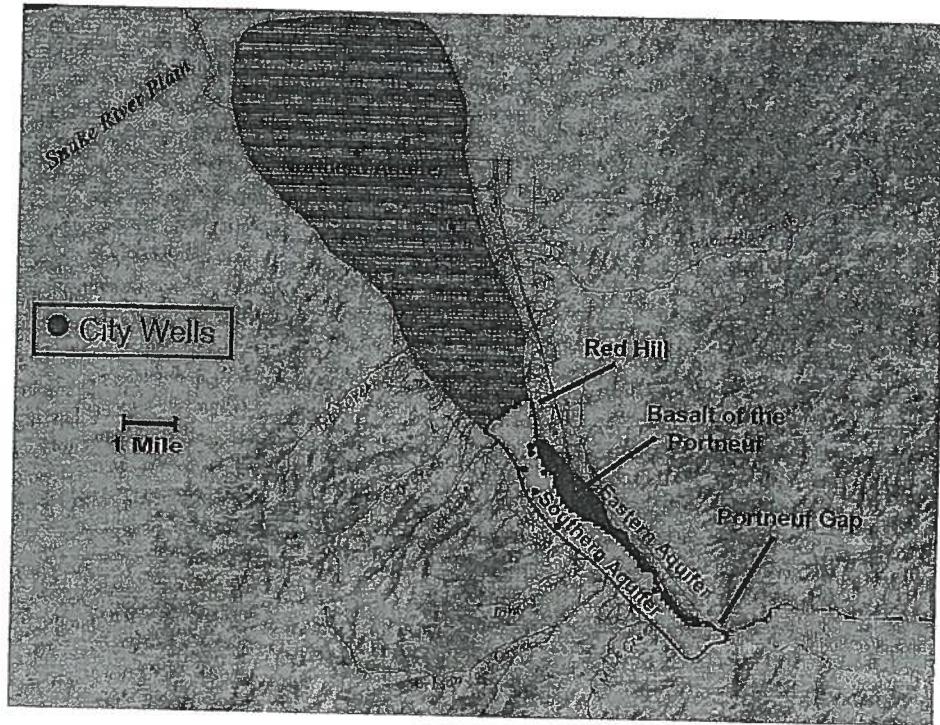


Figure 1-3: Geographic boundaries defining the northern, southern and eastern Pocatello aquifers. Locations of municipal supply wells for the cities of Pocatello and Chubbuck are also displayed.

The lateral boundaries of the southern aquifer are defined by low permeability bedrock and indurated Tertiary sediments to the west, and by low permeability clay-rich Tertiary sediments beneath the Portneuf basalt to the east. The base of the southern aquifer is defined by the top of the Tertiary section (Reid, 1997). Gravity surveys and well logs indicate the top of the Tertiary unit varies considerably at depths from less than 70 feet to more than 250 feet below ground surface. It is assumed that the thickness of the Upper Gravels also varies considerably. Reid (1997) also suggested that the Tertiary units and bedrock high in the Red Hill area may have been channelized and incised by either the Bonneville Flood or the ancestral Portneuf River.

The Upper Gravels can be grouped into two principal units that represent a composite of Bonneville Flood depositional environments: an upper sandy gravel and a coarse, clean gravel (Welhan et al., 1996). The coarse, clean, well-sorted gravels were most likely deposited as a homogenous unit at or soon after peak Bonneville Flood stage. This unit comprises more than half of the upper gravels in the southern aquifer and represents the unit that can be most closely correlated with the Trimble's (1976) Michaud gravels to the north. The top sandy-gravel unit shows variable vertical stratification of gravel, sands, and fines. This stratified material may have been deposited in the recessional stages of the flood or during post-flood fluvial deposition and reworking. In the southern valley, Portneuf River flooding events have capped these Upper Gravels with a 5 to 20 foot layer of silt. On the western margin of the valley, the Upper Gravels interfinger with alluvial fan deposits, which slumped into the valley after being truncated by the Bonneville Flood, or aggraded over time in contemporaneous deposition with the Upper Gravels (Welhan et al., 1996).

The mean hydraulic conductivity (K) estimated from specific capacity and pump tests of wells in the southern aquifer is 2400 ft/d, with a range of 200-8200 ft/day (Welhan et al., 1996; CH2MHill 1994b). The overall hydraulic gradient along the axis of the valley is of the order of 0.0015, but varies significantly in places. In the vicinity of the bedrock high near Red Hill, the gradient appears to increase abruptly as groundwater moves through this constriction in the aquifer. At an effective porosity of 0.3, measured from grab samples of gravels from the Highway Ponds borrow pits, linear groundwater flow velocity is estimated to be of the order of 5-25 ft/day (Welhan et al., 1996). CH2MHill (1994b) determined comparable linear flow velocities ranging from 3.4 to 16.2 ft/day.

Pump tests indicate that the southern aquifer behaves in a confined or semi-confined manner, possibly due to moderate vertical/horizontal anisotropy (of the order of .001; Welhan et al., 1996) due to stratification of facies in the top of the Upper Gravel unit. A pump test performed on municipal well 36 showed an instantaneous response in observation wells 2500 ft away, which is more characteristic of confined conditions. Fluctuations of water levels in municipal wells 13, 28, and 36 in response to barometric pressure variations also suggests the possibility of partial confining conditions.

1.5.3 Water Balance

The sources of recharge to the southern aquifer are believed to be underflow from the upper Portneuf Valley through the Portneuf Gap, Portneuf River seepage losses, underflow from the Bannock Range watersheds, and underflow from the eastern aquifer (Figure 1-4).

Welhan et al. (1996) developed a water balance model for the southern aquifer in an effort to identify and quantify these recharge components. The water balance covered a 510 day period in 1993 and 1994, spanning above- and below-normal water year conditions, respectively. The normalized average during this time period was therefore considered to be a good approximation of the water balance for long-term, average conditions.

Groundwater levels measured in 1993 and 1994 suggested that underflow from the Bannock Range may be more significant than underflow through the Portneuf Gap. In autumn of 1993 through spring of 1994, water levels measured in the aquifer upgradient of Mink Creek and south of the gap were in a decline while water in the aquifer downgradient of Mink Creek stayed at a steady state. Water levels in the aquifer downgradient of Mink Creek did not begin to decline until late summer, which seemed to

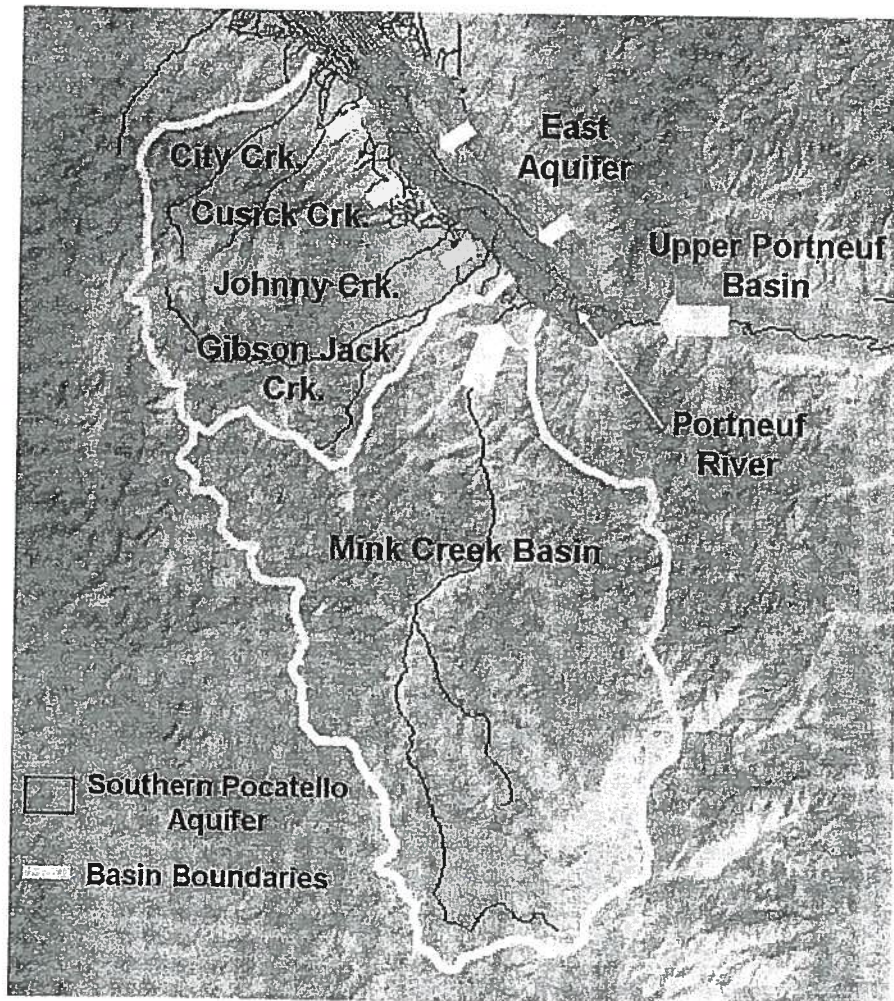


Figure 1-4: Sources of recharge to the southern Pocatello aquifer. Sources include underflow from the upper Portneuf Valley through the Portneuf Gap, Portneuf River seepage losses, underflow from the Bannock Range watersheds, and underflow from the eastern aquifer.

correspond to a drop in groundwater levels in the Mink Creek watershed one month earlier. This suggests that groundwater recharge from the Bannock Range watersheds was responsible for maintaining aquifer storage at near steady state through early summer in 1994 (Welhan et al., 1996).

Several lines of evidence indicated that the Portneuf River was not a significant source of water to the aquifer. Most notable was the lack of correlation between river discharge and changes in aquifer storage. There is a much stronger relationship between aquifer storage changes and local snowfall measurements, indicating that the aquifer is influenced by long-term (6-10 year) variations in precipitation and basin recharge (Welhan et al., 1996).

Norvitch and Larson (1970) measured the river's base flow discharge in 1968 just west of the Portneuf Gap (247 cfs) and in downtown Pocatello (252 cfs). The 5 cfs gain over this reach of the river can be attributed to Bannock Range tributary contributions, and does not indicate significant river loss. Considering that the Portneuf's mean annual discharges for the years 1993 and 1994 were comparable to the measurements Norvitch and Larson measured in December 1968, Welhan et al. (1996) inferred that river loss was not a significant component of recharge in the 1993-94 water balance. These conclusions were corroborated by pump tests that indicated there is limited hydraulic influence on the Upper Gravels when the river is at or near base flow (CH2MHill, 1994b). However, during periods of high river discharge, river losses may be significant (Norvitch and Larson, 1970).

The terms in the water balance equation were calculated from direct measurements and from physical constraints imposed by the geometry of the aquifer. The water balance equation is:

$$\Delta S = Q_g + Q_b + Q_e + Q_r - Q_o - Q_p$$

The change in aquifer storage (ΔS) was determined from water level measurements.

Production records from city wells provided information for pumping withdrawals (Q_p).

Estimates of Portneuf Gap underflow (Q_g) and southern aquifer outflow at Red Hill (Q_o)

were made from cross-sectional aquifer geometry, hydraulic conductivity, and hydraulic gradient. As stated above, river losses (Q_r) were considered negligible, leaving Bannock Range inflow plus eastern aquifer inflow ($Q_b + Q_e$) as a residual unknown term. A direct input from precipitation was not considered to be significant since annual potential evapotranspiration is much higher than annual precipitation on the valley.

Southern aquifer outflow was determined at a point just south of Red Hill where the cross sectional area is well defined from well logs and gravity modeling. Applying the measured hydraulic gradient to transmissivity estimates derived from pump tests, the annual average outflow (Q_o) in 1993-94 was approximately 5.3 billion gallons (Bgal). A similar method was applied to determine inflow through the cross sectional area of the Gap, producing an annual average inflow (Q_g) of 1.06 Bgal. After accounting for annual pumpage ($Q_p = 1.26$ Bgal) and storage changes ($\Delta S = -0.34$ Bgal), the residual component, namely annual inflow from the Bannock Range and east aquifer ($Q_b + Q_e$) was calculated to be 6.1 Bgal (Welhan et al., 1996).

Welhan et al. (1996) surmised that the eastern aquifer's contribution (Q_e) was considerably less than the Bannock Range (Q_b). Groundwater in the eastern aquifer is highly elevated in salts, specifically nitrate, sulfate, and chloride (see section 1.6). A preliminary chemical mass balance evaluation for the aquifer indicated that contributions from the eastern aquifer were less than 10% of the Bannock Range (Welhan et al., 1996). Based on this, the Bannock Range's annual contribution was estimated to be greater than 90% of the water balance residual (5.5 Bgal), and the eastern aquifer's, less than 0.55 Bgal (Welhan et al., 1996).

These water balance calculations indicate that the southern aquifer is heavily dependent on recharge from the Bannock range during winter and spring to replenish

storage lost during peak pumping periods, and that pumping has a significant impact on the aquifer, especially during drought periods (as documented in 1994). As previously stated, the southern aquifer is thought to be the primary source of recharge to the northern aquifer. Total municipal pumpage from the northern and southern aquifer was 8.3 Bgal during the 510 day period of the water balance budget. Total calculated recharge to the system for the same time was 10 Bgal, indicating that some 85% of available recharge was withdrawn and consumed through pumping (Welhan et al., 1996). This excludes private and commercial withdrawals.

Welhan (2001b) updated the water balance model for the entire LPRV (northern and southern aquifers) using the 1993-94 water balance results in combination with precipitation/evapotranspiration accounting to estimate potential recharge from other important tributary drainages, namely Pocatello Creek, to the northern aquifer. Modeling indicated that annual average inflow from Pocatello creek was no more than 0.4 Bgal, and that the southern Bannock Range was the predominant recharge source area, contributing an estimated 7.5 Bgal annually. Pumping withdrawals by the Cities of Pocatello and Chubbuck accounted for 80-85% of this recharge, leaving approximately a 1.1 to 1.5 Bgal annual capacity in normal water years. However, pumping withdrawals from domestic, industrial, and irrigation wells may account for more than 1.2 Bgal/yr (BBC Research and Consulting, 2001), suggesting that groundwater use in the aquifer may be near full capacity, possibly exceeding capacity during dry years.

Appendix C contains data and calculations used in this study to update the water balance model for the period from June 1994 through October 2003. These updated models show that the pumping withdrawals in the southern aquifer are in a delicate balance with aquifer recharge potential, and are currently depleting water from the

aquifer due to drought conditions in the past seven years.

Based on current growth projections, water demand in the LPRV could be more than 9 billion gallons annually by the year 2020, exceeding the estimated 7.5 billion gallon capacity for recharge (BBC Research and Consulting, 2001).

1.6 Water Quality

Information related to groundwater quality in the LPRV can be found in many of the investigative reports referred to in section 1.5.2 (Kindel et al., 1991; Brown & Caldwell, 1994; CH2MHill, 1994b; Welhan et al., 1996; and Welhan, 2001a). The Idaho Department of Environmental Quality (IDEQ) characterized the groundwater quality and the impact of nitrate in the eastern aquifer (Winter et al., 1994), and a report by the USGS describing concentration trends of nitrate in some of Idaho's Groundwater-Quality Management areas includes some discussion of the Pocatello area (Parliman, 2002). Chen (2001) defined the hydrogeochemistry of the Portneuf River in the LPRV for part of 1999 and 2000.

Welhan et al. (1996) summarized information on temporal and spatial trends of inorganic constituents in the southern and northern aquifer systems and suggested possible sources of the observed salt contamination trends, and BBC Research and Consulting (2001) estimated future economic impacts of aquifer protection strategies for the LPRV in relation to projected growth.

As of 1996 much of the aquifer had shown little or no trends in dissolved salt concentrations. Historical records from municipal wells 33, 16, 12, and 10 in the southern and central portions of the aquifer suggested that impact from salts was minimal (Figure 1-5). One single deep well completed in a deep confined gravel unit in the northern aquifer (Well 18) also showed no increase over time, suggesting that the deep

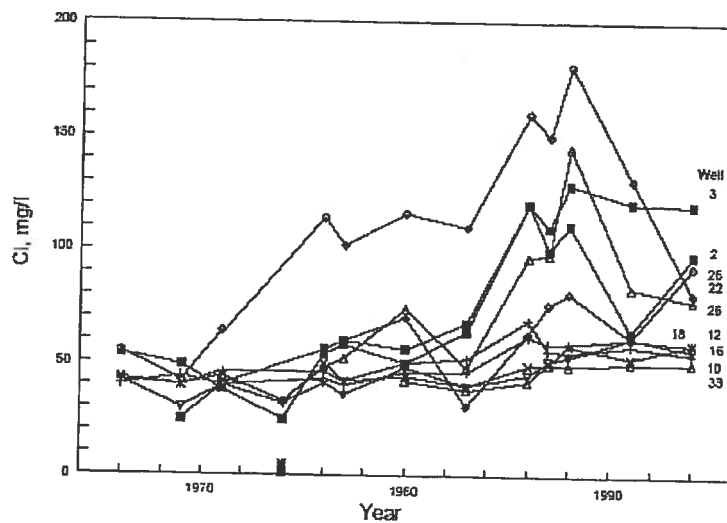


Figure 1-5: Temporal trends in chloride concentrations in Pocatello municipal production wells as presented in Welhan et al., 1996.

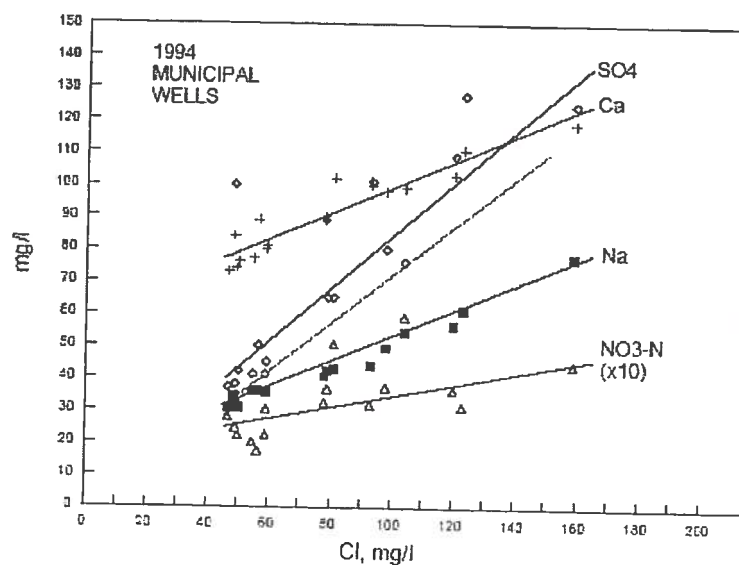


Figure 1-6: Correlation of major ionic constituents in municipal well water based on 1994 data as presented in Welhan et al., 1996.

aquifers were not impacted (Welhan et al., 1996).

In those areas where salt concentrations had increased, coupled proportions of Cl and SO₄ were consistently observed, indicating their enrichment may be from a common source (Figure 1-6). One area displaying salt impacts in the shallow system (in municipal wells 22 and 26) was in the northern aquifer downgradient of Pocatello Creek. Temporal trends indicated that salt concentrations in municipal wells had gradually increased since about 1970 (Figure 1-5). The coupled increases of Cl, SO₄, Na, Ca, Mg, and NO₃-N suggest that septic leachate may be responsible. Alternatives include combined sources of dissolved salt from road salting, and nitrate and other salts derived from agricultural/animal waste sources. A landfill located in the Pocatello Creek drainage was ruled out on the basis of a site-specific groundwater investigation (CH2MHill, 1994a).

The second area where substantial salt impacts have been observed is in the southern aquifer adjacent to the northern terminus of the Portneuf lava flow in the vicinity of municipal wells 2, 3, and 28. These wells have shown chronic elevated levels of nitrate and other salts, most likely due to capture of contaminated groundwater from the eastern aquifer. The eastern aquifer is known to be heavily contaminated with these constituents and has been designated as a nitrate priority area (Winter et al., 1994). A study to determine the source of nitrate in the eastern aquifer showed that there was a strong correlation between nitrate and chloride, and concluded that the source of contamination was local septic tanks (Winter et al., 1994). Municipal well 15, which is located in the eastern aquifer, was abandoned shortly after it was installed, due to nitrate contamination.

The southern aquifer has also been impacted by trichloroethylene (TCE) contamination, which was first detected in several municipal wells in 1991 (CH2MHill,

1994b). The source of the TCE is generally believed to originate from the old Fort Hall Canyon landfill (Figure 1-7). An elongated plume of TCE extends along the western boundary of the southern aquifer, and terminates in the southern portion of the northern aquifer. A total of nine municipal wells have been impacted by the plume, with one well requiring closure due to exceedence of EPA's drinking water standards.

Of particular interest is how the geometry of the plume, acting as a tracer, may reflect hydrodynamic controls on contaminant dispersion. The TCE plume is very elongated and narrow and hugs the western boundary of the southern aquifer for most of its length. Due to the high linear velocities in the aquifer, longitudinal dispersion should be significantly greater than lateral dispersion (Welhan and Meehan, 1994; Welhan et al., 1996). Notably the TCE plume begins to migrate away from the western margin of the aquifer in the Red Hill area (Figure 1-7), to follow the central axis of the valley. One possibility for this change in direction is that the plume is following channelized features cut into the paleo-valley by the ancestral Portneuf River and/or by the Bonneville Flood (Welhan, pers. comm., 2004). In his gravity survey, Reid (1997) hypothesized that a channel may have been cut through the bedrock high in the Red Hill area, very close to the location where the plume begins diverging from the western margin of the valley. Alternatively, the plume may be responding to hydraulic sinks created by pumping withdrawals in municipal wells 12, 16, and 30.

1.7 Management Issues

The economic implications of additional measures to protect groundwater in the LPRV were outlined in a recent study (BBC Research and Consulting, 2001). One of the primary recommendations was to begin managing storm water runoff and septic leachate impacts. The City of Pocatello and Bannock County have made efforts to address water

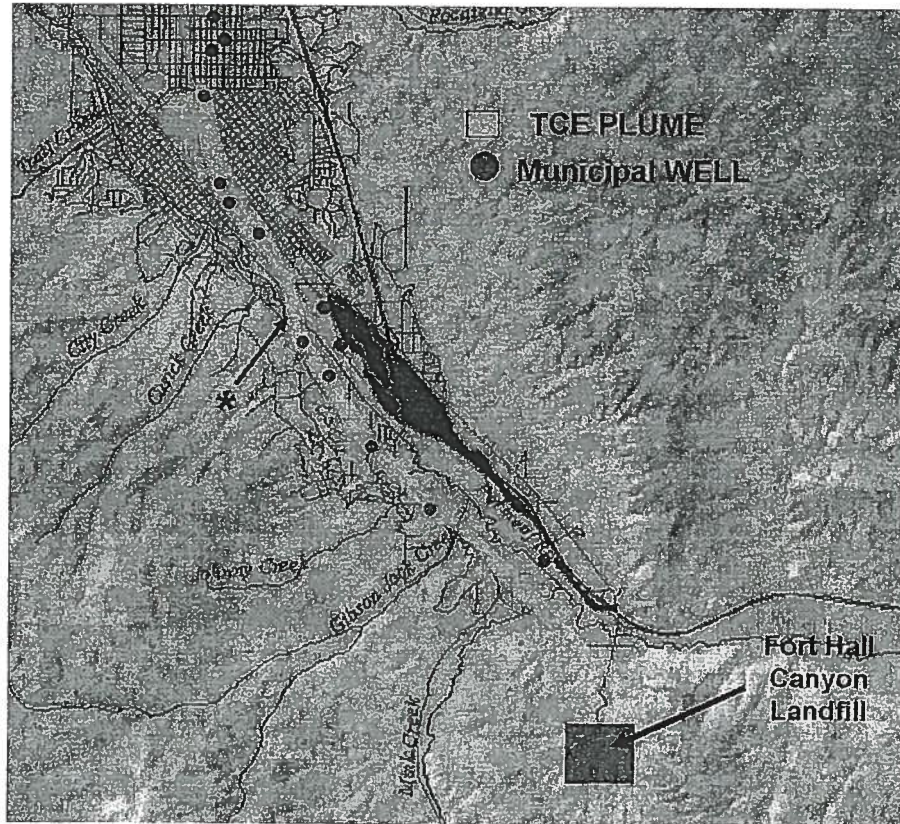


Figure 1-7: Plume of TCE contamination in the southern and northern Pocatello aquifer. The plume most likely originates from the Fort Hall Canyon landfill at the southern end of the valley. * Locates the point where the TCE plume begins to migrate away from the western margin of the aquifer possibly due to channelized features cut into the paleo-valley.

quality issues such as TCE, but, proactive programs have not been implemented to address the impact of other contaminants like nitrate and salts from non-point sources such as septic leachate, road salting, stormwater runoff, and runoff from agriculture and ranching operations

Nearly 90% of homes in unincorporated areas of the LPRV use septic systems. The 1990 census indicated that approximately 1,600 homes on septic overlie the aquifer

and its recharge zones; this grew to 2,000 homes by the year 2000. At this rate of growth, another 1000 households could be on septic by the year 2020 with the potential to affect groundwater quality.

The maximum contaminant level (MCL) allowed in drinking water by the U.S. Environmental Protection Agency is 10 mg/l of nitrate-nitrogen ($\text{NO}_3\text{-N}$). Human health risks associated with nitrate levels above 10 mg/l include methemoglobinemia in infants (blue baby syndrome), non-Hodgkin's lymphoma, gastric cancer, thyroid disorder, hypertension, and birth defects (West, 2001).

Portions of the southern aquifer, and more particularly some of the major recharge areas in the Bannock Range and south of the Portneuf Gap, are unincorporated areas where many households have septic systems. Most of these households also have their own water supply well or are connected to a local community well. To date, there has been no systematic assessment of the potential impact of septic tanks on groundwater in the LPRV.

Chapter 2: Methodology

2.1 Selection of Historical Data

Historical groundwater chemistry data was available from several local and state government agencies. The City of Pocatello provided chemical analyses from municipal supply wells dating back to 1952, though only analyses later than 1970 were considered sufficiently reliable. The city also provided information on monitor wells sampled in 1996 as part of a sewer extension study (Montgomery Watson, 1996).

Water analyses for local community wells and the City of Inkom's municipal wells were obtained from IDEQ. The Idaho Department of Water Resources (IDWR) supplied water quality data for domestic wells in the Portneuf River Valley sampled from 1990 to 2003. Water analyses from monitoring wells and domestic wells in the vicinity of the Fort Hall Canyon landfill were provided by Bannock County.

Several data quality problems associated with the municipal well analyses provided by the City of Pocatello were discovered in this study. Some years (1966-1976) appeared to have elevated levels of nitrate, but the analytical reports for those years indicated that nitrate had not been reported as nitrate-N ($\text{NO}_3\text{-N}$), as it had been for other years. Other problems included sampling dates that didn't match dates on city reports, and some constituents that had exceedingly high or low concentrations, possibly due to decimal errors. Analytical results and sampling dates that could not be verified by the reported values on the laboratory analytical reports were omitted from the data set.

The inorganic water chemistry data collected from the agencies outlined above were incorporated into a single data set (provided as Appendix A and in digital format on the accompanying CD-ROM). Geographically, well locations in the data set extend from

the upper LPRV a few miles south of the City of Inkom, to parts of the northern Pocatello aquifer system. Water analyses from wells and streams in drainages of Mink Creek, Gibson Jack Creek, Johnny Creek, Cusick Creek and City Creek are also included in the data set.

2.2 Selection of Sample Locations

2.2.1 1993-1994 Sampling

The selection of sampling locations for the samples collected in 1993 and 1994 were based on meeting three of the study's objectives: 1) characterizing groundwater chemistry in the southern Pocatello aquifer, 2) identifying chemistry of various components recharging the aquifer, and 3) defining spatial and seasonal changes in chemistry.

Four sampling events over a one-year interval were performed in June-July, 1993, December, 1993, March-May, 1994, and July-August, 1994. Seventeen wells were sampled from the main aquifer in the first sampling event. By the summer of 1994 the number of sampling locations in the main aquifer had grown to 75. Most of these groundwater samples were from domestic wells but included eight municipal wells and ten monitor wells (Figure 2-1). Surface water samples were also collected from the "Highway Ponds", where pits excavated for gravel periodically expose the water table (Welhan, 2001).

To characterize the chemistry of recharging groundwater, water samples were collected from the east aquifer (three locations), south of the Portneuf Gap (three locations), West Bench tributaries of the Bannock Range (28 locations) and the Portneuf River (1 location) (Figure 2-1). Water samples collected from the West Bench drainage

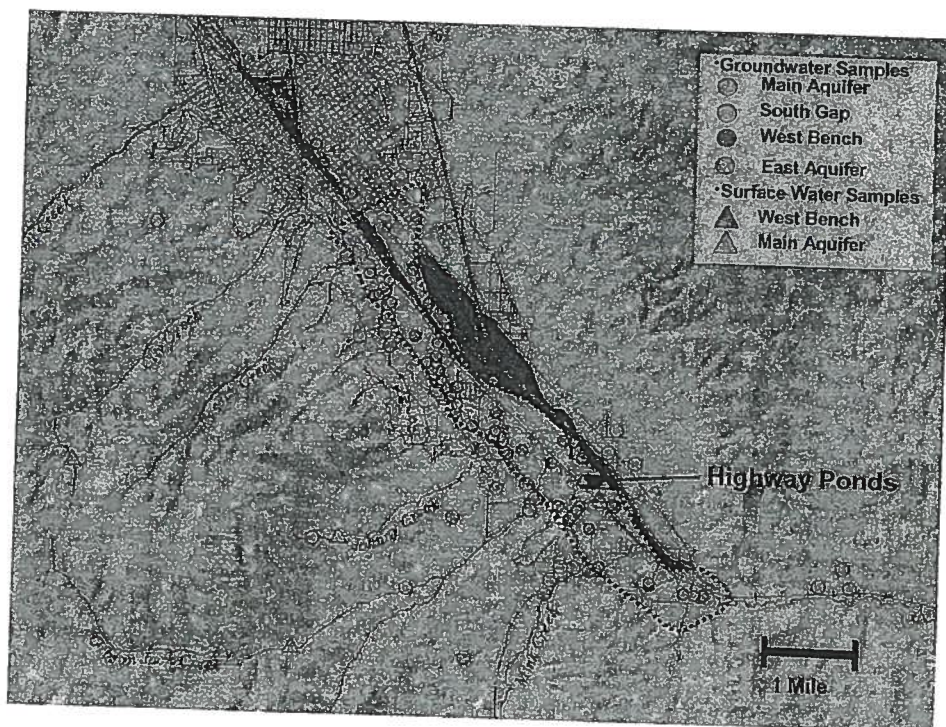


Figure 2-1: Locations of water samples collected in 1993 and 1994. The boundary of the southern Pocatello aquifer is outlined with a dotted line. The locations of surface water samples collected at the Highway Ponds are also shown.

basins included surface water samples from each tributary stream.

2.2.2 2004 Sampling

In the summer of 2004, another suite of water samples was collected to determine changes in groundwater chemistry since 1994 and to evaluate water quality in areas where high densities of septic tanks may impact groundwater quality. Forty of the main aquifer wells sampled in 1994 were re-sampled in 2004. Declining water levels in the main aquifer since 2000 left many wells inoperable, which precluded them from being resampled. An additional 22 wells in the main aquifer were sampled in 2004 that had not been previously sampled (Figure 2-2). More than 80% of the locations that had been

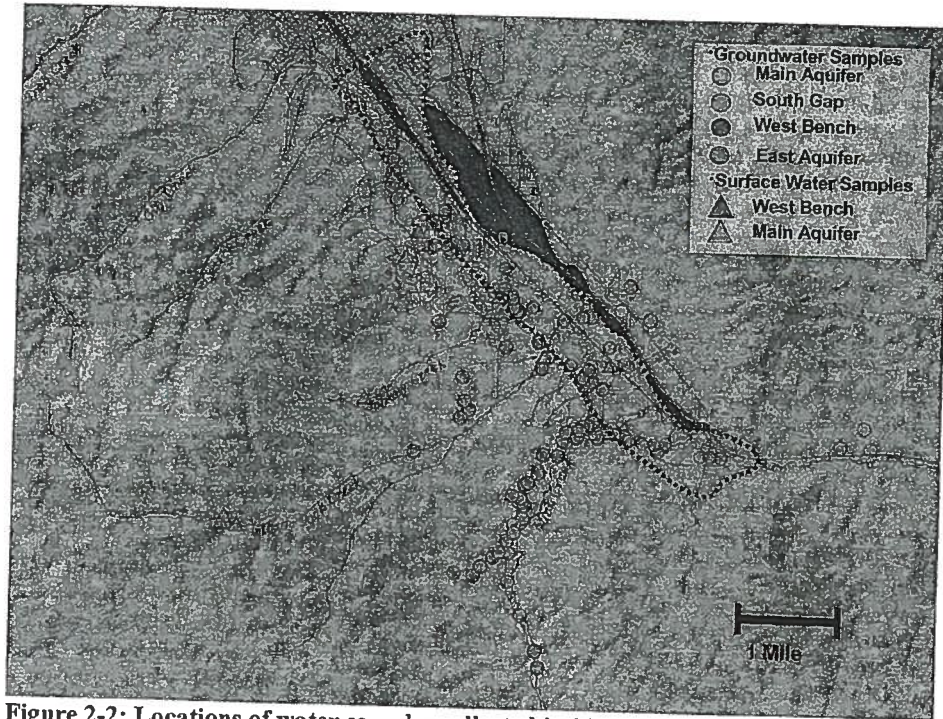


Figure 2-2: Locations of water samples collected in 2004. Notice that there are many more samples collected in Mink Creek drainage as compared to the 1993-1994 sampling period.

used to define the chemistry of recharge components in 1994 were re-sampled.

Figure 2-3 shows the inferred locations of septic tanks in the LPRV as obtained from IDEQ (Byrd, M., written comm., 2004).

The methodology for determining septic locations was based on Bannock County parcel and address data bases and city sewer information. Where an address was associated with a county land parcel, it was assumed that there was a residence on that plot. If the residence was outside of an area serviced by sewer, or the residence was more than 200 feet from a municipal sewer line, it was assumed that the residence relied on septic disposal. Although the accuracy of this assessment has not been quantified, the IDEQ map is the best and most representative depiction of residential septic use that is

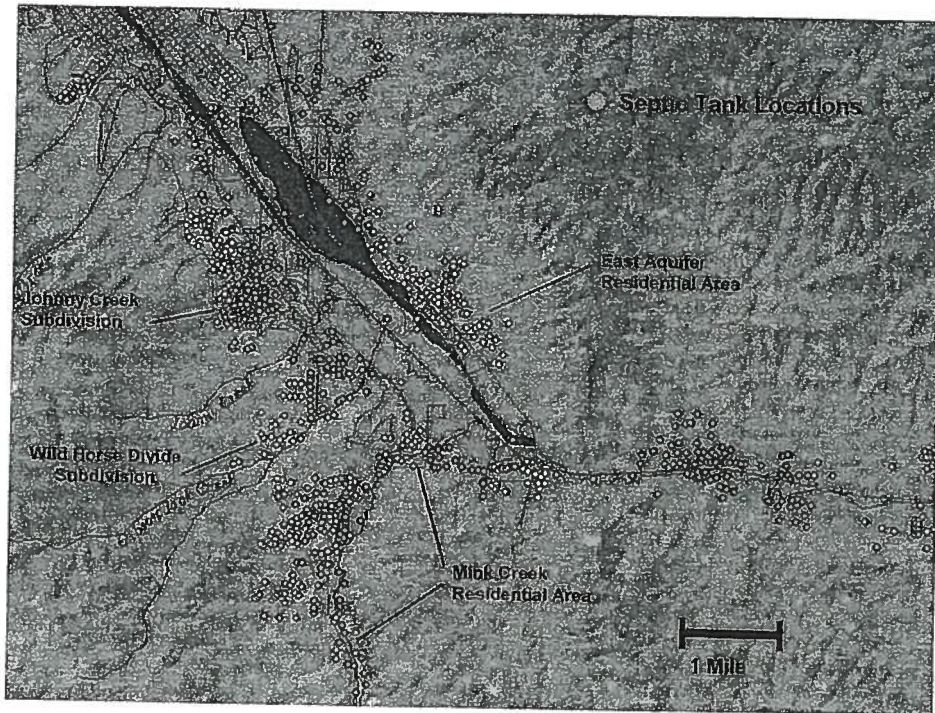


Figure 2-3: Inferred locations of septic tanks in the lower Portneuf River valley. Septic location data was obtained from IDEQ and the Idaho Geological Survey. Areas of high septic density in the Johnny Creek subdivision, Wild Horse Divide subdivision, Mink Creek residential area, and the residential area that overlies the east aquifer are identified.

currently available. IDEQ is planning a quality control check to determine the accuracy of this information.

Figure 2-3 depicts several areas of high septic density in the LPRV, most notably in the east aquifer, the Johnny Creek subdivision, and in the Mink Creek and Gibson Jack drainages. Groundwater quality in the east aquifer has been previously studied by IDEQ and was determined to be heavily impacted by septic effluent (Winter et al., 1994). This study focused on evaluating septic impacts in the West Bench drainages, due to their importance as major recharge pathways to the southern aquifer. Most

residences in the Mink Creek and Gibson Jack areas rely on localized residential and community well service that may be impacted by septic effluent.

An additional 32 locations in Mink Creek and three in Gibson Jack (Wild Horse Divide subdivision) that had not been sampled during the 1993-1994 study were sampled in 2004. Only three domestic wells and two monitor wells were sampled in the Johnny Creek subdivision because it is served by the municipal water system.

2.3 Sampling Methods

All water samples were collected, transported, and stored in 1-liter polyethylene plastic bottles. Samples from domestic wells were collected from spigots closest to the wellhead to ensure that the sample had not passed through a water softener. Wells were purged for 5 to 15 minutes and measurements of temperature, specific conductance, and pH were monitored during each purge. A sample was collected when these constituents stabilized.

Samples collected from monitor wells were collected with a 2.5-foot long stainless steel "thief sampler" that was lowered to the middle of the screened interval in each monitor well. A weighted messenger was then sent down the well to activate the spring loaded mechanism on the sampler, capturing a 1-liter sample at that depth. The thief sampler was decontaminated with de-ionized water after each use.

Surface water samples were collected directly into 1-liter polyethylene plastic bottles.

All samples were stored in ice coolers and refrigerated until analyzed.

2.4 Laboratory Methods and Quality Control

Samples collected in 1993-1994 were analyzed for calcium (Ca), magnesium

(Mg), sodium (Na), potassium (K), sulfate (SO_4), chloride (Cl), nitrate-N ($\text{NO}_3\text{-N}$), silica (Si), total alkalinity (HCO_3), and total hardness (CaCO_3). Total alkalinity and $\text{NO}_3\text{-N}$ analyses were performed within 24 hours of sampling and prior to filtration. Total alkalinity was obtained by titration with standardized acid. $\text{NO}_3\text{-N}$ was analyzed on unfiltered samples using a HACH DR/2000 spectrophotometer (HACH method 8039; HACH, 1991).

After filtering samples through a 0.45 micron filter, a HACH DR/2000 spectrophotometer was used to analyze sulfate (HACH method 8051), chloride (HACH method 8113), and silica (HACH method 8189). Total hardness and calcium (calcium hardness) were analyzed by EDTA titration (HACH, 1988; HACH method 8226). Magnesium was calculated by subtracting calcium's contribution to total hardness.

Sodium and potassium were analyzed at the Idaho State University geochemistry laboratory using flame atomic absorption spectroscopy.

Samples collected in 2004 were analyzed for Cl, SO_4 , and $\text{NO}_3\text{-N}$ using the same procedures as described above. A split set of 46 samples was sent to Intermountain Analytical Services Inc. (IAS) in Pocatello, Idaho, for analysis of Cl, SO_4 , $\text{NO}_3\text{-N}$, Ca, and total hardness (as CaCO_3).

Multiple analyses were performed to check the precision of the HACH spectrophotometer and the EDTA titration methods. HACH standards were used for SO_4 (50 ppm), Cl (50 ppm) and $\text{NO}_3\text{-N}$ (2 and 10 ppm). No standards were available for Si, Ca, Mg, or Alkalinity so laboratory tap water was used. Table 2-1 shows the analytical precision for each analyte as the relative standard deviation (equivalent to a 68% confidence level). The standard deviations for SO_4 , $\text{NO}_3\text{-N}$, and Si were all less than 1

Table 2-1: Analytical precision of HACH analyses for samples collected in 1993-1994 and 2004. Multiple analyses were performed on standards and tap water (Tap*) to identify the precision of the HACH spectrophotometer and EDTA titration methods. All results are represented as ppm (mg/l).

	SO ₄	Cl	NO ₃ -N	NO ₃ -N	Si	Ca	Mg	Alk. (HCO ₃)
1993-1994 Data Precision								
Standard	50 ppm	50 ppm	2 ppm	10 ppm	Tap*	Tap*	Tap*	Tap*
Median	49	47.8	2.1	10.7	23.8	77.9	33.9	343
Std Dev.	0.57	2.99	0.14	0.18	0.84	6.32	4.02	6.48
2004 Data Precision								
Standard	50 ppm	50 ppm	2 ppm	10 ppm				
Median	51.0	51	2.1	10.7				
Std Dev.	0.92	3.02	0.11	0.99				

Table 2-2: Analytical precision as a function of sample dilution.

Sample Conc. (ppm)	Cl < 100	Cl 100 to 250	Cl > 250	SO₄ < 70	SO₄ > 70
1993-1994 Std. Dev.	2.99	7.45	29.9	.57	2.9
2004 Std. Dev.	3.02	7.55	30.2	0.92	4.6

mg/l, and for Cl, Ca, Mg, and total alkalinity (HCO₃), ranged from 3 – 6.5 mg/l.

In some cases, dilution with deionized water was necessary to bring a sample to within the absorption limits of the spectrophotometer. When this became necessary, several analyses were made at varying dilution ratios and the deionized water itself was tested to ensure integrity. Table 2-2 shows the standard deviations for Cl and SO₄ determined from multiple dilutions, over the range of sample concentrations.

Two different test statistics were used to determine whether analytical results of duplicate samples sent to the IAS laboratory in 2004 were statistically equivalent to results from the HACH spectrophotometer.

A paired Student's t-test was performed between the 46 pairs of HACH and IAS

measurements for Cl, SO₄ and NO₃-N. At the 95% confidence level, the duplicate sets of analyses for Cl, SO₄, and NO₃-N all passed the t-test, indicating that HACH and IAS analytical accuracies are comparable.

The duplicate lab analyses were also compared using the Z-statistic as defined by Williams (1997). When the analytical standard deviations are known, it is possible to determine, at a specified confidence level, whether the results of a replicate pair of analyses are statistically equivalent. The Z-statistic is defined as:

$$Z = \frac{|x - y|}{\sqrt{((Sx)^2 + (Sy)^2)}} \quad (1)$$

where

x is the result of the HACH analysis, in ppm

y is the result of the IAS laboratory analysis, in ppm

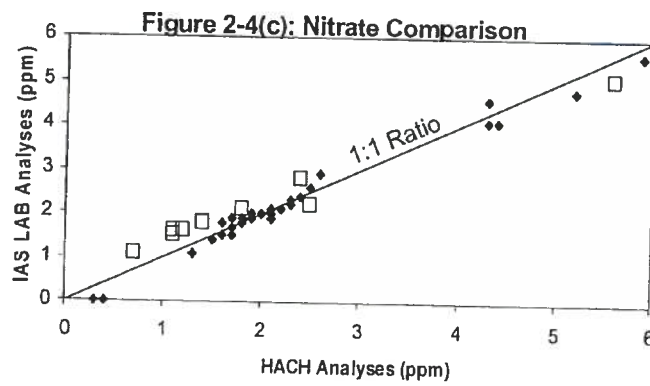
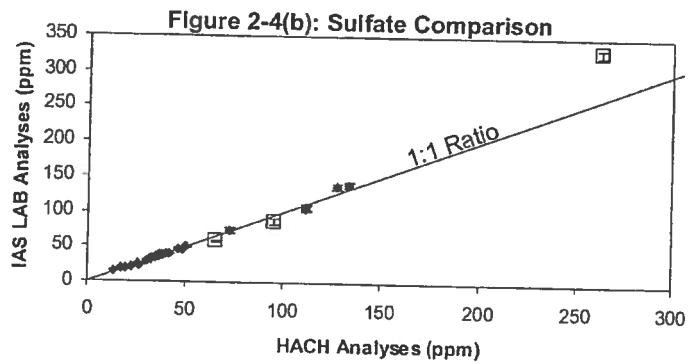
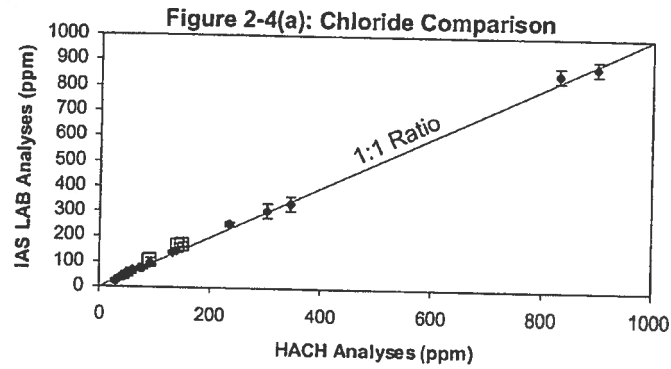
Sx is the standard deviation of x

Sy is the standard deviation of y.

The standard deviations for the HACH analyses were taken from Tables 2-1 and 2-2. Standard deviations for IAS lab analyses were approximately 5% of the reported concentration of each analysis (Ryan Pattie, personal comm., 2005)

The analytical results of replicate samples can be considered statistically equivalent at the 95-percent confidence level if the Z-statistic is less than or equal to 1.96. Calculated Z-values for the 46 duplicate lab analyses indicated that only three chloride, three sulfate and nine nitrate-N sample pairs failed the comparability test at the 95% level. Comparability tests at the 99-percent confidence level (Z-statistic ≤ 2.575), indicated no chloride, one sulfate and five nitrate-N samples pairs failed.

Figures 2-4 (a) (b) and (c) compare the paired of HACH and IAS lab analyses for Cl, SO₄ and NO₃-N. The paired T-test and Z-value comparisons indicate that the



Figures 2-4 (a) (b) and (c): Sample pairs of the HACH analyses and IAS lab analyses for Cl, SO₄ and NO₃-N. The solid line represents a 1:1 graphic ratio. Larger open squares (□) represent samples pairs with Z-values greater than 1.69. Error bars are displayed for sulfate and chloride for the standard deviations due to dilution affects as shown in Table 2-2.

duplicate laboratory analyses can be said to be statistically equivalent to the HACH analyses with a high level of confidence.

2.5 Chemical Mass Balance Modeling

Welhan et al. (1996) developed a water balance model for the southern Pocatello aquifer based on the physical hydrogeologic constraints of the aquifer (see section 1.5.3). In this study a chemical mass balance approach was used to determine the relative contribution of multiple recharge components to the southern aquifer. The following assumptions were made:

- The chemical tracers are conservative (no chemical reactions).
- All the recharge components have significantly different concentrations of at least one tracer.
- Tracer concentrations are temporally constant or their variations are known.
- Other, unknown recharge components do not contribute significantly to the chemical mass balance.
- There must be complete mixing of all recharge components in the aquifer.

A chemical mass balance equation representing the relative contributions of components can be written as:

$$C_a f_a + C_b f_b + C_c f_c = C_{\text{aquifer}} \quad (2)$$

Where C is the tracer concentration in recharge components a , b , c , and the aquifer, and f is the recharge fraction of each component such that:

$$f_a + f_b + f_c = 1 \quad (3)$$

The inorganic chemical tracers that were considered most conservative and have the best data coverage are Cl, NO₃-N, and SO₄. NO₃-N was assumed to behave conservatively because all groundwaters in this system are believed to be oxidizing and chemical or microbial nitrogen transformations are considered negligible for the short residence time of water and nitrate in the southern aquifer.

2.6 Mixing Cell Models

Mixing models were created using spreadsheet software to determine how well the fluxes of septic effluent and naturally recharging groundwater account for observed concentrations of chemical tracers in the southern aquifer. Cl, SO₄, and NO₃-N were used for this evaluation due to their conservative nature in groundwater, and because they are common contaminants associated with septic effluent.

Analyses of water samples collected in 2004 were used to characterize the concentration of tracers in the southern aquifer. Linear regression analysis was used to evaluate if there was an observed trend of tracer concentrations along the groundwater flow path from the Portneuf Gap to Red Hill. Regression analyses were performed with StatMost® statistical software. A Student's t-test was applied to evaluate the statistical significance of the linear regression coefficient at the 95% confidence level.

The southern aquifer was split into five "cells", M1 through M5 (Figure 2-5). The boundaries partitioning each cell were drawn to separate areas potentially impacted by major influxes from adjacent tributary drainages (i.e. Mink Creek, Gibson Jack Creek) and to reflect high septic densities in the West Bench area (i.e. Johnny Creek subdivision). The water balance results of Welhan et al. (1996) (see section 1.5.3) were used to constrain the magnitude of groundwater recharge through the five mixing cells.

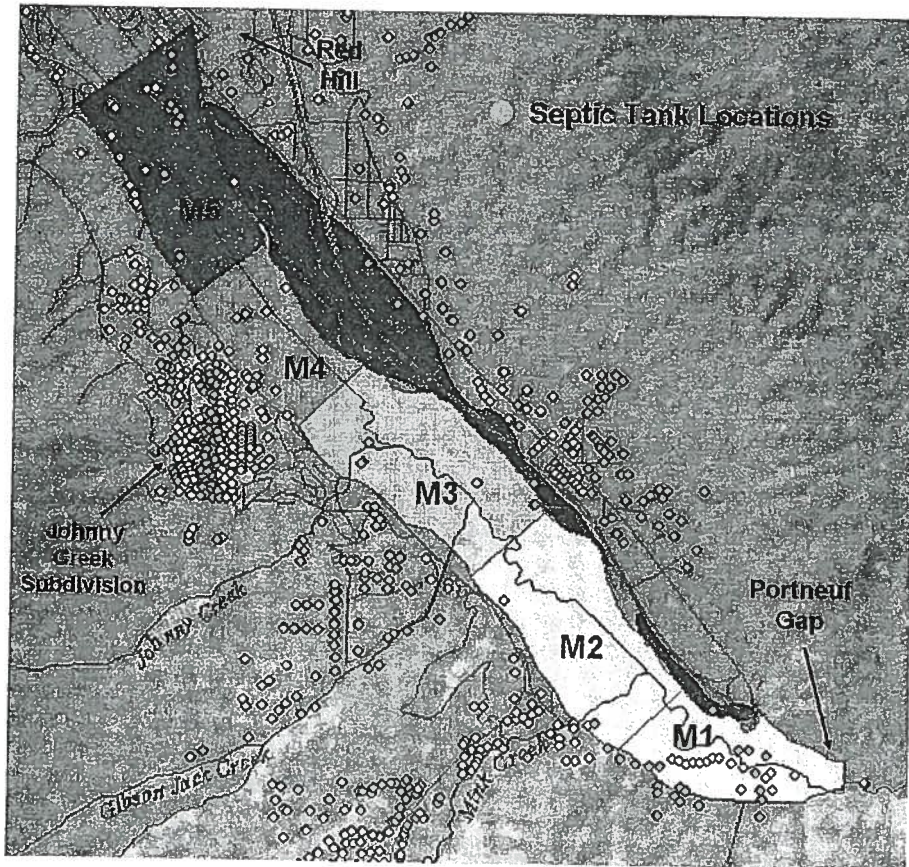


Figure 2-5: Boundaries of cells used for septic mixing models. The boundaries partitioning each cell were drawn to separate areas potentially impacted by major influxes from tributary drainages (i.e. Mink Creek, Gibson Jack Creek) and high septic densities in the west bench area (i.e. Johnny Creek subdivision).

Progressively more complex models were constructed in three stages starting with the simplest mixing scenario. The first model assessed the mixing of septic effluent and main aquifer water. The second modeling approach incorporated an influx of West Bench recharge water in addition to septic effluent contribution. The third modeling scenario considered the impact of recharge from the Portneuf River in addition to the West Bench. Figure 2-6 shows a flow diagram for the third modeling scenario.

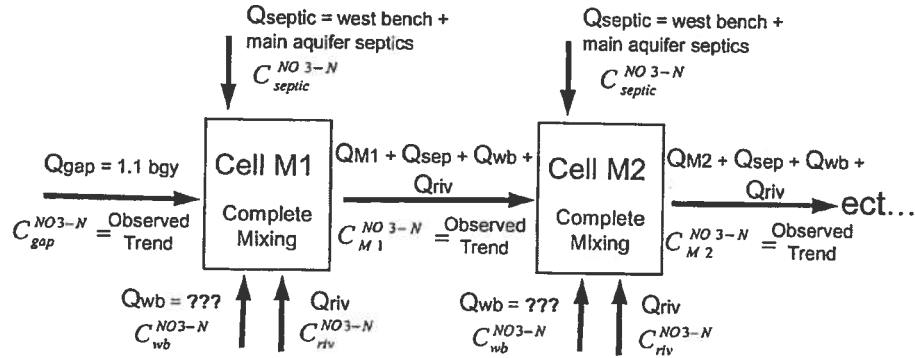


Figure 2-6: Flow diagram of third modeling scenario for septic mixing models. This scenario considers the impact of recharge from the Portneuf River in addition to the West Bench.

2.7 Variogram Methods

Variogram analysis is a statistical technique that characterizes the variability of data values as a function of intersample distance. Samples collected close together tend to show less variability than samples collected farther apart. The term autocorrelation describes this correlation of a single variable with itself in space. A measure of the spatial autocorrelation as a function of distance and direction can be inferred via a semivariogram plot.

Semivariogram plots were constructed using Variowin® statistical software. The semivariogram statistic is defined as:

$$\gamma(h) = \frac{\sum (v_h - v_{x+h})^2}{2 N_h} \quad (4)$$

where N_h is number of pairs of samples separated by a distance h . $V(x)$ and $V(x+h)$ represent a pair of data values at a separation distance, h . When the semivariogram statistic at a separation, h , is independent of direction, $\gamma(h)$ defines the omnidirectional semivariogram. Figure 2-7 depicts the general form of the semivariogram plot.

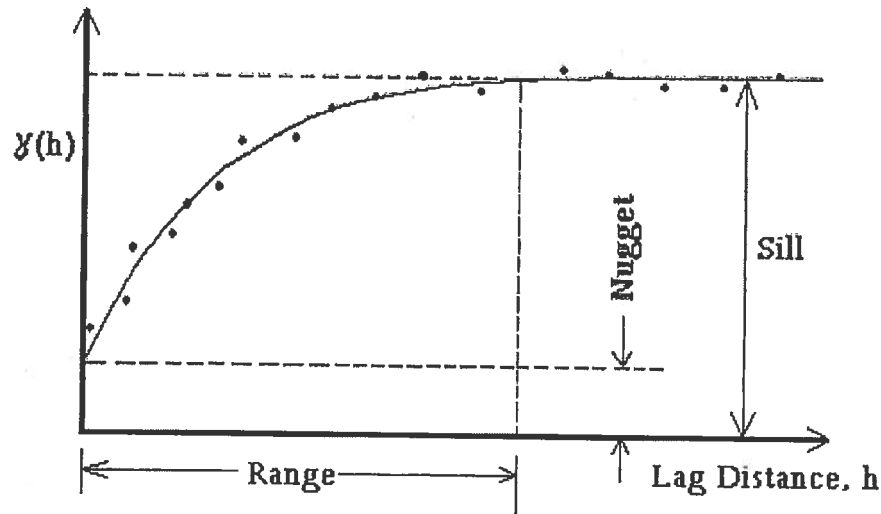


Figure 2-7: General form of a semivariogram for a spatially autocorrelated variable.

Variogram shape is characterized by model type; its range, nugget, and sill; and the presence or absence of drift (Isaaks and Srivistava, 1989; Long et al., 1997). The model type is the mathematical function that describes the shape of the variogram. Two of the most commonly used functions are the spherical and exponential models. The nugget indicates the amount of uncorrelated variation at the smallest separation. This variation may be intrinsic to the data or may reflect sampling error. The range is the distance over which the variable is autocorrelated. Beyond this distance, the variogram ceases to increase; the sill is equivalent to the sample population variance (Isaaks and Srivistava, 1989; Long et al., 1997).

Autocorrelation statistics may be calculated with several estimators, including the semivariogram, covariance and correlogram statistics in order to best infer the nature of the autocorrelation. Further information regarding these measures of statistical

variability may be found in Isaaks and Srivistava (1989).

Chapter 3: 1993 – 1994 Water Quality Results

3.1 Major Ion Chemical Composition

Analytical results of water samples collected in 1993 and 1994 identified by their Location IDs in this chapter can be found in Appendix B. Figure 3-1 shows locations for samples collected in 1993 and 1994, with certain sites referred to in this chapter identified by their Location ID. Seventy five wells were sampled in the southern aquifer, including eight municipal wells. To characterize the chemistry of recharging groundwater, water samples were collected from the east aquifer (three locations), south of the Portneuf Gap (three locations), West Bench tributaries of the Bannock Range (28 locations) and the Portneuf River (1 location).

Figure 3-2 is a Piper (trilinear) plot showing the composition of major ions in water samples from the southern aquifer and its recharge sources. The term “West Bench” identifies samples collected in the Bannock Range drainages (Mink Creek; Gibson Jack, Johnny Creek, Cusick Creek and City Creek).

The Piper plot reveals that, except for the nitrate and salt-contaminated eastern aquifer, the compositional fields for these waters overlap to a large extent. The West Bench waters display the largest compositional diversity which may reflect the varied geologic setting of the Bannock Range, which includes such chemically diverse rock types as limestone, shale, volcanic ash, and quartzite.

Figure 3-3 shows the chemistry of these waters in the form of Stiff diagrams overlain on a map of the southern LPRV. Major ion values displayed as milliequivalents per liter (mEq/l) were calculated from the median concentrations for all the samples collected in each area. The standard deviations are also displayed. A separate Stiff

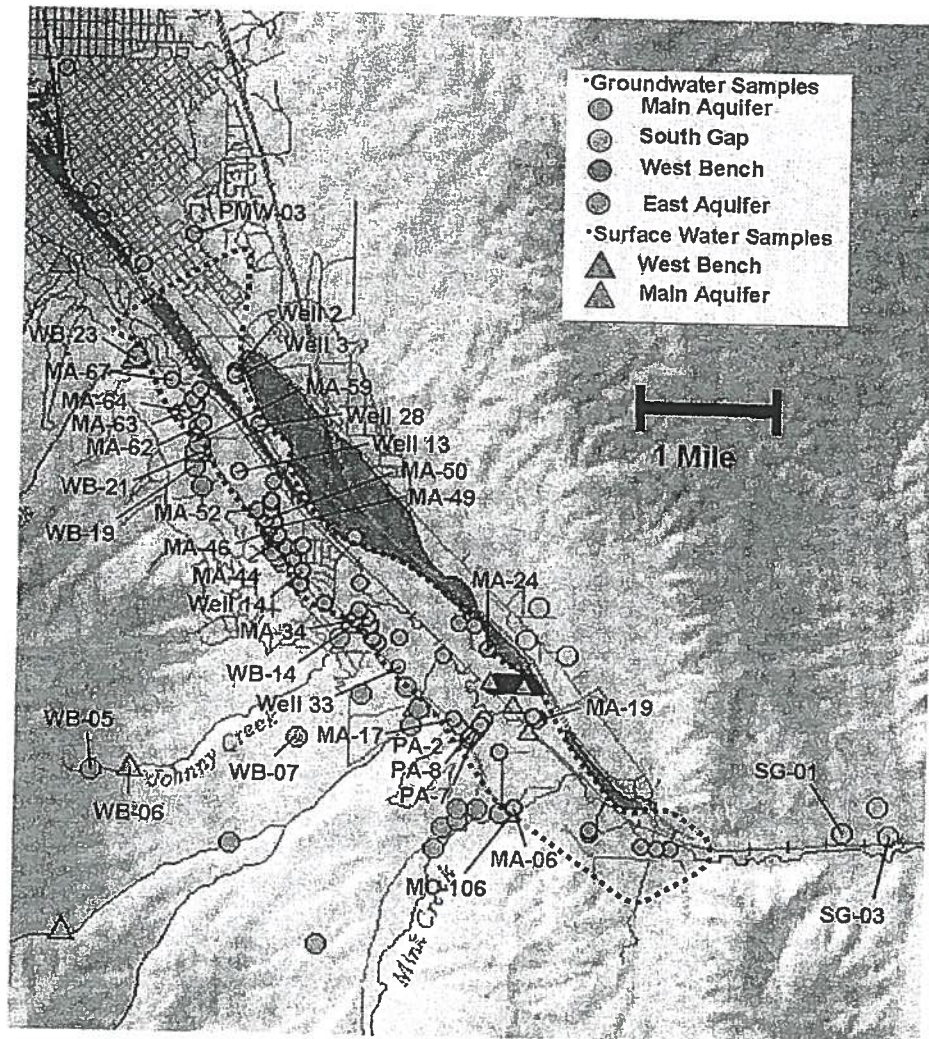


Figure 3-1: Locations of water samples collected in 1993 and 1994. Sample locations referred to in Chapter 3 are identified by their location IDs.

diagram was made for Mink Creek to show its differing chemistry from other West Bench waters.

These Stiff diagrams clearly show the high concentrations of salts in the eastern aquifer. Major ion concentrations south of the Portneuf Gap and the Portneuf River are

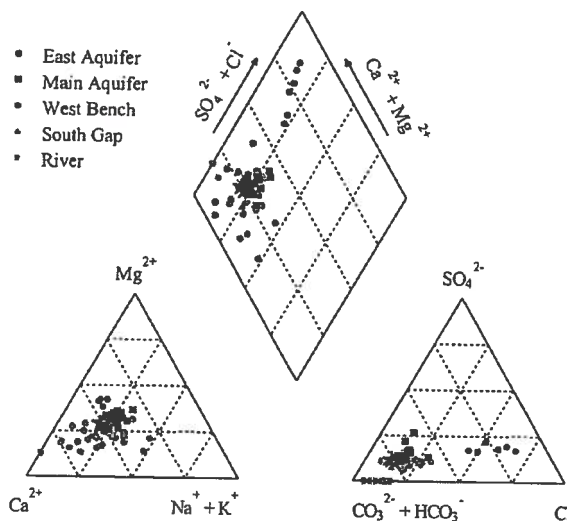


Figure 3-2: Piper plot showing the composition of major ions in water samples from the southern aquifer and its recharge sources. The term “West Bench” identifies samples collected in the Bannock Range drainages. Except for the nitrate and salt-contaminated eastern aquifer, the compositional fields for these waters overlap to a large extent.

very similar to those of the southern aquifer. Aside from the minor compositional diversity displayed in the Piper plot, essentially all West Bench waters are low in TDS, and chemically distinct from the main aquifer. Mink Creek groundwater also is lower in TDS than main aquifer water, but has double the ion concentrations compared to other West Bench waters.

As stated in section 2.5, chemical mass balance modeling requires that recharge components and the southern aquifer should have significantly different concentrations of at least one tracer in order to solve for unknown fluxes. The close similarity in water chemistry in the southern aquifer and water entering through the Portneuf Gap made for dubious solutions to the chemical mass balance (see section 6.1).

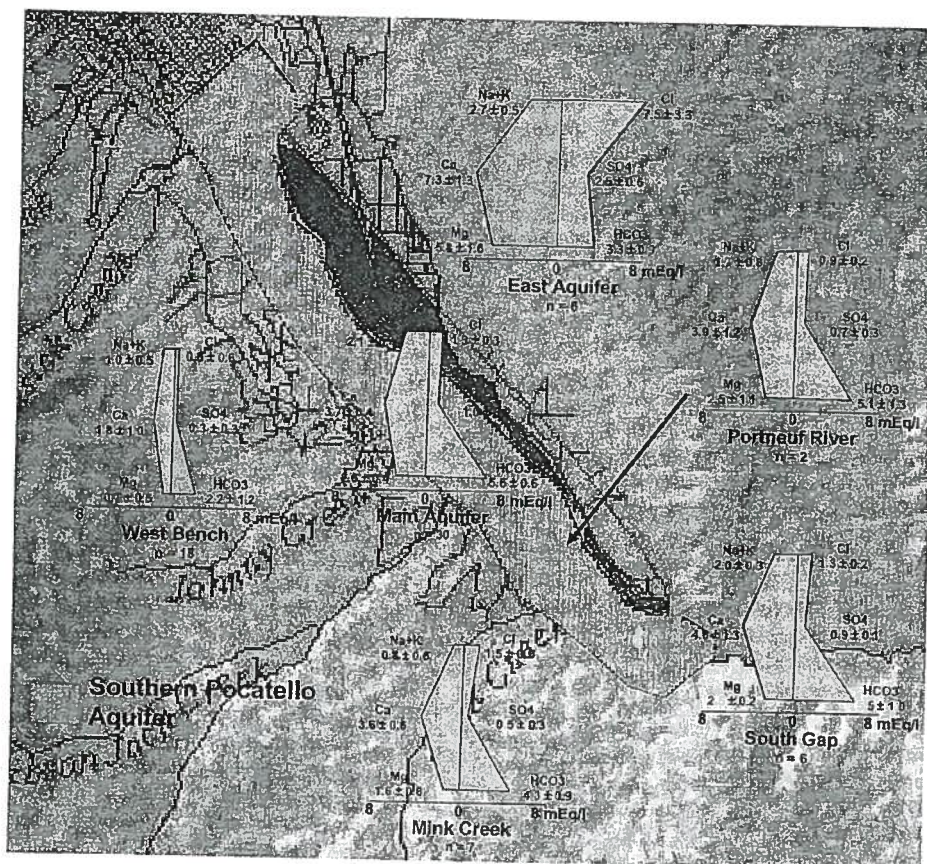


Figure 3-3: Stiff diagrams displayed by locations of water types. Major ion values are in milliequivalents per liter (mEq/l) calculated from the median concentrations for all the samples collected in each area. Note the similarity of water chemistry between the main aquifer and waters south of the Portneuf Gap and Portneuf River. West Bench waters tend to be low in TDS, with Mink Creek waters showing relatively elevated concentrations of salts. Eastern aquifer waters have the highest salt concentrations.

3.2 Water Quality Trends

Figures 3-4(a) through (e) show correlations of hardness (CaCO_3), Ca, Mg, Na, Si and their relationship to Cl. Evident in these plots is the consistently increasing proportions of all constituents relative to chloride. Sodium concentrations in the eastern aquifer do not increase in the same proportion, possibly reflecting cation exchange

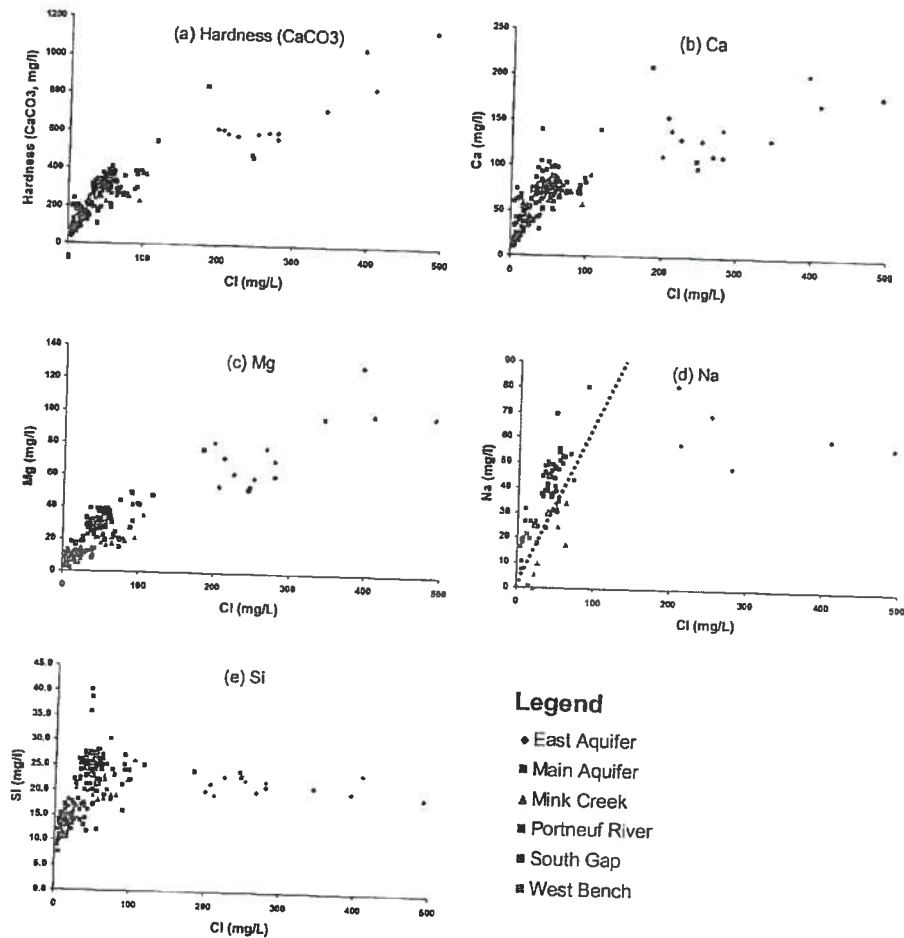
processes (Figure 3-4(d)). The dashed line in Figure 3-4(d) represents the expected sodium trend for the addition of pure NaCl, which indicates a deficit of sodium in the eastern aquifer.

Silica concentrations in most waters plateau at concentrations of 20 to 30 mg/l (Figure 3-4(e)). Although this has not been tested by chemical equilibrium calculations, silica is likely buffered by precipitation reactions due to Si saturation with respect to amorphous silica or clay minerals in these waters.

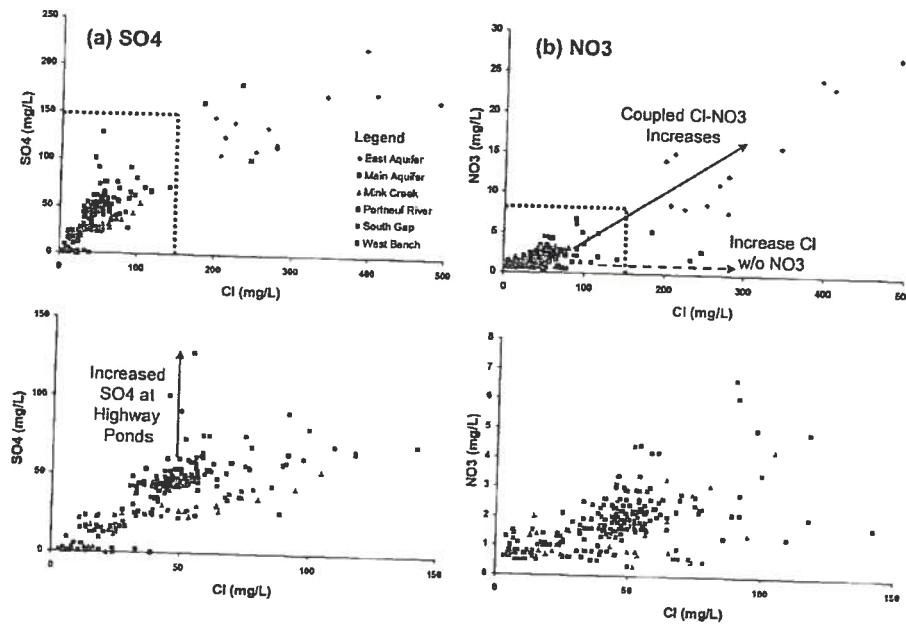
Figures 3-5(a) and (b) display correlations of chloride with sulfate and nitrate respectively, indicating a strong correlation among these constituents.

Several water samples collected in an area known as the Highway Ponds (section 2.2.1 and Figure 2-1) were elevated in sulfate (expanded scale, Figure 3-5(a)). An investigation of this sulfate contamination indicated that road aggregate stockpiled in the area was the probable source (Meehan and Welhan, 1994, see appendix D). Leach tests performed on the aggregate revealed that sulfate, calcium, and potassium were the major soluble components. Analyses of a groundwater sample directly down gradient of the stockpile (Location MA-24) showed significant increases in sulfate and potassium levels above background (Location MA-19). Calcium did not show a significant increase, suggesting that its concentration was chemically buffered (e.g. by precipitation or ion exchange).

Several samples from the main aquifer, West Bench, and Mink Creek display a trend of increasing chloride without the coupled increase in nitrate (Figure 3-5(b)). This trend may reflect a contaminant source high in chloride and other salts with a low-nitrate component, or a depletion of nitrate selective to Cl, such as by denitrification (discussed



Figures 3-4 (a) through (e): Plots showing correlations of hardness (CaCO_3), Ca, Mg, Na, Si and their relationship to Cl. Evident in these plots is the consistently increasing proportions of all constituents with chloride. Sodium concentrations in the eastern aquifer do not increase in proportion to other waters, which may reflect cation exchange processes. The dashed line in Figure 3-4(d) represents the expected sodium trend for the addition of pure NaCl, indicating a deficit of sodium in the eastern aquifer. Silica concentrations plateau at high chloride concentrations likely due to saturation with respect to amorphous silica in these waters.



Figures 3-5(a) and (b): Plots showing correlations of chloride with sulfate and nitrate respectively. The boxed areas outlined in the upper plots are shown in expanded views in the lower plots. These plots also indicate a strong coupled correlation of increasing trends for Cl, SO₄, and NO₃-N. Two other trends are also noted. Figure 3-5(a) shows the effects of increased SO₄ concentrations due contamination from road aggregate stockpiled near the Highway Ponds area. Another trend of increasing Cl without coupled nitrate increases can be seen in Figure 3-5(b).

further in Section 5.3.2). Additional sample locations in Mink Creek and the West Bench from the 2004 sampling event (Section 4.1) showed that high-chloride, low- nitrate waters are common in these areas.

With the exception of the high-chloride/low-nitrate type water and high-sulfate water at the Highway Ponds, Figures 3-4 and 3-5 show strong coupled correlations among all major ions in the vast majority of sample sites. The fact that the main aquifer and its recharge waters are all characterized by similar anionic ratios suggests that a common source may be responsible for adding these salts to groundwater.

3.3 Evidence for Contaminant and Recharge Contributions to the Southern Aquifer

Figure 3-6 displays histograms of Cl, SO₄, and NO₃-N for samples collected in the southern aquifer in May through July of 1994. Samples cluster relatively tightly around mean concentrations of Cl, SO₄, and NO₃-N of 53.5 mg/l, 47 mg/l, and 1.9 mg/l respectively. A small number of samples show concentrations at the high and low ranges, indicating impacts from the types of water found in recharge and contaminant sources described in the previous section.

Figure 3-7 shows cross plots of chloride with sulfate and nitrate for all samples collected in the southern aquifer in 1993 and 1994. Shaded and hatched areas indicate locations where impacts from recharge and contaminant fluxes reflect mixing trends.

Figure 3-8 shows the geographical locations of these impacts.

Low-TDS waters are found in wells completed in the Tertiary Starlight Formation (Figure 3-1, locations MA-17, MA-34, PA-2, and PA-7) that defines the base of the southern aquifer (see section 1.4.2 and 1.5.2). These waters are chemically similar to low-TDS waters sampled in the Bannock Range (Figure 3-5), suggesting that the same source is involved. Several samples in the Bannock Range with low TDS are associated with flowing-artesian conditions (locations WB-05, WB-19, WB-21, and WB-23). A paired set of shallow and deep monitor wells (locations PA-7 and PA-8) also shows a small but significant upward vertical hydraulic gradient between the Tertiary sediments and Upper Gravels (CH2MHill, 1994(b)). The strongest evidence for upward vertical migration of low-TDS Bannock Range waters into the base of the municipal aquifer comes from two flowing-artesian wells (Locations MA-100 and MA-115) recently drilled in the southern aquifer and completed in the Tertiary sediments that have low-TDS

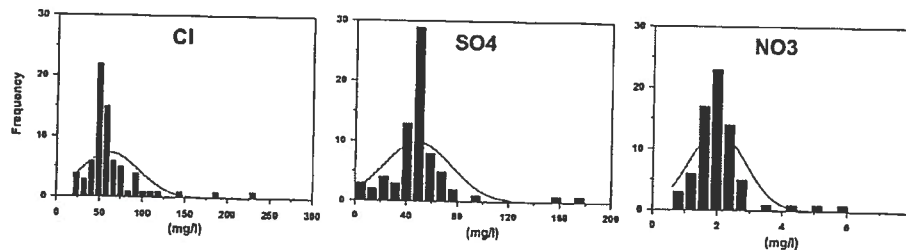


Figure 3-6: Histograms of Cl, SO₄, and NO₃-N concentrations for samples collected in the southern aquifer in May through July of 1994. Samples cluster relatively tightly around the mean concentrations of Cl, SO₄, and NO₃-N at 53.5 mg/l, 47 mg/l, and 1.9 mg/l respectively.

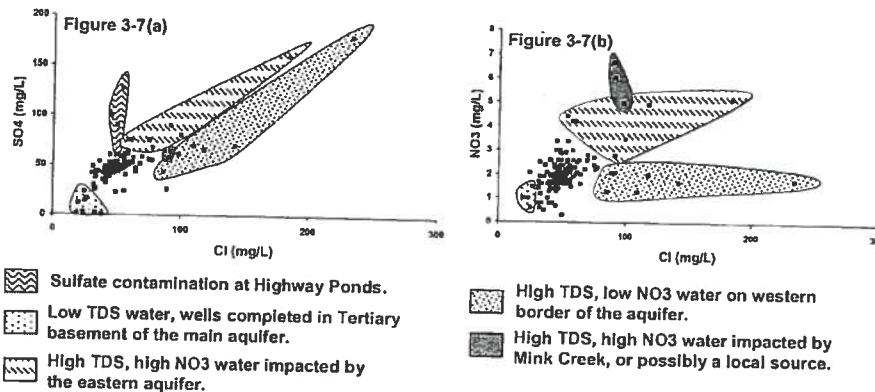


Figure 3-7(a) and (b): Cross plots of chloride with sulfate and nitrate for all samples collected in the southern aquifer in 1993 and 1994. Shaded and hatched areas outline locations where impacts from recharge and contaminant fluxes reflect possible mixing trends. Geographical locations of impacted areas are shown in Figure 3-8.

waters (discussed further in section 4.2).

The above evidence confirms that Bannock Range waters are entering the southern aquifer from below, as well as recharging laterally from the west.

Except for the sulfate contamination at the Highway Ponds, which was due to a localized source of stockpiled road aggregate, elevated salt concentrations tend to occur near the margins of the aquifer, possibly reflecting influence from adjacent recharge

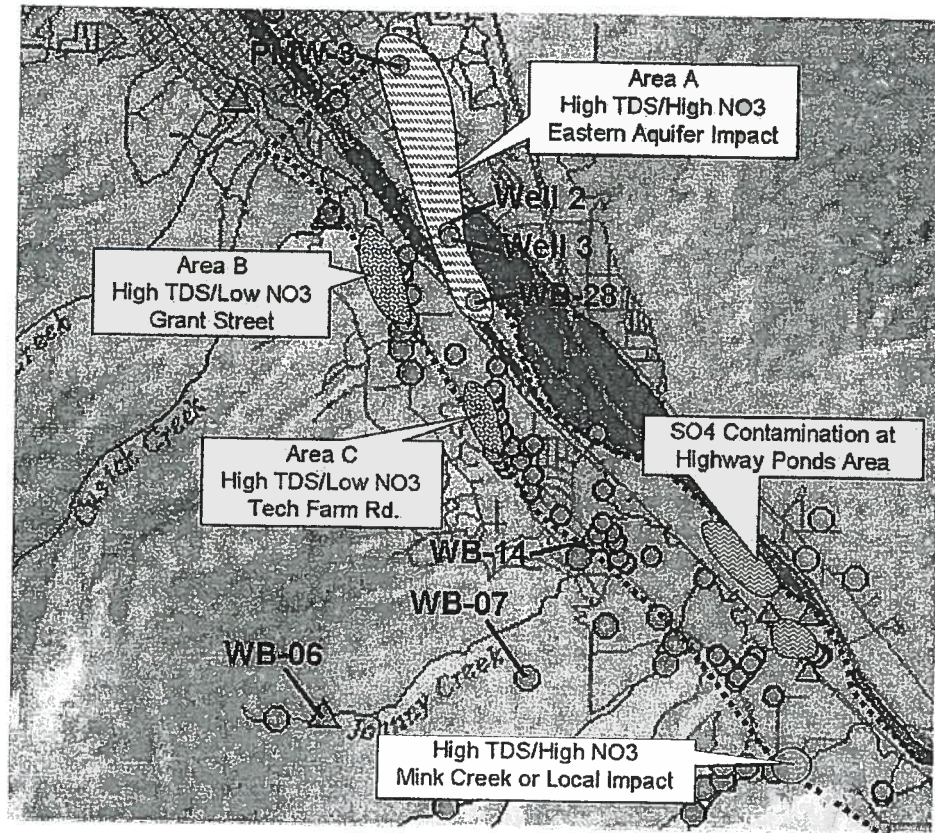


Figure 3-8: Map of the southern aquifer showing areas where groundwater samples in 1993 and 1994 indicated impacts from salt contamination. The coupled trends of increasing Cl, SO₄, and NO₃-N concentrations associated with these locations are shown in Figures 3-7(a) and (b). Location IDs identify municipal wells (Well 2, Well 3, and Well 28) and a monitor well (PMW-3) impacted by high-TDS/high-NO₃ waters from the eastern aquifer. Also identified are samples where high-Cl/low-NO₃ type waters were observed at the head and mouth of Johnny Creek (WB-06 and WB-14 respectively) and a domestic well (WB-07).

waters.

Salt contamination reflected in the coupled increases of Cl, SO₄, and NO₃-N (Figure 3-7 and Area A, Figure 3-8) along the north eastern boundary of the southern aquifer indicates impacts from the eastern aquifer. The area of contamination is defined by three municipal wells (Locations Well 2, Well 3, and Well 28) and one monitor well

(Location PMW-3). Large pumping withdrawals from these municipal wells are likely drawing contaminated waters under the Portneuf Basalt from the eastern aquifer.

Two localized plumes of high-TDS/Low-NO₃ waters were observed in the areas of Grant Street (Area B, Figure 3-8, Locations MA-59, MA-62, MA-63, MA-64, and MA-67) and Tech Farm Road (Area C, Figure 3-8, Locations MA-44, MA-46, MA-49, MA-50, MA-52, MA -53) on the western border of the aquifer. These plumes may result from localized sources of contamination originating near the surface, like road de-icing salts, septic leachate, fertilizers, or animal manure. Another possibility is that they emanate from waters in the West Bench area of Bannock Range that are already elevated in salts. Surface water samples collected from the head and mouth of Johnny Creek (Locations WB-06 and WB-14) and from a domestic well in the Gibson Jack area (Location WB-06), had similar types of high-TDS/low-NO₃ waters (Figure 3-8).

Samples collected from one domestic well (Location MA-06) located in the main aquifer near the mouth of Mink Creek had high-TDS/high-NO₃ type water (Figures 3-7(a) and (b), and 3-8). The source of this contamination may be localized (such as old cattle holding pens located directly adjacent to the well) or from elevated salts and nitrate observed elsewhere in Mink Creek groundwater (Figures 3-4 and 3-5).

Further discussion of potential contamination sources is reserved until 2004 water quality data have been presented.

3.4 Seasonal Variations in Water Chemistry

Time series plots depicting variations in concentrations of Cl, SO₄, and NO₃-N for the four sampling events in 1993 and 1994 are shown for the aquifer recharge source waters in Figures 3-9(a) through (j), and for the southern aquifer in Figures 3-10(a)

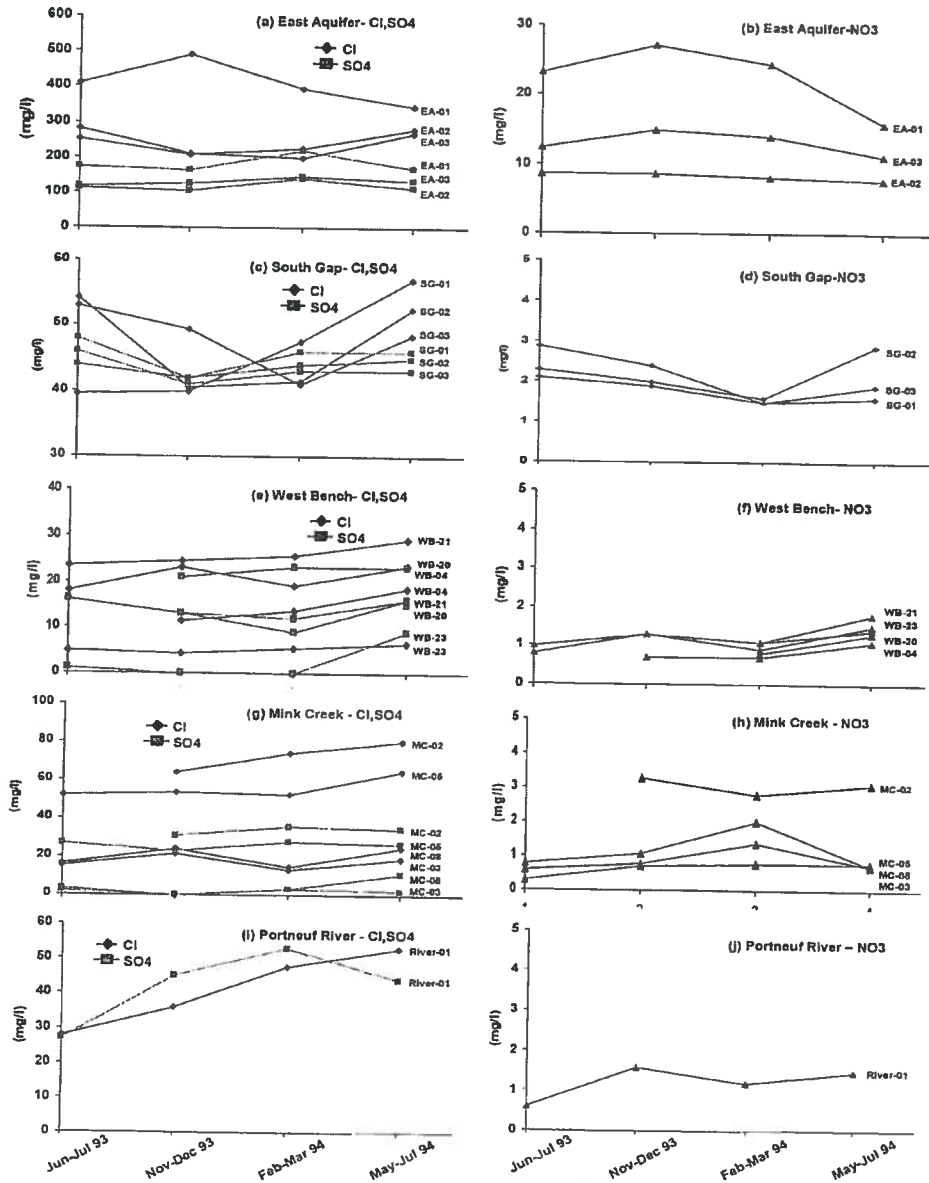
through (c).

The plots do not depict consistent coupled trends between Cl, SO₄, and NO₃-N that one might expect given the correlation of these constituents shown in bivariate plots in sections 3.2 and 3.3. However, there does appear to be a trend of increasing chloride concentrations in most of the sample locations for the period spanning February-March 1994 to May-July 1994. A Student's t-test of the summer 1993 and summer 1994 Cl concentrations in the main aquifer shows that the increase is significant at the 95% confidence interval.

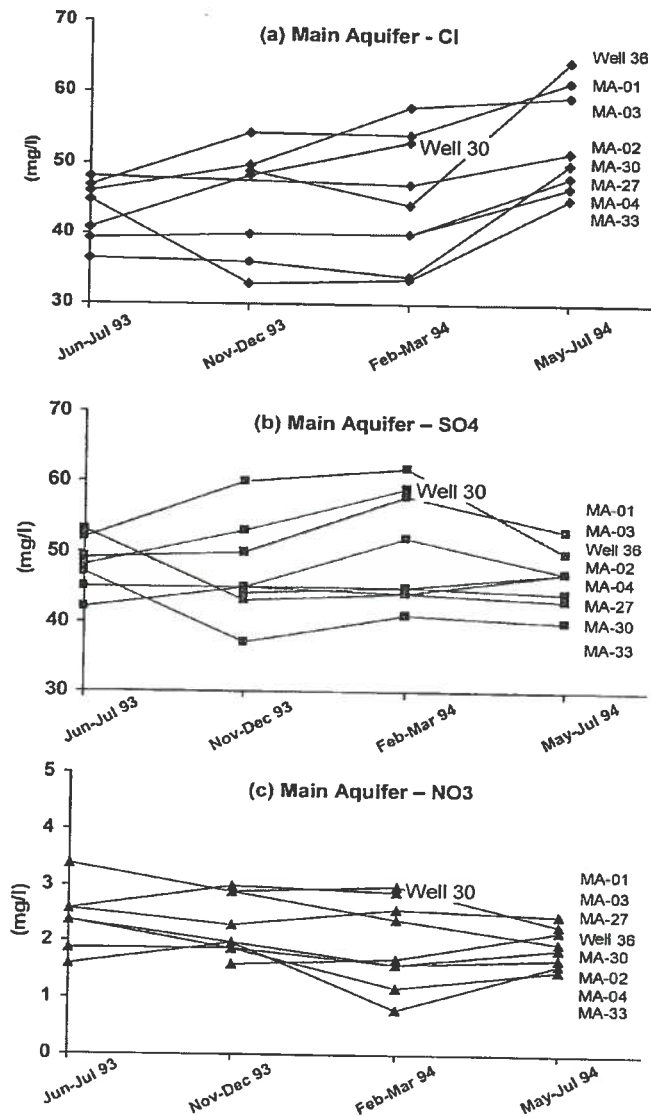
For the same time period, a few of the SO₄ and NO₃-N concentrations also increased slightly, but at most locations concentrations remained steady with a few that decreased. A Student's t-test for the summer 1993 and summer 1994 SO₄ and NO₃-N concentrations showed that SO₄ and NO₃-N did not increase at the 95% confidence interval.

It should be noted that these variations are, for the most part, outside the range of analytical precision. As discussed in Section 2.4, the analytical precision of Cl ranges from about 3 mg/l for concentrations less than 100 mg/l to more than 30 mg/l at concentrations over 250 mg/l. The precision for sulfate is less than 1 mg/l at concentrations less than 70 mg/l to more than 3 mg/l at higher concentrations. Nitrate precision is about 0.15 mg/l at concentrations less than 10 mg/l.

The lack of clear ionic correlations in the time-series plots does not discredit the assumption that a single source may be responsible for the coupled trends of Cl, SO₄, and NO₃-N in LPRV groundwater. It is possible that a separate source of chloride may be involved (such as from road de-icing salt or brines discharged from water softeners), but,



Figures 3-9(a) through (j): Time series plots depicting variations in concentrations of Cl, SO₄, and NO₃-N in aquifer recharge sources for the four sampling events in 1993 and 1994. Location IDs for sample locations are displayed on the right side of each plot. There is no consistently clear coupled trends between Cl, SO₄, and NO₃-N. Cl appears to show a trend of increasing concentrations in most of the sample locations for the period that spans from February-March 1994 to May-July 1994.



Figures 3-10(a) through (c): Time series plots depicting variations in concentrations of Cl, SO₄, and NO₃-N in the southern aquifer for the four sampling events in 1993 and 1994. Location IDs for sample locations are displayed on the right side of each plot. A Student's t-test of the summer 1993 and summer 1994 Cl concentrations in the main aquifer shows that the increase is significant at the 95% confidence interval. While there were fluctuations in SO₄ and NO₃-N concentrations remained relatively steady. A Student's t-test for of the summer 1993 and summer 1994 SO₄ and NO₃-N concentrations showed that there was no significant increase at the 95% confidence interval.

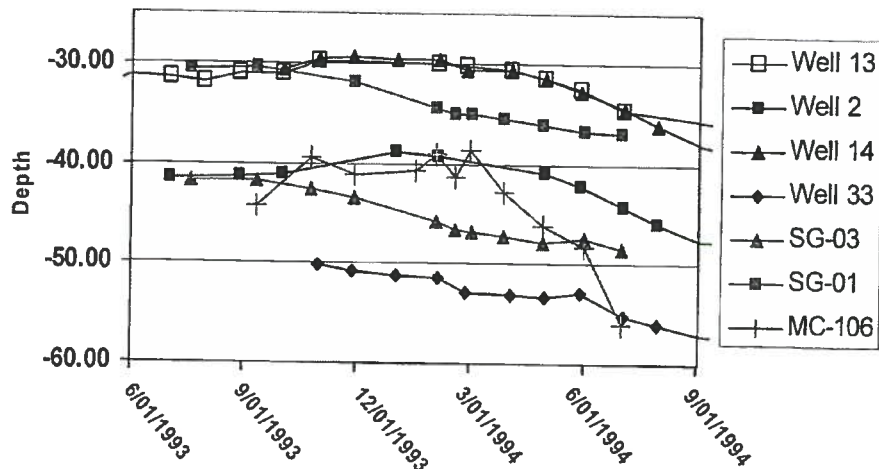


Figure 3-11: Depths to groundwater measured in wells located in the southern aquifer (Locations Well 13, Well 2, Well 14, and Well 33), south of the Portneuf Gap (SG-01 and SG-03) and Mink Creek (MC-106). This figure shows that from March through July 1994, groundwater levels dropped about seven feet in the southern aquifer and about 15 feet in Mink Creek. Water levels south of the Portneuf Gap declined about seven feet from September 1993 through July 1994.

the lack of a clear pattern and the magnitude of the decoupled variations shown in Figures 3-9 and 3-10 suggest that noise may preclude any conclusion from being drawn from such relatively short time series plots.

The increases in Cl concentration may also reflect variations in the strength of groundwater recharge. Figure 3-11 displays depths to groundwater measured in wells located in the southern aquifer (Locations Well 13, Well 2, Well 14, and Well 33), south of the Portneuf Gap (SG-01 and SG-03), and Mink Creek (MC-106). This figure shows that from March through July 1994, groundwater levels dropped about seven feet in the southern aquifer and about 15 feet in Mink Creek. Water levels south of the Portneuf Gap declined about seven feet from September 1993 through July 1994. Declining water levels were in part a response to an increase of pumping withdrawals from wells during

the spring and summer of 1994, but also by below-average snow pack the previous winter (Nov. 1993 through Mar. 1994; Welhan et al., 1996).

The snow pack in November 1992 through March 1993 was above average (Welhan et al., 1996). The lower concentrations of Cl observed in June-July 1993 may reflect dilution by a larger flux of low-Cl recharge water. In contrast, snowpack (and hence recharge) in the spring of 1994 was much smaller decreases in recharge may have amplified any contributions from contaminant sources.

Chapter 4: 2004 Water Quality Results

4.1 2004 Water Quality Trends

Water samples in the summer of 2004 were collected to determine changes in chemistry since 1994 and to evaluate whether areas with high densities of septic tanks may be affecting groundwater quality. The 2004 sample locations are shown in Figure 4-1. Analytical results of water analyses identified by their location ID in this chapter can be found in Appendix B. Sample locations specifically mentioned in this chapter are shown in Figure 4-1.

Forty southern aquifer wells including almost all of the locations sampled in 1993-1994 were re-sampled. Additional sample locations not previously sampled included 22 wells in the southern aquifer, 32 in Mink Creek, and 3 in Gibson Jack (Wild Horse Divide subdivision).

Samples were analyzed for Cl, SO₄, and NO₃-N with a split set of 46 samples sent to a commercial laboratory for analyses of Cl, SO₄, NO₃-N, Ca, and total hardness (as CaCO₃) (see section 2.4).

Figures 4-2(a) and (b) show correlations of the same strong coupled correlation of chloride with sulfate and nitrate, respectively, that were observed in the 1993-1994 sampling (see Section 3.2). The increased number of sample locations in Mink Creek and Gibson Jack revealed that more of the groundwater in these areas is elevated in salts, as indicated by the relatively high Cl, SO₄, and in some cases high NO₃-N concentrations. As discussed in Section 3.2, the 1993-1994 analytical results suggested that most of the waters in the West Bench tributaries of the Bannock Range were low in TDS with very few locations having elevated salt concentrations.

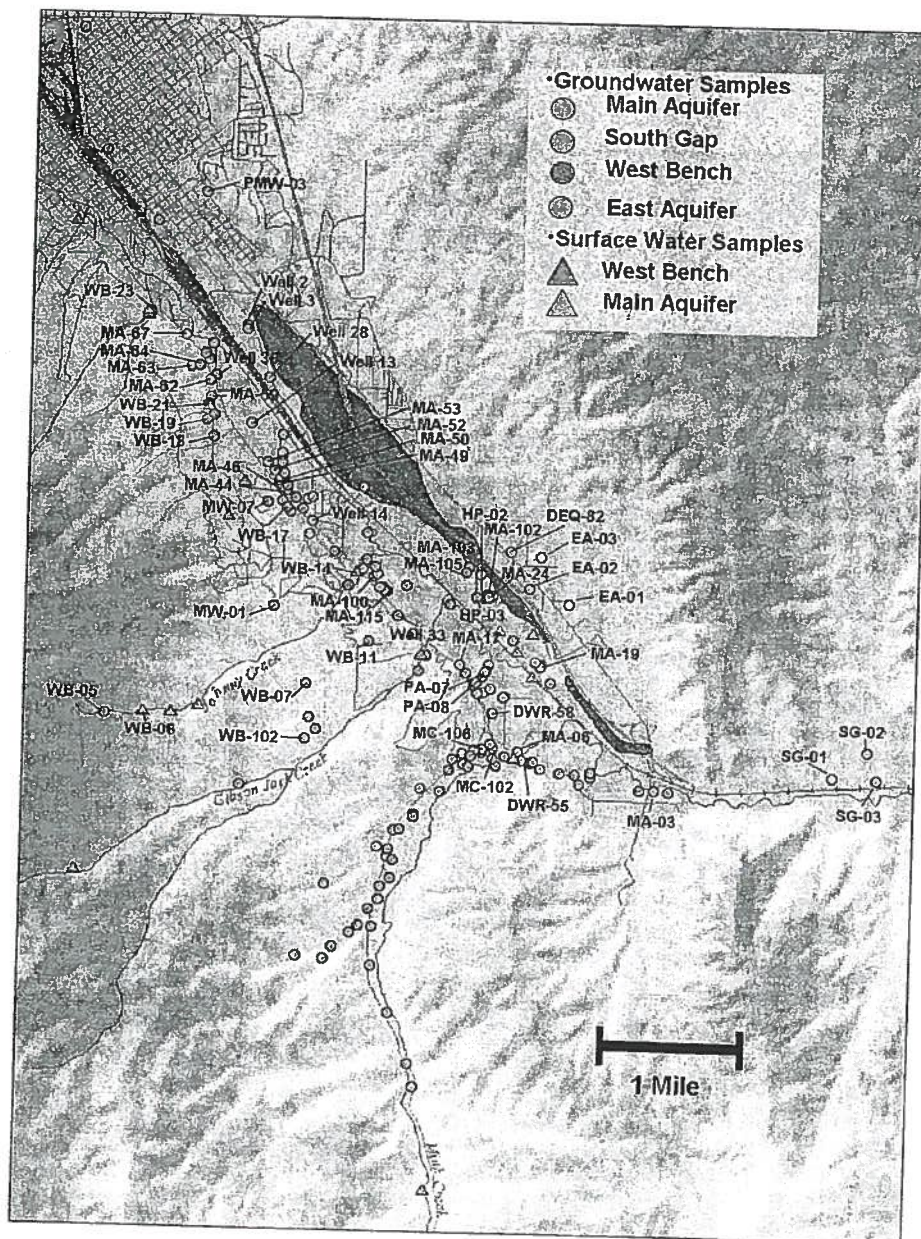
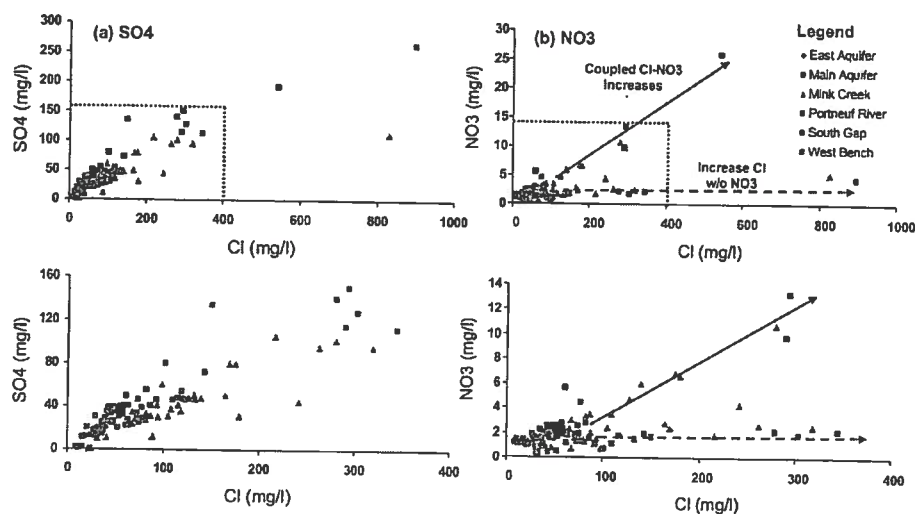
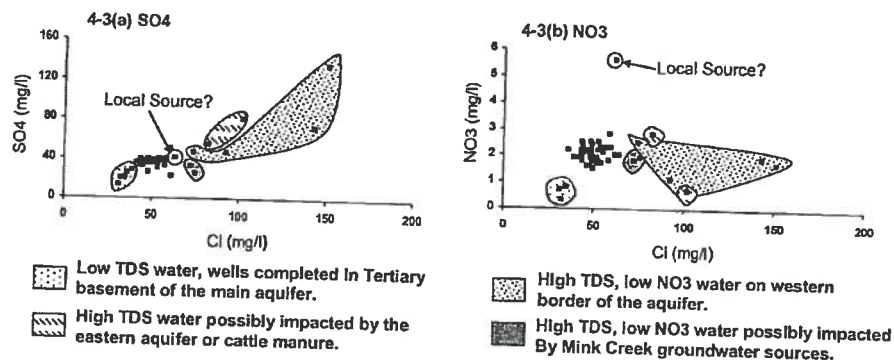


Figure 4-1: Locations of water samples collected in 2004. Sample locations referred to in the text of this thesis are identified by their Location IDs.



Figures 4-2(a) and (b): Correlations of chloride with sulfate and nitrate for water samples collected in 2004. The boxed areas in the upper plots are shown in expanded views in the lower plots. These plots show the same strong coupled correlation of increasing trends of these constituents that was seen in the 1993-1994 trend analyses (see section 3.2). The solid arrows in Figure 4-2 (b) represent coupled increases of Cl and NO₃-N. The dashed arrows represent increases in Cl concentrations without NO₃-N.

The trend of high-chloride/low-nitrate type waters in the Mink Creek and West Bench areas is more obvious in 2004 (compare Figures 3-5(b) and Figures 4-2(b)). Sixteen samples had concentrations of chloride greater than 100 mg/l with nitrate levels at about 2 ± 1 mg/l. Two samples (locations MC-102 and WB-102) had chloride concentrations greater than 800 mg/l with nitrate levels about 5 mg/l (Figure 4-2(b)). These results indicate there may be a contaminant source high in chloride and sulfate but not nitrate, or a depletion of nitrate, possibly by denitrification (discussed further in section 5.3.2).



Figures 4-3(a) and (b): Cross plots of chloride with sulfate and nitrate respectively for all samples collected in the southern aquifer in 2004. Shaded and hatched areas outline sample locations where impacts from recharge and contaminant fluxes reflect mixing trends similar to those shown in Figures 3-7(a) and (b).

4.2 Evidence for Mixing in the Southern Aquifer

Figures 4-3(a) and 4-3(b) are cross plots of chloride with sulfate and nitrate, respectively for all samples collected in the southern aquifer in 2004. Shaded and hatched areas highlight impacts from possible recharge and contaminant fluxes and reflect mixing trends similar to those shown in Figures 3-7(a) and (b).

Absent from Figure 4-3(a) are high-sulfate waters that were observed in 1993 at the Highway Ponds area possibly due to stockpiled road aggregate (Appendix D). This stockpile was removed following the 1993-1994 sampling. Samples collected in the Highway Ponds area in 2004 showed no significant sulfate contamination (locations HP-02, HP-03, MA-102, MA-104), corroborating the inference that the stockpiled aggregate was responsible.

Wells sampled in the areas of Grant Street and Tech Farm Road (locations MA-44, MA-46, MA-59, MA-62, and MA-67) indicated that these two localized plumes of high-TDS/low-NO₃ waters on the western margin of the aquifer are still present and with

similar levels of contamination. One domestic well (location MA-62) on Grant Street sampled in 2004 had much lower levels of chloride and sulfate. Chloride and sulfate dropped from 77 mg/l and 54 mg/l, respectively, to 58mg/l and 32 mg/l. However, this well was deepened by sixty feet in 2001. If the lower levels of contamination reflect this, then the source of contamination in these wells is most likely near the surface. Evidence for a near-surface contaminant source also comes from two of the wells at Grant Street and four of the wells on Tech Farm Road that were contaminated in 1994 and that are now dry due to lower groundwater levels. These wells were producing water from near the groundwater table in 1994, and therefore were reflecting near-surface sources of contamination.

The sample identified as a localized source in Figures 4-3(a) and (b) (Location MA-03) is the only well sampled in the main aquifer in 2004 that had a relatively high concentration of nitrate (5.6 mg/L). This well is located within a horse corral and may be contaminated due to a combination of an animal manure source and poor well construction.

Other wells reflecting impacts from high-TDS eastern aquifer water in Figures 4-3(a) and (b) may also be affected by animal waste (Locations MA-103 and MA-105). They are within close proximity to the boundary of the eastern aquifer, but are also in areas where residents are raising cattle. It is difficult to identify a source, though, because nitrate was not elevated as would be expected from animal manure or eastern aquifer septic impacts.

As was found in 1993/94, most wells completed in the Tertiary Starlight Formation exhibit low-TDS type waters that reflect recharge from the Bannock Range

(see Section 3.3). Two wells recently drilled in the southern aquifer and completed in the Tertiary sediments are flowing-artesian wells and have low-TDS type waters (Locations MA-100 and MA-115). These wells are in close proximity to each other (Figure 4-1) and encounter flowing water under artesian pressure between depths of 250 and 280 feet. This provides the strongest direct evidence that low-TDS Bannock Range waters are recharging the aquifer from below the Upper Gravels.

4.3 Mink Creek Water Quality

Figure 4-4 shows the locations of the thirty seven groundwater samples and two surface water samples collected in the Mink Creek drainage in August and September of 2004. Also shown are the inferred locations of septic tanks in this area. Additional sampling was conducted in the Mink Creek area in 2004 to evaluate possible impacts on groundwater quality from septic effluent. Residences in Mink Creek are not served by the Pocatello municipal water system and must rely on local residential and community wells for drinking water.

An abandoned poultry farm located in the Mink Creek drainage (the old Frazier's egg farm, Figure 4-4) was also considered to be a potential contaminant source in this area. This farm went out of business in 1998, but residual chicken manure in the soil could still be leaching nitrates and other salts to groundwater.

Groundwater samples were collected along the entire reach of the drainage, from below the U.S. Forest Service boundary, at the first residential properties, to the valley's terminus at the boundary of the southern aquifer. Several groundwater samples were also collected in the tributary drainage of Campbell Creek (Figure 4-4). Surface water samples were collected above the Forest Service boundary and at the mouth of the

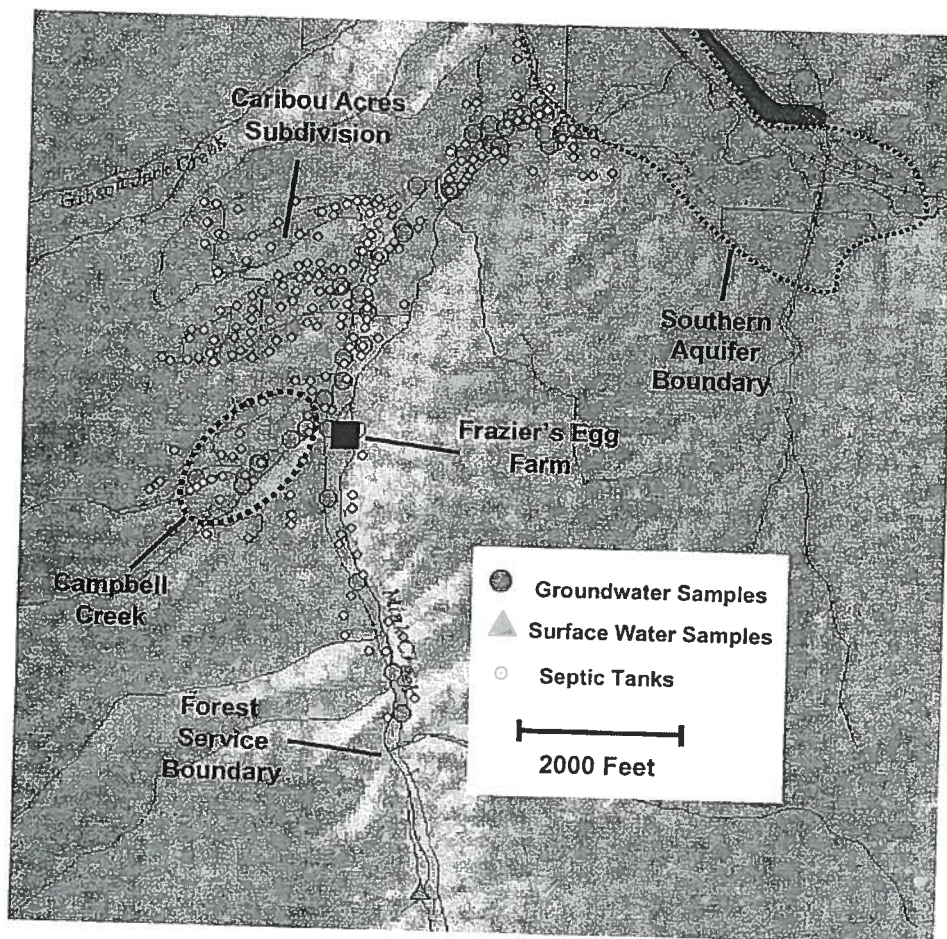
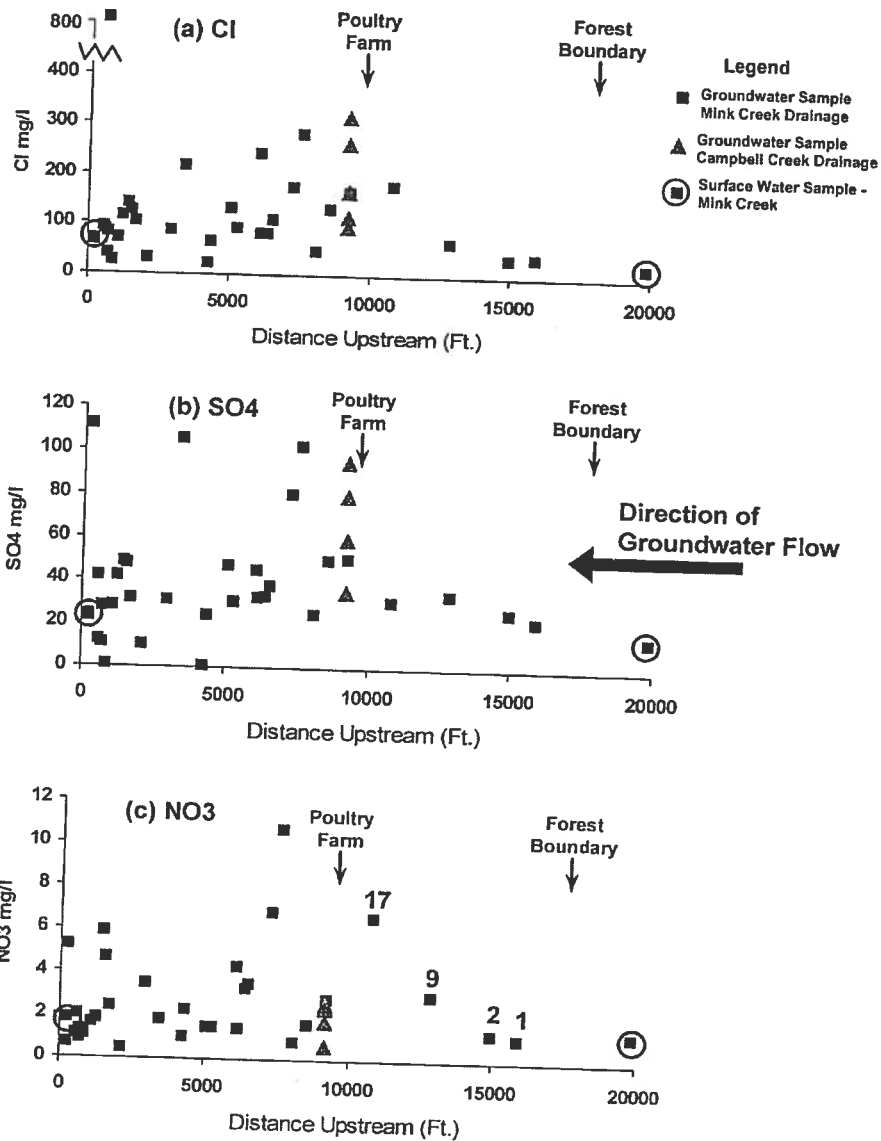


Figure 4-4: Locations of the thirty seven groundwater samples and two surface water samples collected in the Mink Creek drainage in 2004. Also shown are the locations of septic tanks which number greater than 200 in this area. Frazier's Egg Farm went out of business in 1998, however, residual animal manure could still potentially leach nitrates and other salts to groundwater.

drainage near the boundary of the southern aquifer.

Figures 4-5(a), (b), and (c) show the concentrations of Cl , SO_4 , and $\text{NO}_3\text{-N}$ respectively in water samples collected in the Mink Creek drainage. The surface water samples show an increase in Cl , SO_4 , and $\text{NO}_3\text{-N}$, in the downstream direction, as do



Figures 4-5(a), (b), and (c): Concentrations of Cl, SO₄, and NO₃-N in water samples collected in the Mink Creek drainage plotted for each sample location's distance upstream from the southern aquifer boundary. These plots reflect changes in chemistry along the groundwater flow path. Groundwater samples collected in the Campbell Creek tributary are designated with separate symbols. The cumulative number of septic tanks above each sample location are shown above sample locations upgradient of the poultry farm in plot (c).

groundwater samples between the Forest Service boundary and the old poultry farm. Septic tank effluent is the only likely source for the observed increases. There are no other agricultural or animal manure sources in this reach of the creek. Downgradient of the poultry farm, salt concentrations show much greater variability along the length of the valley. No clear trends were evident in the lower part of the drainage below the poultry farm. There was no statistical significance in any regression trend in Cl, SO₄, or NO₃-N along the groundwater flow path in the lower part of the drainage.

Above the poultry farm, Student's t-tests indicated that the observed increases of Cl and NO₃-N were statistically significant at the 95% confidence interval. The linear regression performed for sulfate did not represent a statistically significant trend with distance, though this is based on only four data points.

All of the samples collected in the Campbell Creek tributary drainage were high-Cl, SO₄/low-NO₃ type of waters.

4.3.1 Autocorrelation of Mink Creek and Main Aquifer Chemistry

The large dispersion in groundwater chemistry downgradient of the poultry farm may be a reflection of fracture-controlled groundwater flow. Domestic well logs indicate that very few wells in the Mink Creek drainage are completed solely in alluvium; most are completed in bedrock, with a few in both bedrock and alluvium.

In fractured media, permeability can vary by orders of magnitude over short distances, and the three-dimensional configuration of transmissive fractures and fracture zones can exert complex controls on fluid migration pathways. The heterogeneity of the fractured medium would have considerable impact on the fate and transport of contaminants introduced to groundwater.

Variogram plots were constructed to compare the spatial autocorrelation of Cl, SO₄, and NO₃-N in Mink Creek groundwater relative to their spatial distribution in the southern aquifer. Wells in the southern aquifer are completed in alluvium, where transport characteristics should be more homogeneous than a fracture-dominated flow system. Mink Creek and southern aquifer groundwater show similar correlations among Cl, SO₄, and NO₃-N (see section 3.1 and 4.1). If these contaminants are being introduced from similar sources in both areas (i.e. septic tanks), then differences in their spatial autocorrelation properties might be expected due to their different scales of transport heterogeneity (fracture vs. porous intergranular).

All semivariograms were created as omnidirectional statistics (independent of direction) using the methodology of Section 2.7. Both spherical and exponential models were evaluated in describing the spatial autocorrelation structure. When comparing a chemical constituent in Mink Creek groundwater to the southern aquifer, the same model type was used for both. Variogram statistics were calculated using the semivariogram, standardized semivariogram, covariance and correlogram estimators to ensure that the nature and structure of the autocorrelation was properly inferred. Variogram results are summarized in Appendix E.

Table 4-1 summarizes the variogram ranges estimated at different lag sizes. The variogram range values for nitrate and sulfate tend to be lower in Mink Creek, but are similar for Cl.

The higher range values observed in the southern aquifer for nitrate and sulfate would be expected given a more homogeneous alluvial aquifer in comparison to fractured flow in Mink Creek. That is, the variability of water chemistry should be greater at

smaller intersample distances in a fractured media.

The fact that chloride's autocorrelation may be weaker in the main aquifer suggests that other, perhaps localized, sources of chloride may be impacting groundwater in the main aquifer. This was also hypothesized in section 3.4 to explain some seasonal changes in chloride chemistry that are de-coupled from sulfate and nitrate.

	Mink Creek	Main Aquifer
Chloride	1000 ± 200	800 ± 200
Sulfate	800 ± 200	> 2500
Nitrate	< 600	1600 ± 400

Table 4-1: Best estimates of autocorrelation ranges for Cl, SO₄ and NO₃-N concentrations in Mink Creek and the main aquifer. Range values for nitrate and sulfate tended to be greater in the southern aquifer; Range values for chloride were similar in both Mink Creek and the southern aquifer. Variogram results are summarized in Appendix E.

4.4 Temporal Changes in Water Quality from 1994 – 2004

4.4.1 Southern Aquifer and Groundwater South of the Portneuf Gap

Forty-two locations in the southern aquifer were sampled in both 1994 and 2004. Figures 4-6(a), (b), and (c) summarize the changes in groundwater chemistry as a function of each location's spatial distance from the Portneuf Gap, including three sample locations south of the Portneuf Gap (SG-01, SG-02, and SG-03). Shaded areas represent the 95% confidence level of analytical precision for each constituent (see Section 2.4). Measurements outside of these shaded areas represent a potentially significant temporal change.

Figure 4-6(a) shows that Cl differences at most locations lie within the 95% confidence level of analytical precision, and equal numbers of positive and negative

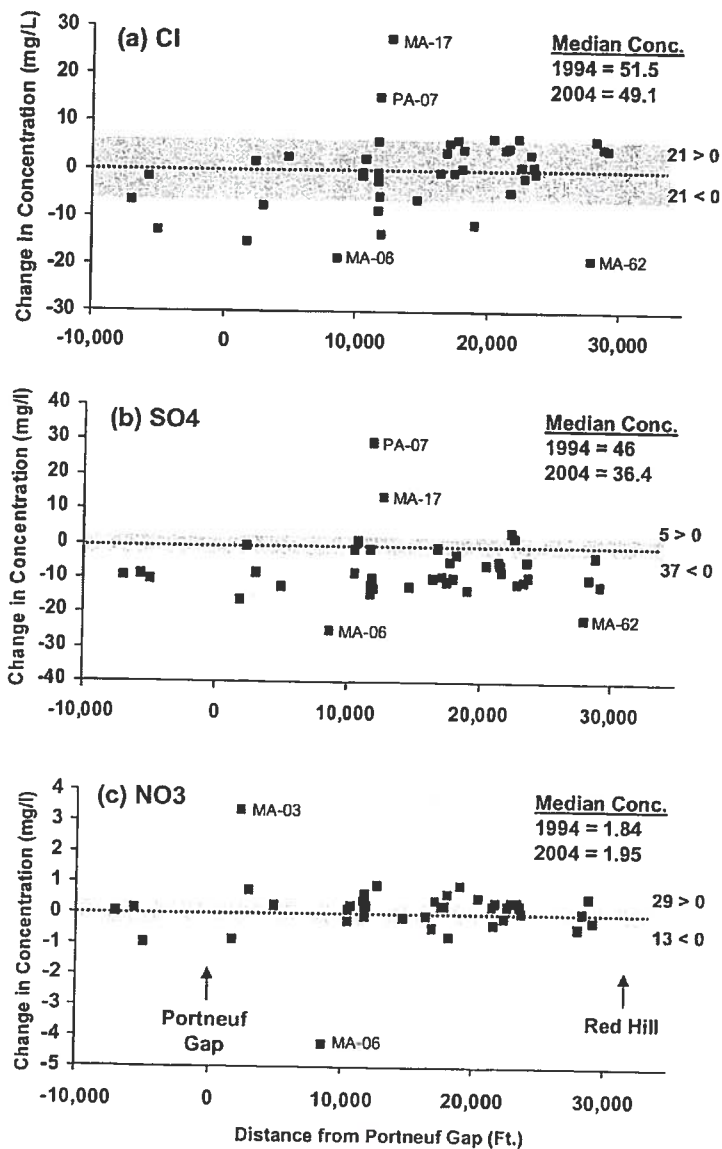
temporal shifts. The median Cl concentration of all samples in 1994 (51.5 mg/l) and 2004 (49.1 mg/l) are correspondingly similar. Most locations also showed no significant change in NO₃-N concentrations (Figure 4.6(c)), although twice as many locations showed positive shifts, with the median concentration increasing only slightly from 1.84 in 1994 to 1.95 in 2004.

In contrast, sulfate concentrations decreased on average by about 10 mg/l at most locations (Figure 4-6(b)). The median concentration changed from 46 mg/l in 1994 to 36.4 mg/l in 2004, and negative shifts outnumbered positive shifts ten to one.

Considering that most locations showed no discernible changes in Cl or NO₃-N concentrations but that SO₄ levels decreased by about 10 mg/l, suggests that impacts to groundwater may be attributable to different sources. The decoupled nature of changes in Cl and SO₄ has been suggested on the basis of seasonal variations (see section 3.4) and in autocorrelation behavior (see Section 4.3.1).

However, another possibility for the decrease in sulfate concentration must be considered. Sulfate standards were made by diluting a 500 mg/l SO₄ HACH standard solution with laboratory deionized water to concentrations of 20 mg/l, 40 mg/l, and 60 mg/l. These standards were made in the ISU geochemistry lab with no way to ensure their absolute accuracy. In contrast, samples collected in 2004 had 45 duplicate analyses performed by an outside laboratory, adding a level of quality assurance to the 2004 data that was unavailable in 1994 (see Section 2.4).

Two wells completed in the Tertiary sediments beneath the southern aquifer, MA-17 and PA-07, showed significant increases in Cl and SO₄ concentrations (Figures 4-6(a) and (b)). In 1994, these wells had low concentrations of Cl and SO₄, typical of wells



Figures 4-6(a), (b), and (c): Changes in groundwater chemistry from 1994 to 2004 displayed for each location's spatial distance from the Portneuf Gap. Including the three sample locations south of the Portneuf Gap (SG-01, SG-02, and SG-03). Shaded areas represent the 95% confidence level of analytical precision of each constituent. Only measurements outside of these shaded areas represent a potentially significant difference in concentration. The number of apparent positive and negative temporal shifts are shown for comparison.

completed in the Tertiary sediments. Monitor well PA-07 was sampled with a thief sampler in 1994 and 2004. Well MA-17 was sampled with a thief sampler in 1994, but the well has since had a pump installed for irrigation use. Pumping withdrawals in the well could now be pulling additional water from the overlying southern aquifer, accounting for the increased concentrations observed in 2004.

MA-62 is a domestic well located in the Grant Street plume that had elevated Cl and SO₄ in 1994. This well was deepened sixty feet in 2001 and now shows much lower salt concentrations suggesting a greater contribution from West Bench waters and/or diminished near-surface influence (see Section 4.2).

Well MA-06 showed significant decreases in Cl, SO₄ and NO₃-N. MA-06 is a domestic well located adjacent to abandoned cattle holding pens and had elevated salts and nitrate in 1994, possibly due to a local source (see Section 3.3 and Figure 3-8). This source may no longer be impacting local groundwater in 2004.

The increased NO₃-N observed in MA-03 (Figure 4-6(c)), may also represent impacts from a local source. This well is located within a horse corral and is possibly contaminated by animal manure (see section 4.2 and Figures 4-3(a) and (b)).

4.4.2 West Bench and Mink Creek Groundwater

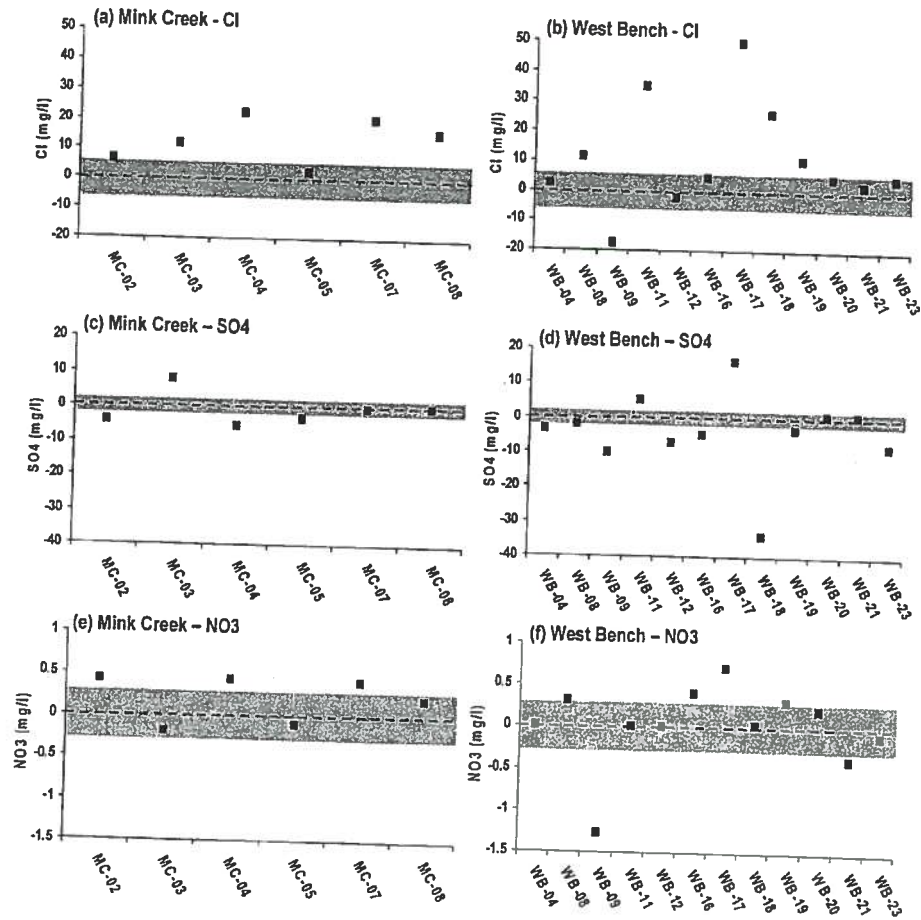
Figures 4-7(a) through (f) display graphs representing difference in Cl, SO₄, and NO₃-N concentrations in West Bench and Mink Creek groundwater from samples collected in 1994 and 2004. The graphs show that higher Cl concentrations were observed in 2004 at most sample locations, whereas SO₄ levels tended to be lower at many locations, similar to the trend observed in the southern aquifer (Section 4.5.1).

Nitrate concentrations in 2004 were only slightly higher or similar to the levels

found in 1994.

4.4.3 Eastern Aquifer Groundwater

The three wells sampled in the eastern aquifer (Locations EA-01, EA-02 and EA-03) in 1993 showed no significant changes in Cl, SO₄, or NO₃-N between 1994 and 2004.



Figures 4-7 (a) through (f): Changes in groundwater chemistry from 1994 to 2004 in West Bench and Mink Creek groundwater. The graphs show that at most sample locations higher Cl concentrations were observed in 2004, while SO₄ levels tended to be lower at many locations. Nitrate concentrations in 2004 were also slightly higher or similar to the levels found in 1994. Shaded areas represent the 95% confidence level of analytical precision for each constituent.

Chapter 5: Discussion

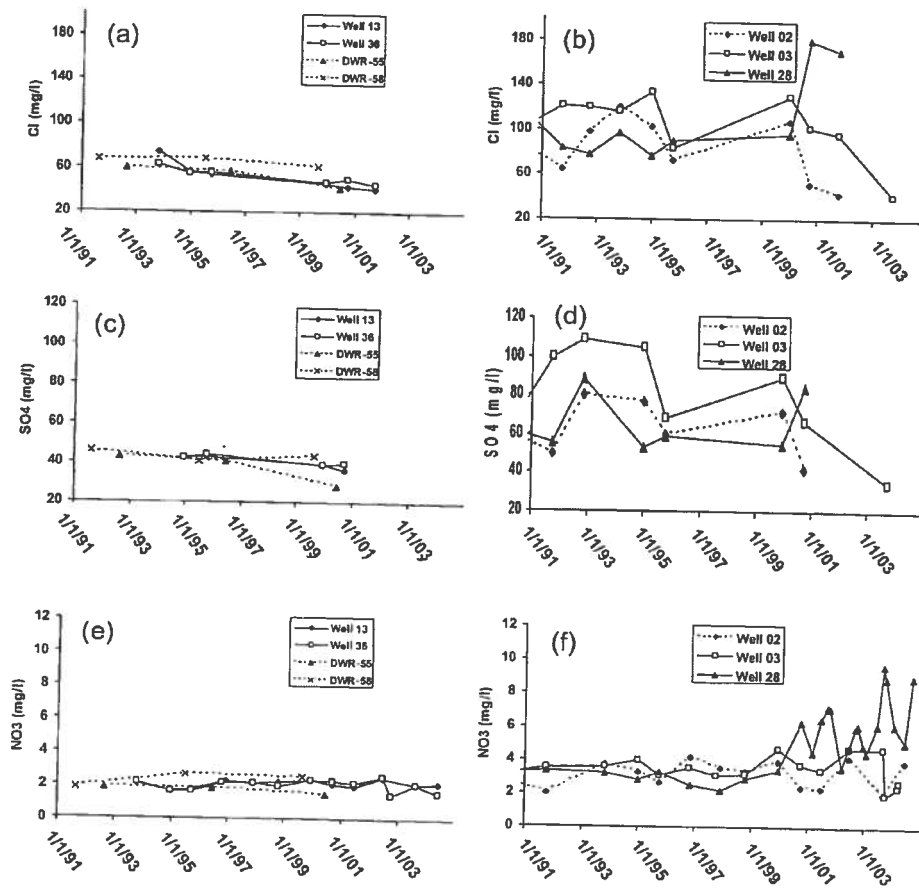
5.1 Temporal Trends in the Southern Aquifer

5.1.1 Decreasing Salt Trends

Temporal trends of chemistry in the southern aquifer have been monitored in just a few wells with any frequency. These include municipal wells 02, 03, 13, 28, and 36, and two domestic wells monitored by IDWR (DWR-55 and DWR-58). Concentrations of Cl, SO₄, and NO₃ in these wells from 1991 through 2004 are summarized in Figures 5-1 (a) through (f). Note that wells 02, 03, and 28 are likely impacted by seepage of contaminated water in the eastern aquifer (see Section 3.3), whereas wells 13, 36, DWR-55 and DWR-58 are not.

Wells 13, 36, DWR-55 and DWR-58 are located along the western margins of the southern aquifer (Figure 4-1). Figures 5-1(a) and (c) display decreasing trends of Cl and SO₄ for these wells; nitrate levels remained fairly steady at background concentrations (2 mg/l; Fig. 5-1(e)), chloride levels decreased by 10 to 15 mg/l between 1994 and 2001 and sulfate concentrations decreased by 4 to 14 mg/l.

The magnitude of decreasing sulfate concentrations (4-14 mg/l observed in wells 13, 36, and DWR-55) is consistent with the decrease observed in samples collected for this thesis between 1994 and 2004 (section 4.4.1, Figure 4-6(b)), where sulfate appeared to decrease on average by about 10 mg/l. The same data showed the median concentration of chloride in the southern aquifer decreasing by 2.4 mg/l (Figure 4-6a, Section 4.4.1)) between 1994 and 2004. However, decreases between 6 to 20 mg/l were observed in 10 wells (Figure 4-6(a)), and are comparable to the 10 to 15 mg/l decreases seen in the wells of Figure 5-1(a).



Figures 5-1 (a) through (f): Temporal trends of Cl, SO₄, and NO₃ in wells in the southern aquifer which include municipal wells 02, 03, 13, 28, and 36, and two domestic wells monitored by IDWR, DWR-55 and DWR-58. Wells 13, 36, DWR-55 and DWR-58 are located along the western margins of the southern aquifer and seem to show decreasing trends of Cl and SO₄. Wells 02, 03, and 28 are located along the eastern boundary of the southern aquifer. The higher concentrations of Cl, SO₄ and NO₃ observed in these wells indicate impacts from groundwater in the eastern aquifer contaminated with salts and nitrate.

The steady background nitrate concentration of about 2 mg/l in Figure 5-1(e) is consistent with that observed in the 1994 - 2004 comparison shown in Figure 4-6(c).

It is difficult to hypothesize a likely cause for the Cl and SO₄ decreases observed over the past decade. Poor analytical precision of the 1994 sulfate analyses was posed as a possible explanation, but the sulfate trends shown in Figure 5-1(c) suggest that the decreases may be real.

Anaerobic reduction of SO₄ could not account for the decrease in Cl, which is unaffected by redox chemistry. Furthermore, nitrate would be reduced before SO₄ due to its higher energy yield (McQuillan, 2004).

Figures 4-6(a) and (b) show that the three wells sampled south of the Portneuf Gap in 1994 and 2004 all showed decreases in sulfate, and two showed significant decreases in chloride, indicating that recharge waters from the upper Portneuf Basin may have been involved. The Portneuf River is known to have gaining and losing reaches in the upper Portneuf Basin (Barton, 2004). Groundwater levels in the southern aquifer and in the City of Inkom (10 miles south of the Portneuf Gap) have dropped 21 feet since 1997 due to drought conditions (see Appendix C). As groundwater levels decline, more of the Portneuf River upstream of the LPRV may become a losing stream. Sulfate and Cl concentrations in the Portneuf River tend to be lower than in the southern aquifer, especially during high river stages (see Section 6.2.3, Figure 6-9). More surface water may contribute to groundwater recharge during high river stage and thus dilute groundwater chemistry. Nitrate concentration in the river appears to be independent of river discharge, which could also explain why nitrate levels in groundwater have remained fairly steady.

At high river stage both chloride and sulfate concentrations are around 20 mg/l (Figure 6-9). At these concentrations, the ratio of main aquifer water to river water necessary to decrease Cl and SO₄ in the main aquifer by 10 mg/l would be approximately 2 to 1, and the contribution of river water to the southern aquifer would be greater than 2 billion gallons per year. This is highly unlikely given the hydrologic evidence that the Portneuf River is not a significant source of water to the aquifer (see section 1.5.3). However, in the upper parts of the Portneuf River and Marsh Creek drainages, river seepage likely comprises a larger component of groundwater recharge (Barton, 2004; Norvitch, and Larson, 1970) and could have proportionally greater impacts on groundwater chemistry, as evidenced by the similarity of river and aquifer chemistry upgradient of the Portneuf Gap.

5.1.2 Eastern Aquifer Impacts and Trends

Wells 02, 03, and 28 are located along the eastern boundary of the southern aquifer. The higher concentrations of Cl, SO₄ and NO₃ (Figure 5-1(b), (d), and (f)) observed in these wells are consistent with impacts from the eastern aquifer (see sections 1.6, 3.1 and 4.2). The large fluctuations observed in these wells may be related to temporal changes in groundwater chemistry in the eastern aquifer and/or variations in the mixing ratio of eastern aquifer water in these wells. One well in the eastern aquifer, DEQ-82, has been monitored periodically since 1981 and shows large fluctuations in NO₃ concentrations over time. These temporal changes are plotted in Figure 5-2, together with a five year running average of snowpack (as inches of water) in the Wild Horse Divide area in the Bannock Range. There appears to be an inverse relationship between NO₃ concentrations and snowpack, such that during wet periods (1983-1985,

1996-1998) nitrate levels in the eastern aquifer are low, increasing during dry periods (1988-1990, 2000-2001).

The IDEQ has determined that salt and nitrate contamination in the eastern aquifer is due to septic effluent (Winter et al., 1994). The temporal relationship in Figure 5-2 may reflect the dilutional effect due to enhanced groundwater recharge during wet periods. Since the eastern aquifer consists primarily of poorly sorted, silt- and clay-rich sediments (see Section 1.4), its hydraulic conductivity is expected to be much lower than the main aquifer. Hence, its temporal groundwater quality variation would be more sensitive to the balance between septic effluent and groundwater recharge than in the southern aquifer.

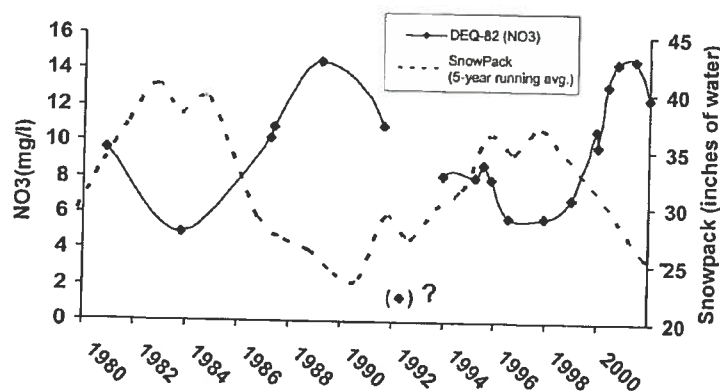


Figure 5-2: Temporal changes in NO₃ concentrations in well DEQ-83, located in the eastern aquifer. A five year running average of snowpack (as inches of water) in the Wild Horse Divide area in the Bannock Range is also displayed. This graph shows an inverse relationship of NO₃ concentrations in this well with levels of precipitation. Note that the 1992 data point may be suspect. It is the only value less than 5 mg/l; if it were a decimal-point error (10 instead of 1.0), its value would be very similar to 1991 and 1993 measurements.

5.2 Spatial Relationships

5.2.1 Summary of Spatial Distribution of Water Types

As described in sections 3.2, 3.3, 4.1, and 4.2, mixing trends in the southern aquifer provide evidence for three types of groundwater composition found in the West Bench tributary drainages of the Bannock Range and eastern aquifer. These have been described as high-TDS/high-NO₃ water, high-TDS/low-NO₃ water, and low-TDS/low-NO₃ water. Figure 5-3 summarizes the spatial distribution of these water types for samples collected in 1993-1994 and 2004, as well as septic system locations. Note that some sample locations are from the IDEQ database (Appendix A).

Figure 5-3 shows a clustering of similar water types at a few locations, but over all there is a fairly high degree of spatial variability with sharp changes in water chemistry over small distances. In the Mink Creek drainage downgradient of DEQ-04, this variability may be due to the heterogeneous nature of groundwater transport due to fracture-dominated flow, as discussed in section 4.3.1.

All of the wells with low-TDS/low-NO₃ type water in the main aquifer are completed in the Tertiary sediments that comprise the base of the municipal aquifer. All of the artesian wells in the West Bench areas of the Bannock Range have low-TDS/low-NO₃ type water. Well logs indicate these artesian wells are completed in the Late Proterozoic Caddy Canyon Quartzite of the Brigham Group. Groundwater associated with pure quartz formations can be among the lowest in dissolved solids (Hem, 1980), which may explain the low TDS chemistry observed in these artesian wells. Low TDS waters, in general, may reflect little rock-water interactions in the Bannock Range, consistent with the inference that this is the principal recharge source area and that flow

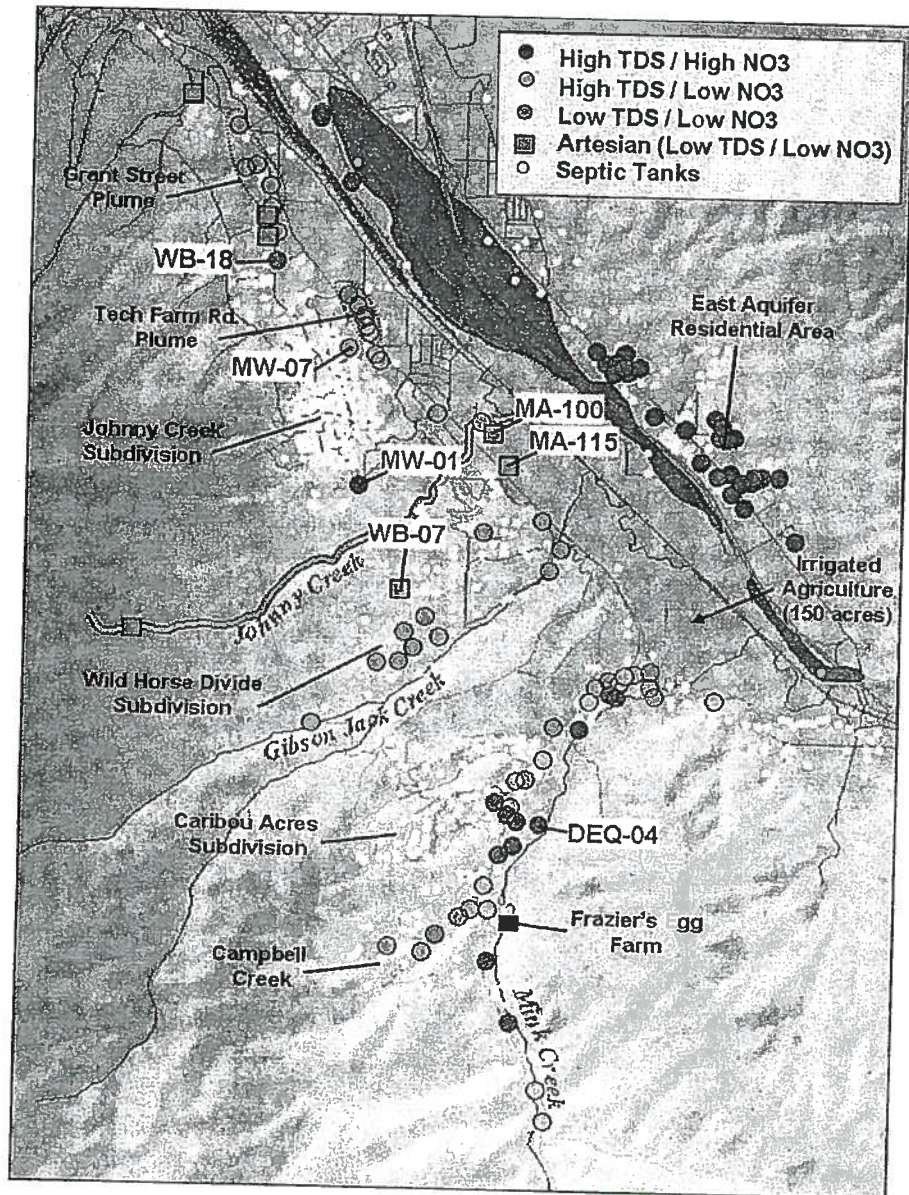


Figure 5-3: Spatial distribution of high-TDS/high-NO₃ water, high-TDS/low-NO₃ water, and low-TDS/low-NO₃ water in samples collected in 1993-1994 and 2004. Several sample Location IDs referred to in the text are identified. The area of a 150 acre farm, which grows alfalfa and potatoes seasonally, is highlighted. Johnny Creek is the only surface water in the West bench elevated in salts.

paths and travel times are short.

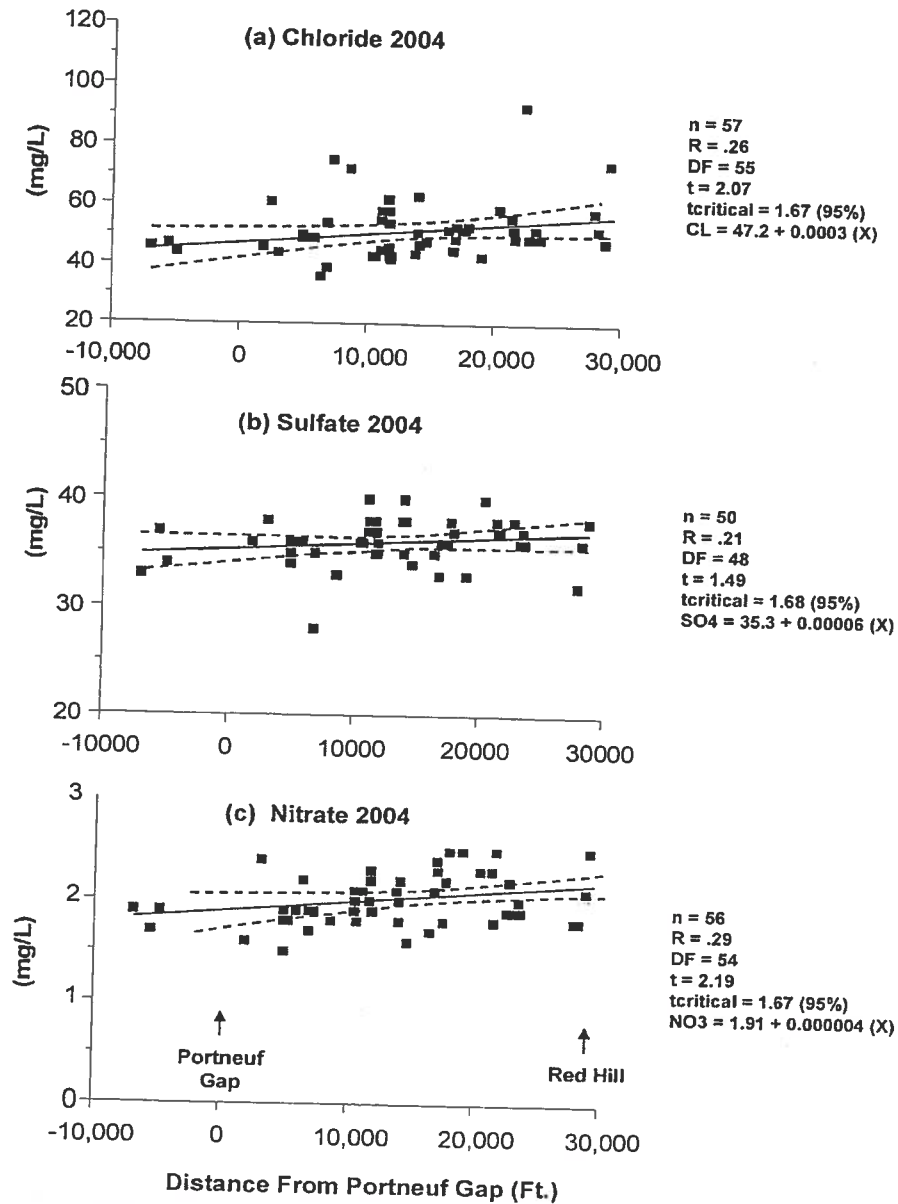
Several areas are characterized by high-TDS/low-NO₃ waters. In the main aquifer there are two localized “plumes” near Grant Street and Tech Farm Rd (see sections 3.3 and 4.2). These types of waters (high in salts and low in NO₃) can also be found in the Campbell Creek area and the Wild Horse Divide subdivision in the Bannock Range. Surface water samples collected from the head and mouth of Johnny Creek are also high-TDS/low-NO₃. Surface water in all other West Bench tributaries of the Bannock Range are low-TDS/low-NO₃.

Groundwater contaminated with NO₃ and other salts (high-TDS/high-NO₃ waters) are found in the eastern aquifer, where septic tank effluent was responsible for contamination (Winter et al., 1994) and in Mink Creek upgradient of the poultry farm and directly below the Caribou Acres subdivision. The spatial relation between septic tanks and areas impacted by NO₃ in the Mink Creek drainage suggests that septic effluent may also be responsible for groundwater contamination in that area.

5.2.2 Spatial Changes Along the Flow Path of the Southern Aquifer

Figures 5-4(a) through (c) summarize the concentrations of Cl, SO₄, and NO₃ in the southern aquifer in 2004. Values are plotted as a function of each sample location's spatial distance along the groundwater flow line from the Portneuf Gap. Data locations that reflect localized contaminant impacts (i.e. eastern aquifer and Tech Farm Rd. samples) and wells that are completed in the Tertiary sediments (low TDS) are not shown (see Section 3.3 and 4.2).

A regression analysis was performed on each constituent along the groundwater



Figures 5-4(a) through (c): Concentrations of Cl , SO_4 , and NO_3 in the southern aquifer in 2004. Values are plotted as a function of each sample location's spatial distance from the Portneuf Gap to identify changes in chemistry along the groundwater flow path. A t-test was applied to each to evaluate the statistical significance of each linear regression coefficient at the 95% confidence interval. Only the chloride and nitrate data have t-statistic values greater than t-critical values. Dashed lines represent the 95% confidence bands around the regression slopes.

flow path from the Portneuf Gap to Red Hill. A Student's t-test was applied to evaluate the statistical significance of each linear regression coefficient. The t-statistic is defined as (Till, 1974):

$$t = |r| \sqrt{\frac{(n-2)}{(1-r^2)}} \text{ with degrees of freedom (DF) } = (n-2)$$

where n is the number of sample data plotted in each regression.

Both the Cl and NO₃ regression slopes are significant at the 95% confidence level whereas the sulfate regression slope is not.

Groundwater samples collected in the summer of 1994 showed similar increasing trends along the aquifer flowpath for Cl and SO₄, whereas NO₃ concentrations remained steady. However, a Student's t-test applied to each regression indicated that none were statistically significant at the 95% confidence level, possibly due to smaller sample populations than the 2004 sampling event.

Chemical mass balance models will be developed in Chapter 6 to determine if these trends in chloride and nitrate observed in 2004 can be attributed to suspected contaminant sources.

5.2.3 Evidence for a Near-Surface Source of Salt Contamination

As described above, artesian wells completed deep in the aquifer and West Bench reflect upwelling recharge waters that are not contaminated with near-surface-derived salts. All these wells have low-TDS/low-NO₃ waters.

Domestic well WB-07 (Figure 3-8 and 4-1) is an important sampling location for discriminating shallow sources of groundwater salt contamination. When sampled in 1994, concentrations of Cl, SO₄, and NO₃ in this well were 248 mg/l, 100 mg/l, and 2.6 mg/l, respectively, indicating high-TDS/low-NO₃ type water. The well at that time was

less than 180 feet deep. In 2001, the well went dry and another well was drilled 30 feet away to a depth of 565 feet. This well is flowing-artesian and exhibits low-TDS/low- NO_3 type water typical of West Bench waters and artesian wells in the area: Cl , SO_4 and NO_3 concentrations in 2004 were 33 mg/l, 20 mg/l, and 0.8 mg/l, respectively.

Evidence for near-surface contamination can also be found in the localized plumes on Grant Street and Tech Farm Road. Domestic well MA-62 on Grant Street had levels of chloride and sulfate more than 20 mg/l lower in 2004 (see section 4.2). In 2001, this well was deepened by sixty feet. If the lower salt levels are attributable to this deepening, then a shallow contamination source is implicated. Two of the wells at Grant Street and four of the wells on Tech Farm Road that had elevated salt levels in 1994 are now dry due to lower groundwater levels. This suggests that these wells were producing water from near the water table in 1994 and, therefore, would have been sensitive to near-surface sources of contamination.

In summary, groundwater chemistry in the LPRV is the result of a combination of factors which include: depth of well completion, proximity to sources near the surface, and the lithology of the water bearing unit(s).

5.3 Sources of Salt Contamination

An overview of potential sources of inorganic salts and nutrients (NO_3) in groundwater is provided in Appendix F. Potential sources include landfills, chemical fertilizers, animal waste, road salt, road aggregate, water softeners, natural sources, and sewer and septic tank effluent. The appendix includes examples of typical chemical concentrations of leachate from these sources and what is known of their chemical evolution during transport in the vadose zone and in groundwater.

The approach used to determine the most likely source(s) of contamination was to first exclude sources that could not account for the spatial distribution and trends observed in water chemistry presented in Chapters 3 and 4.

As shown in Figures 3-4, 3-5, and 4-2, concentrations of Cl in groundwater have strong correlations with all major ions, including hardness (CaCO_3), Ca, Mg, Na, K, Si, SO_4 , and NO_3 . This suggests that a single source is responsible for most of the variability in these constituents in groundwater.

Figures 3-5(b) and 4-2(b) also indicate that a second source high in salts but low in NO_3 may be involved, or that nitrate is being attenuated substantially by some chemical process such as denitrification.

5.3.1 Least Likely Sources of Salt Contamination

LANDFILLS

The Fort Hall Canyon landfill is the only landfill in the study area (see Figure 1-7). Some monitor wells at the landfill have elevated concentrations of salts and NO_3 ; however, groundwater samples collected during this study, as well as by Bannock County, downgradient of the landfill indicate that the southern aquifer is not impacted by Cl, SO_4 , or NO_3 from landfill leachate (Bannock County, 2004). Also, landfill leachate could only affect the southern aquifer and not the West Bench drainages. Furthermore, any salt plume deriving from the landfill would be just as narrow as the TCE plume (see Figure 1-7); so a landfill source cannot account for contamination on the east side of the aquifer.

CHEMICAL FERTILIZERS

Chemical fertilizers can impair groundwater quality with salts and NO_3 ,

especially when over-applied or over-irrigated (Minnesota Pollution Control Agency, 2000). Agricultural activities in the LPRV are restricted to a single 150 acre farm on the valley floor between the Mink Creek and Gibson Jack drainages (Figure 5-3) which grows crops of alfalfa and potatoes seasonally. Seven monitor wells installed to characterize the TCE contamination in the southern aquifer are located directly down gradient and within the area of this irrigated cropland. All monitor wells that are completed close to the water table show background concentrations of salts and nitrates, suggesting no impact from fertilizers.

Currently, there is no major agricultural activity in the West Bench or adjacent areas. The only fertilizer use would be for domestic gardens and turfgrass. In this semi-arid climate, losses of fertilizers past the root zone would tend to be minimal, and would only occur if there were significant over-irrigation. Nitrogen losses tend to be low (< 5% of applied fertilizer) from established turfgrass that is not over-irrigated (Barton and Colmer, 2004).

Well established alfalfa crops do not require nitrogen fertilizers while potato crops typically need more nitrogen and amendments compared to other crops grown in Idaho (Stark et al., 2002; Stark et al., 2004). Potatoes require 200 to 240 pounds of nitrogen (principally as ammonia) per acre for each growing season and up to 600 pounds of potassium chloride (KCl) may be applied (Stark et al., 2004). At these application rates, the ratio of Cl to N applied to crops would be less than 2:1, which is much lower than the Cl to NO₃-N ratio of 27:1 observed in groundwater (Figure 4-2(b)).

Sulfur (S) amendments in potato crops typically range from 18 to 24 pounds per acre, which could yield 54 to 72 pounds of SO₄ after complete oxidation. The ratio of Cl

(from KCl) to SO_4 in fertilized soils would be approximately 4.5:1, which is almost double the 2.5:1 Cl: SO_4 ratio observed in groundwater.

Nitrogen losses to groundwater under irrigated agriculture range from 70 to 150 pounds per acre per year, depending on cropping practices and climate (Minnesota Pollution Control Agency, 2000). Assuming the worst case scenario of 150 pounds per acre of N loss, approximately 10 metric tons of $\text{NO}_3\text{-N}$ could impact groundwater annually from the 150 acre farm in the LPRV. In comparison, this farm would contribute about 20 metric tons of Cl if all of the Cl from the application of 600 pounds per acre of KCl (288 lbs. per acre Cl) were lost to groundwater.

ANIMAL WASTES

Significant volumes of animal manure enriched in nutrients and inorganic salts can be found on dairies, poultry farms, feedlots, stockyards and other facilities used to raise and hold animals. Currently there are no such facilities in operation in the LPRV. Frazier's poultry farm, located in Mink Creek (see Figure 5-3), went out of business in 1998. It is not known whether residual manure at this facility may continue to impact local groundwater (see section 4.3).

Though many residents maintain small corrals and holding pens for horses and other livestock, it would be difficult to mobilize significant amounts of manure leachate through the vadose zone to the water table in a region where evapotranspiration exceeds precipitation.

Two domestic wells (MA-03 and WB-18) may have elevated salts and nitrate that are derived from local sources of animal waste. MA-03 is located within a horse corral and the only well in the main aquifer with nitrate levels greater than 3 mg/l in 2004 (see

Section 4.2). WB-18 is located in the Johnny Creek subdivision on a property that was formerly a poultry farm. In both cases, surface contaminants may be reaching the aquifer via leakage along poorly sealed well casings.

Table F-2 in Appendix F shows average concentrations of Cl, SO₄, and N (as NO₃ and NH₄) found in leachate from cattle manure. The ratio of Cl to NO₃ and total N (NO₃ + NH₄) in manure leachate ranges from 3.3:1 to 6.5:1. These values are much lower than the 27:1 ratio of Cl to NO₃ observed in groundwater. The ratio of Cl to SO₄ in manure leachate is 3.5:1, which is slightly higher than the 2.5:1 ratio observed in groundwater.

An average 1,000 pound horse or cow can produce 50 to 60 pounds of manure a day with a nitrogen content of 0.5% by weight (Davis and Swinker, 2004; Queensland Government, 2004; Ecochem, 2005). The number of livestock in the LPRV is unknown, but for every one hundred animals, approximately 5 metric tons of nitrogen would be available annually from such a source.

ROAD AGGREGATE

Meehan and Welhan (1994) proposed that leachate from local roadways may be a source of contamination, reflecting the combination of road-deicing salts (NaCl) and leachable-sulfate found in road aggregate (see Section 3.2 and Appendices D and F). Leach tests performed on samples from a stockpile of road aggregate revealed that SO₄, Ca, and K were major soluble components, and geochemical modeling indicated that ion-exchange of sodium with Ca and Mg in clays could explain the deficit of sodium concentrations compared to Cl (see Section 3.2), and the observed coupled increases of Mg with other salts (Appendix D).

It is not known where and for what time period such road aggregate was used in

the LPRV. Its use would have had to have been ubiquitous throughout the study area and other parts of the Portneuf River valley to account for the spatial distribution of salts found in groundwater. This is unlikely considering that most road construction projects use local material for road aggregate that would probably not contain significant amounts of leachable sulfate. Furthermore, leachate derived from road-aggregate does not contain NO_3 or Si, both of which are positively correlated with groundwater Cl and SO_4 (see Sections 3.2 and 4.1).

SEWER EFFLUENT

Domestic waste distributed to municipal sewers can have high concentrations of all salts and nitrate (Wilhelm et al., 1994, 1996). Within the study area, only residences inside the city limits of Pocatello and a few small sections of the Johnny Creek subdivision are serviced by municipal sewer. Furthermore, municipal sewage could not account for the salt and nitrate contamination found in the West Bench drainages of the Bannock Range, the eastern aquifer and most of the southern aquifer, where domestic waste is treated by septic technology (see Figure 5-3). In the LPRV municipal sewage is treated at a facility six miles north of the study area and would not impair local groundwater unless a sewer line broke.

In this regard, exceedingly high concentrations of nitrate (that did not correlate with the trends displayed in Figure 4-2) were found in one groundwater sample (MW-01) that could reflect a broken sewer line. Monitor well MW-01 was installed as part of an investigation for a city sewer extension project (Montgomery Watson, 1996). Depth to groundwater is only 36 feet and the total well depth is 45 feet, indicating that a near-surface source was responsible for the contamination.

Four samples were collected from this well. Bailed and pre-bailed nitrate concentrations ranged from 48 mg/l to 111 mg/l. Chloride and sulfate concentrations varied from 115 mg/l to 124 mg/l and 45 mg/l to 54 mg/l, respectively. At these chloride and sulfate concentrations, high-TDS/high-NO₃ type waters observed elsewhere in the valley typically have nitrate concentrations around 4 to 5 mg/l (see Figures 4-2(a) and (b)).

As shown in Figure 5-3, this well is located up gradient of most of the septic tanks in the Johnny Creek subdivision. Personnel from the City of Pocatello Waste Operations Division were on site during the second round of sampling at this well and indicated that almost all residences in this section of Johnny Creek are serviced by municipal sewer. It is possible that the high levels of nitrate in this well reflect raw sewage emanating from a break in a sewer line that passes within tens of feet of this well location.

A sample collected from this well in 1996 only had 4 mg/l of NO₃, with chloride and SO₄ concentrations at 107 mg/l and 37 mg/l, respectively. If a sewer line is responsible, it must have ruptured after 1996.

5.3.2 Most Likely Sources of Salt Contamination

After excluding sources that could not account for the spatial distribution and trends observed in groundwater chemistry, the most likely sources remaining are: natural salts, septic effluent, and water softener and road de-icing salt. Septic tanks, and de-icing salts applied seasonally to local roads, in particular, represent potentially significant sources of chloride and nitrate.

NATURAL SOURCES

The chemistry of groundwater is influenced by the composition of recharge water, aquifer mineralogy, groundwater flow path, and residence time. With longer flow paths and residence times, there is a higher probability that groundwater will come into contact with soluble minerals, thereby increasing its salinity and TDS (Hem, 1985).

Some sample locations in this study had concentrations of Cl, SO₄, and especially NO₃ that are atypical of background groundwater. However, there are lithofacies present in the LPRV that could elevate the levels of these constituents. The natural geochemical processes outlined below are discussed in more detail in Appendix F.

The northern Bannock and Portneuf Ranges consist of chemically diverse lithofacies which include limestone, shale, argillite, volcanic ash, and quartzite (Trimble, 1976; Link et al., 1987; Link and Stanford, 1999; Rodgers and Othberg, 1999, Osier 2004). Many of these were deposited in Late Proterozoic through early Mesozoic marine environments.

High levels of salts in groundwater are typically associated with sedimentary rocks, particularly evaporites (Hem, 1985). Deposition of evaporites could have been contemporaneous with many of the other sedimentary lithofacies found in the LPRV; however, it is not likely that such soluble minerals would persist.

Sedimentary and metasedimentary deposits formed in a sea or enclosed basin, or submerged in such environments after their formation, may become impregnated with soluble salts (Hem, 1985). Redox reactions can also increase salinity of groundwater (e.g. sulfate, produced from the oxidation of pyrite and other sulfide minerals; Hem, 1985).

Two areas in the West Bench drainages of the Bannock Range that have high concentrations of salts in groundwater are the Wild Horse Divide subdivision and the Campbell Creek area in the Mink Creek drainage (see Figure 5-3). Wells in the Wild Horse Divide subdivision are completed in Quaternary-Tertiary pediment gravels and underlying Cambrian Elkhead Limestone (IDWR, 2003). Wells in the Campbell Creek area are completed in black shales, possibly of the Gibson Jack Formation. Well logs from both areas show that groundwater rises by as much as 365 feet above the point where water is first encountered, indicating that groundwater at these locations originates under confined conditions. Since the thick sequences of clays and shales and the confined groundwater conditions offer some degree of protection from near-surface contaminants such as septic effluent, the implication is that at least some of the salts in these deep groundwaters may be naturally derived.

An investigation conducted by CH2MHill(1994(a)) at the Pocatello Creek landfill concluded that some of the elevated salts in groundwater were naturally derived. The Pocatello Creek landfill is located east of Pocatello in the Pocatello Creek drainage (see Figure 1-3). Shale facies in the upper member of the Pocatello Formation were found to contain pyrite (2 to 3 percent) and wells completed deep in the Pocatello Formation tended to have sulfate as the major ion (60 to 85%), suggesting that pyrite oxidation is responsible. Other minerals may be released into solution to buffer the acidity produced in the pyrite oxidation process (CH2MHill, 1994(a)).

It is less likely that elevated nitrate levels are due to a natural source such as evaporitic minerals because of lack of evidence of their presence and the relative scarcity of nitrate rich evaporate minerals in evaporate deposits. Nitrogen in rocks can be present

in large quantities in the form of ammonia (NH_4^+) and organic-N derived from the decay of buried organic material (Hem, 1985), both of which may become oxidized to NO_3 . However, naturally occurring levels of NO_3 above 3 mg/l in groundwater are rare (see appendix F).

The highest levels of salts and nitrate in groundwater tended to be in shallow wells (see Section 5.2.3). Of the ten wells completed in the Wild Horse Divide subdivision, the wells with the highest TDS and NO_3 values are the four completed in the pediment gravels and not the deeper limestone unit. Investigations at the Pocatello Creek landfill and the eastern aquifer also determined that shallow wells had higher concentrations of Cl and NO_3 (CH2MHill, 1994(a), Winter et al., 1994). No natural source of Cl was found during the landfill investigation and the strong coupled correlation of Cl and NO_3 in shallow waters suggests an anthropogenic surface source.

In summary, natural sources may contribute minor amounts of salts to groundwater, but they are unlikely a major source.

ROAD DE-ICING SALTS

De-icing salts are applied seasonally to local roads throughout the entire LPRV. Stockpiles maintained by Bannock County, the City of Pocatello, and the Idaho Transportation Department, typically contain about five percent by weight of NaCl (City of Pocatello, 1993; Bannock County, 1993; ITD, 1993).

The amount of road salt applied to roadways in the southern LPRV by the City of Pocatello and Bannock County is unknown. However, the Idaho Transportation Department applies approximately 50,000 cubic yards of gravel over 1,800 lane miles in the State of Idaho each winter season (E. Bala, personal communication 2005). With a

salt content of 5% at 25% porosity, approximately 1 cubic yard of salt per lane mile is applied each year. There are approximately 25 lane miles on Interstate I-15 between the Portneuf Gap and Red Hill in the LPRV, so that 43 metric tons of salt are applied to this section of roadway seasonally. At this application rate, 26 metric tons of chloride would be available for leaching to groundwater from Interstate I-15 alone.

SEPTIC EFFLUENT

Septic tanks are commonly associated with impacts of salts and nitrates in shallow groundwater (Wilhelm et al., 1994, 1996). Septic effluent is typically high in salt content, which may even increase as leachates migrate beyond the drain field. Aerobic oxidation of ammonia (NH_4) to NO_3 and of organic S to SO_4 below the vadose zone also increases the acidity of effluent, releasing additional solutes through buffering reactions such as mineral dissolution or cation exchange (Wilhelm et al., 1994, 1996).

Table F-3 in Appendix F shows ionic concentrations found in septic tank effluent from five studies. The table shows that for some constituents, particularly Cl and SO_4 , concentrations in effluent are lower than those found in some groundwater samples in the LPRV. However, levels of Cl and SO_4 in effluent can vary widely due to additions from water softeners, detergents, solvents, and other household products.

Some studies have shown that sulfate concentrations can increase as effluent travels through the vadose zone due to oxidation of residual reduced S (Wilhelm et al., 1994, 1996). The highest observed groundwater sulfate concentration was 262 mg/l; typical concentrations in septic effluent are much lower (12 to 60 mg/l).

If pyrite is present in the vadose zone below a septic leach field, the intrinsic acidity of septic effluent would tend to increase the rate of pyrite oxidation, thereby

releasing additional SO_4 . Initially, pyrite (FeS_2) is oxidized to ferrous iron (Fe^{2+}), sulfate (SO_4^{2-}), and acidity (H^+). Bacterially-mediated conversion of ferrous to ferric iron (Fe^{3+} , which remains soluble in acidic solutions), acts to enhance further pyrite oxidation. Thus pyrite oxidation coupled with ferric iron production can become the principal mechanism for generating acidity and releasing additional SO_4 (Younos et al., 1999; Brady et al., 2000).

Concentrations of NO_3 in septic effluent typically range from 30 to 60 mg/l after complete oxidation of ammonia, (Bauman and Schafer, 1985; Hantzsche and Finnemore, 1992; Wilhelm et al., 1994, 1996; ITRC, 2000). Table 5-1 shows the concentrations of NO_3 in septic effluent as determined in five studies. This table was adapted from Table F-3 in Appendix F, assuming 20% of ammonium in leachate is adsorbed and 80% oxidized to nitrate (Wilhelm et al., 1994, 1996).

Table 5-1: Concentrations of NO_3 in septic effluent as determined in five studies. This table was adapted from values in Table F-3 in appendix F, assuming 20% adsorption of ammonium ions below the septic leach field (Wilhelm et al., 1994, 1996).

Parameter	Robertson et al., 1991	Harman et al., 1996	Wilhelm et al., 1996	Lowe et al., 2004	McQuillan, 2004
$\text{NO}_3\text{-N}$ (mg/l)	24	102	46	5 - 128	0 - 28

The highest nitrate concentrations found in LPRV groundwater are less than 30 mg/l, indicating that if septic effluent is a source of nitrate, it is being diluted by ambient groundwater.

As discussed in Appendix F, high-TDS/low- NO_3 type groundwaters may be the result of denitrification. However, denitrification requires anaerobic conditions as well as an electron donor such as labile organic C (Wilhelm et al., 1994, 1996), the latter being

the most common limitation to denitrification in groundwater settings. Most organic C in waste water is consumed by aerobic oxidation at the same time that NO_3 is created (Bauman et al., 1985; Wilhelm et al., 1994, 1996).

Another possible mechanism for nitrate attenuation is via oxidation of reduced sulfur by NO_3 (Molenat et al., 2002; Robertson and Blowes, 1995). Shales found in the Bannock Range may provide sources of both reduced carbon and reduced sulfur. This may explain why many of the high-TDS/low- NO_3 type waters in the Bannock Range are associated with these types of lithofacies.

Water softeners can significantly increase the Cl concentration of septic effluent, in some cases, to more than 5,000 mg/l (Panno et al., 2002). The highest chloride concentration in LPRV groundwater exceeds 800 mg/l.

The amount of salt (NaCl) used for regeneration of water softener ion exchange resin ranges from as little as 200 pounds per year to over 4,000 pounds per year per unit depending on the level of hardness, water usage, and the type of water softener (Kuntz and McBride, 2004). A distributor of water softeners in Pocatello stated that the newest water softeners use as little as 250 pounds a year, while older models can use more than 1500 pounds per year.

A resident in the Johnny Creek subdivision stated that for a family of four, they use approximately 480 pounds of salt a year. If this were considered as the average use in this area, each home would generate 293 pounds of Cl a year, with more than 80 metric tons of Cl annually discharged from the more than 600 homes using septic tanks in the LPRV.

At a rate of 293 pounds of Cl discharged a year, and average water usage ranging

from 150 to 300 gallons per day per household (Bauman and Schafer, 1985; BBC Research and Consulting, 2001; Hantzsche and Finnemore, 1992; ITRC, 2000; Panno, 2002; Thomas et al., 1999; Wilhelm et al., 1994), the concentrations of Cl in septic effluent would range from 320 mg/l to 640 mg/l. Water softeners that use as much as 1500 pounds of salt per year would produce Cl concentrations from 1000 mg/l to 2000 mg/l.

Based on the above Cl concentration estimates and NO₃ concentrations of 30 to 60 mg/l, the ratio of Cl to NO₃ in septic effluent could range from 5:1 to greater than 60:1, depending on the amount of salt derived from water softeners. The 27:1 Cl to NO₃ ratio observed in groundwater suggests that, if Cl is predominantly derived from water softening, then homes on septic systems annually use approximately 1300 pounds of salt.

A comparison of Cl, SO₄, and NO₃ concentrations in outfall from the Pocatello Waste Water Treatment facility and concentrations in LPRV groundwater is discussed in Section F.8 of Appendix F. Figure F-1 shows that the ratios of Cl, SO₄, and NO₃ in treated sewage (outfall) are similar to groundwater that is suspected to be impacted by septic effluent, and strongly suggests that the elevated concentrations of these constituents found in groundwater reflect the influence of sewage effluent from septic systems.

Table 5-2 summarizes the calculated potential fluxes of and ratios of chloride and nitrate in some of the sources discussed in this section. This table shows that the ratio of Cl/NO₃ found in septic effluent falls within the range of 27:1 Cl/NO₃ found in southern LPRV groundwater, while the ratio observed in other sources is much lower.

Table 5-2: Summary of potential fluxes to groundwater and ratios of chloride and nitrate from sources in the southern LPRV. This table shows that the ratio of Cl/NO₃ found in septic effluent falls within the range of 27:1 Cl/NO₃ found in southern LPRV groundwater.

	Cl (metric tons yr.)	NO ₃ (metric tons yr.)	Cl : NO ₃
Septic Effluent + Water Softeners	> 80	6 - 11	5:1 to 60:1
Chemical Fertilizers (one 150 acre farm)	20	10	2:1
Animal Manure (per 100 cows/horses)	25	5	5:1
Road Salt	> 26		NA*

NA* = not applicable because of no nitrate component in road salt

5.4 Summary of Source Evaluation

Landfills, road aggregate, chemical fertilizers, animal waste, and sewer effluent are considered the least likely sources of salts and nitrate impacting LPRV groundwater. There is only one landfill, and it could not account for the groundwater contamination observed in the West Bench drainages or for contamination in the eastern aquifer. Road aggregate containing leachable sulfate is a possible but unlikely source, and by itself could not account for the positive correlation of NO₃ with SO₄ in groundwater. Chemical fertilizers and animal manure are potentially important sources of NO₃, but, the ratio of Cl to NO₃ typically associated with these sources is more than an order of magnitude lower than the ratio observed in LPRV groundwater (Table 5-2). In addition, agricultural activities involving fertilizer use are limited, and animal waste is restricted to small corrals and holding pens for horses and other livestock. Municipal sewer lines within the city limits, could not account for salt and nitrate impacts observed in rural areas.

After eliminating sources that cannot account for the spatial distributions and trends observed in groundwater chemistry, the most likely remaining sources are natural sources, road de-icing salts, and septic effluent.

Natural sources may contribute to groundwater salinity, but the strong coupled correlation of Cl and NO₃ in shallow groundwater indicates an anthropogenic, near-surface source. The amount of road salt applied seasonally in the LPRV could be a potential source of sodium and chloride, but cannot explain the coupled nitrate contamination.

Septic effluent typically contains high concentrations of NO₃ (30-60 mg/l), and can also have high concentrations of Cl derived from water softening, as well as elevated SO₄ concentrations.

There are approximately 672 residences in the southern LPRV with septic tanks (Figure 5-3). Assuming an average water usage of 225 gallons per day per residence and NO₃-N concentrations in septic effluent ranging from 30 mg/l to 60 mg/l, approximately 6 to 11 metric tons of NO₃-N is discharged via septic tanks each year in the LPRV. The spatial relationship between septic tanks and groundwater high in TDS and NO₃ (Figure 5-3) suggests that septic effluent may be the principal factor in impairing groundwater quality, a hypothesis that will be tested in Chapter 6.

Chapter 6: Modeling

6.1 Simple Chemical Mass Balance Modeling

The chemical mass balance modeling approach uses chemical tracers in the components of aquifer recharge to determine the relative contribution of each component to the southern aquifer. Mass balance computations require the solution of a set of simultaneous equations. Sections 2.5 and 2.6 summarize the assumptions and methodology involved in chemical mass balance modeling.

Two types of software were used for modeling: Microsoft Excel ® and IsoSource (U.S. EPA, 2003). Microsoft Excel ® can be used to solve for a system of simultaneous equations using matrix inversion. Matrices were constructed using tracer concentrations for geochemically distinct components of recharge from three sources: the Bannock Range, east aquifer, and inflow through the Portneuf Gap. Welhan et al. (1996) surmised that the Portneuf River was not a significant recharge source to the southern aquifer; therefore, the river was not considered.

IsoSource (U.S. EPA, 2003), a software package developed by U.S. EPA calculates the ranges of source proportional contributions to a mixture based on chemical or stable isotope tracer chemistry when the number of recharge sources exceeds the number of tracers available to permit unique solutions. Mass balance solutions calculated by IsoSource were compared to the matrix models constructed with the Excel spreadsheets.

Table 6-1 summarizes the concentrations of tracers defining end-members in one matrix model and the resulting calculated contributions to the southern aquifer from each end-member recharge source. The chemistry in domestic well WB-102 was used to

	End Members				
	High TDS/ Low NO₃	Low-TDS/ Low NO₃	High TDS/ High NO₃	Portneuf Gap	Main Aquifer
Cl (mg/L)	897	35	1200	47.3	56.6
SO ₄ (mg/L)	262	20	450	35.7	37
NO ₃ (mg/L)	4.3	0.8	45	1.8	2.18
Contribution (%)	-0.5	21.7	1.4	77.4	

Table 6-1: Concentrations of tracers used to define end-members in one matrix model and the resulting end-member contributions to the southern aquifer.

define the high-Cl/low-NO₃ type water, which had the highest Cl and SO₄ concentrations for these water types. Low-TDS/low-NO₃ water was defined by the chemistry of one artesian well (MA-115) in the southern aquifer that was assumed to represent low-TDS recharge water from the Bannock Range. The average concentrations in the three wells south of the Portneuf Gap were used to characterize inflow through the Gap.

The high-TDS/high-NO₃ type end-member is hypothesized to result from septic effluent (see section 5.3.2), whose NO₃-N concentrations typically range from 30 to 60 mg/l; 45 mg/l was used as an average concentration. Chloride and sulfate concentrations in septic effluent were derived by extrapolating their expected concentrations from the trends observed in Figures 4-2(a) and (b) for waters contaminated with nitrate.

The southern aquifer's ionic chemistry was derived from the regression analyses that showed increasing trends of each constituent along the aquifer flow path (see Figures 5-4a, b, and c). Concentrations predicted by the regression equations for the Red Hill area were used to represent the outflow from the model, with the assumption of complete mixing of all end-member sources within the aquifer.

The results of this simple model indicate that the relative contributions of Portneuf Gap inflow and recharge from the Bannock Range were 77.4% and 21.7%, respectively, whereas septic effluent contributes only 1.4%. In this model, high-TDS/low-NO₃ type waters make a negligible contribution.

Several models were evaluated with different concentrations of tracers in the recharge components. The results were similar, with Bannock Range recharge contributions ranging from about 5% to 22%, and inflow from the Portneuf Gap ranging from 77% to 95%. Similar results were also obtained using IsoSource software.

Welhan et al. (1996) developed a water balance model for the southern Pocatello aquifer based on physical hydrogeologic constraints (see section 1.4.2) in which inflow from the Portneuf Gap accounted for only about 10% to 20% of recharge and Bannock Range recharge contributing more than 80%. The chemical mass balance results contradict this substantially, suggesting that certain assumptions in the chemical mass balance modeling may be incorrect. First, the chemical tracers must be conservative (no chemical reactions). However, as discussed in Section 5.3.2, denitrification and sulfate production may be occurring at some locations. Secondly, there should be complete mixing of all chemical components in the aquifer. Due to the high linear velocities in the aquifer, however, longitudinal dispersion is significantly greater than lateral dispersion and would inhibit lateral mixing of recharge components across the width of the aquifer (Welhan and Meehan, 1994; Welhan et al., 1996). Also, most sample locations in the main aquifer are near the western margin and from wells at shallow depths that may not sufficiently characterize its chemistry.

Third, and most importantly, all end-members components must be characterized

by substantially different concentrations of at least one tracer. In the example shown in Table 6-1, the chemistry of inflow through the Portneuf Gap is similar to the main aquifer, which introduces an automatic bias in the chemical mass balance.

This mass balance model is also inherently weak due to the fact that it is an under-determined model. Ideally there should be at least as many chemical tracers as there are components of recharge to solve for unique solutions. In this case there are more recharge components than tracers.

Due to these limitations, the water balance model of Welhan et al. (1996) is considered to be more reliable. An updated version of the water balance for a period from June 1994 through October 2003 (Appendix C) and spanning periods of below- and above-normal precipitation indicates that annual recharge from the Bannock Range is of the order of 5 to 6.5 Bgy. This range was used to constrain groundwater recharge in the mixing models described in the following sections.

6.2 Segmented Mixing Models

In order to overcome the limitations of the above chemical mass balance modeling approach, segmented mixing models were developed along the direction of flow in the southern aquifer (see Figures 5-4(a) and (c)). The southern aquifer was split into five mixing "cells", M1 through M5 (Figure 6-1), their boundaries drawn to separate areas potentially contributing major influxes from adjacent tributary drainages (i.e. Mink Creek, Gibson Jack Creek) as well as areas of high septic densities in the West Bench area (i.e. Johnny Creek subdivision). The water balance results of Welhan et al. (1996) (see section 1.5.3) and Appendix C were used to constrain the flux of groundwater recharge through the mixing cells.

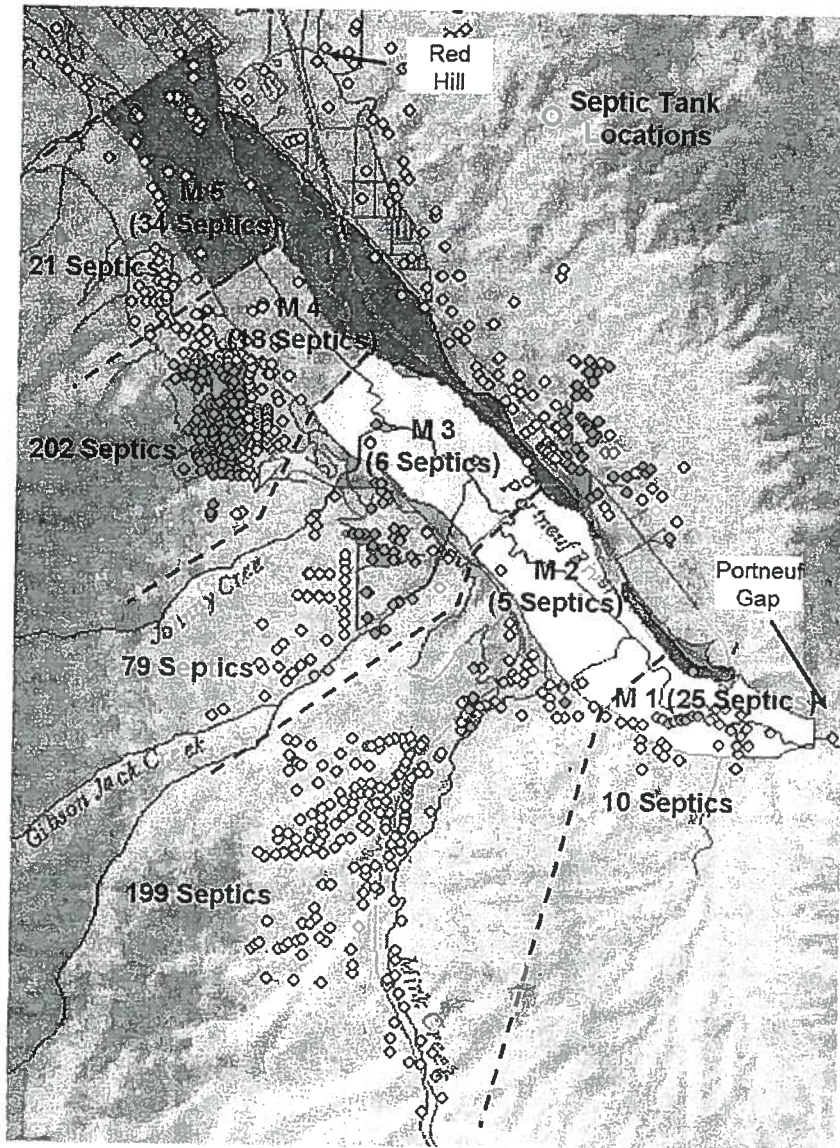


Figure 6-1: Southern aquifer cell boundaries for septic mixing models. The boundaries partitioning each cell were drawn to separate areas potentially impacted by major influxes from adjacent tributary drainages (i.e. Mink Creek, Gibson Jack Creek) and high septic densities in the west bench area (i.e. Johnny Creek subdivision). The flux of septic effluent (Q_{septic}) to each cell was estimated as 225 gallons per day multiplied by the number of septic tanks that overlie each cell in the main aquifer and in the adjacent West Bench drainage areas between the dashed lines.

Progressively more complex models were constructed in three stages. Conceptual diagrams for the three types of models are illustrated in Figures 6-2, 6-5, and 6-7 for $C_X^{NO_3-N}$.

Estimates for septic effluent fluxes and concentrations of NO_3-N in effluent came from literature reviews (Bauman and Schafer, 1985; BBC Research and Consulting, 2001; Hantzsche and Finnemore, 1992; ITRC, 2000; Panno, 2002; Thomas et al., 1999; Wilhelm et al., 1994). An average household generates between 150 – 300 gallons per day of waste water. The average flux of septic effluent (Q_{septic}) to each model cell was assumed to be 225 gallons per day multiplied by the number of septic tanks that contribute to each cell in the main aquifer or the adjacent West Bench drainage area (shown by the dashed lines in Figure 6-1). For example, cell M2 is potentially impacted by the septic tanks that are located over it (5 septic tanks, Figure 6-1) plus all of the septic tanks in the Mink Creek drainage upgradient of M2 (199 septic tanks, Figure 6-1). The total septic effluent recharge flux to the southern aquifer is approximately 49 million gallons per year.

All mixing models were run with septic NO_3-N concentrations ($C_X^{NO_3-N}$) of 30, 45, and 60 mg/l. Some studies have found NO_3-N concentrations in septic effluent as high as 100 mg/l (Hantzsche and Finnemore, 1992), so models were also run with $C_{septic}^{NO_3-N}$ values of 75 and 100 mg/l.

Concentrations of Cl in septic effluent were constrained by their linear relationship to NO_3-N in groundwater (Figure 4-2(b)). The ratio of Cl to NO_3 in high-Cl/high- NO_3 type groundwater assumed to be contaminated by septic effluent is approximately 27 to 1. At this ratio, C_{septic}^{Cl} concentrations ranged from 800 to 2700 mg/l, corresponding to the $C_{septic}^{NO_3-N}$ concentrations of 30 to 100 mg/l.

6.2.1 Model A: Incremental Increase in Aquifer Flux

The simplest model scenario (Figure 6-2) assessed the mixing of only septic effluent from drain fields located over the main aquifer and the adjacent contributing West Bench area. The water balance of Welhan et al. (1996) (see section 1.5.3) required that 1.1 billion gallons a year (Bgy) enter through the Portneuf Gap, with approximately 6.6 Bgy of outflow at Red Hill. In this model, 1.1 Bgy enters cell M1 through the Portneuf Gap with a flux increase of 1.1 Bgy at each cell, so that a total of 6.6 Bgy eventually leaves as outflow from cell M5. The tracer concentrations of water flowing through the Portneuf Gap ($C_{gap}^{NO_3-N}$ and C_{gap}^{Cl}) was determined from the linear regression analyses in Figures 5-4(a) and (c).

The first model was run with septic effluent concentrations of NO_3 at 30 mg/l and 60 mg/l, and corresponding Cl concentrations of 800 mg/l and 1600 mg/l (from the 27:1 ratio observed in groundwater). Figures 6-3(a) and (b) summarize the results of Model A, which show that in this mixing scenario, Cl and NO_3 concentrations in the main

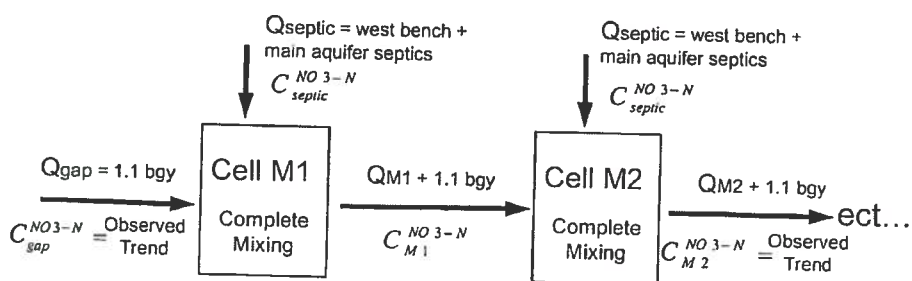
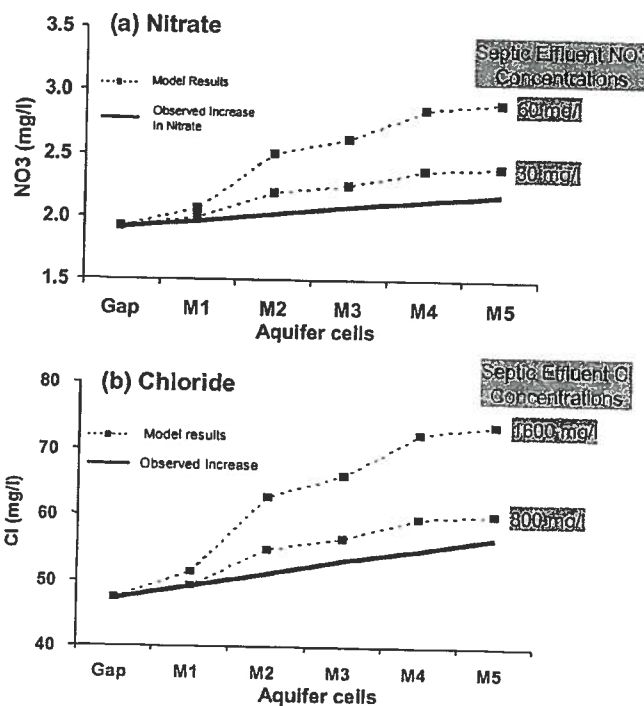
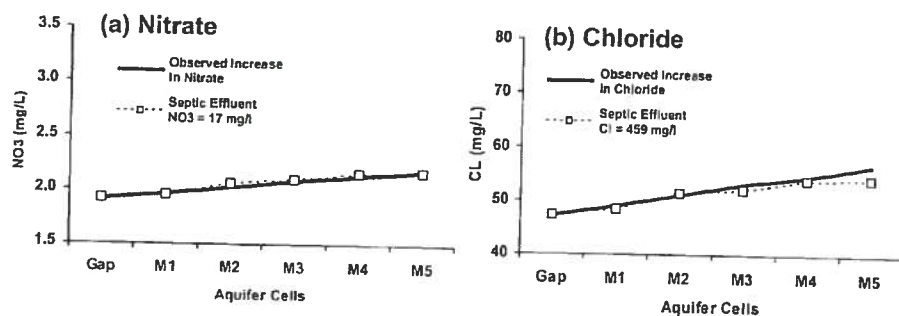


Figure 6-2: Septic mixing model A flow diagram. In this model, 1.1 bgy enters cell M1 through the Portneuf Gap with a flux increase of 1.1 bgy at each cell, so that a total of 6.6 bgy eventually leaves as outflow from cell M5. The tracer concentrations of water flowing through the Portneuf Gap ($C_{gap}^{NO_3-N}$ and C_{gap}^{Cl}) was determined from the linear regression analyses in Figure 5-4.



Figures 6-3(a) and (b): Results of Septic Mixing Model A representing a total septic recharge flux of 49 million gallons per year. The model results show that in this mixing scenario, Cl and NO_3 concentrations in the main aquifer would be greater than the observed trend.



Figures 6-4(a) and (b): Matching the spatial trend predicted by the model requires septic effluent at 17 mg/l NO_3 and a Cl concentration of 459 mg/l, as constrained by its linear relationship to NO_3 -N in LPRV groundwater. Figure 6-4(b) shows the model matches very closely with the observed 27:1 Cl/ NO_3 ratio in LPRV groundwater.

aquifer would be greater than the observed trend.

The model was recalculated a number of times by decreasing the concentration of NO_3 in septic effluent incrementally until the trend predicted by the model matched the observed trend. Figure 6-4(a) shows that with septic effluent at 17 mg/l NO_3 , the modeled trend and observed trend match very closely. Concentrations of Cl in septic effluent at this NO_3 concentration was estimated to be 459 mg/l, as constrained by the 27:1 ratio observed in groundwater. Figure 6-4(b) shows that the model matches the observed Cl trend very closely at this Cl/ NO_3 ratio.

6.2.2 Model B: West Bench Groundwater Contributions

The model results of Section 6.2.1 indicate that NO_3 and Cl in septic effluent could account for the observed trends of increasing concentrations of these constituents in the southern aquifer. However, the concentration of 17 mg/l $\text{NO}_3\text{-N}$ needed to make the model fit the observed trend is lower than the 30 to 60 mg/l range that is typical of septic effluent, suggesting that septic effluent is being diluted by a natural recharge component.

Model A does not explicitly account for any recharge flux other than septic effluent, and also arbitrarily increases the aquifer flux between each cell without identifying another recharge source besides septic effluent. To address these shortcomings, a second modeling approach incorporated an influx of West Bench recharge in addition to septic effluent (Figure 6-5).

In model B, the flux of water and tracer into cell M1 from contributing septic tanks and the Portneuf Gap was the same as in model A. A West Bench tracer concentration ($C_{wb}^{\text{NO}_3\text{-N}}$) was assigned and used to solve for the flux of West Bench recharge (Q_{wb}) to cell M1 that was necessary to fit the tracer concentration in M1

(C_{M1}^{NO3-N}) to the regressed NO_3 and Cl trend concentrations at that point (Figure 5-4).

The flux of water entering cell M2 therefore, was the sum of the fluxes added to cell M1 ($Q_{septic} + Q_{wb} + Q_{gap}$). This process continued for each cell, keeping C_{wb}^{NO3-N} constant.

Each model could then predict, for a specified C_{wb}^{NO3-N} , the total flux of West Bench water that was necessary to fit the observed trends in aquifer chemistry. These models were used to constrain the tracer concentrations that would be necessary in West Bench water to match the 5.5 bgy of West Bench flux predicted by Welhan et al. (1996).

Figures 6-6(a) and (b) summarize the relationships between computed West Bench flux, West Bench background recharge concentration, and the average concentration of septic effluent. The dashed lines identify the 5.5 ± 1 bgy of West Bench recharge flux as constrained by the water balance of Welhan et al. (1996). This range

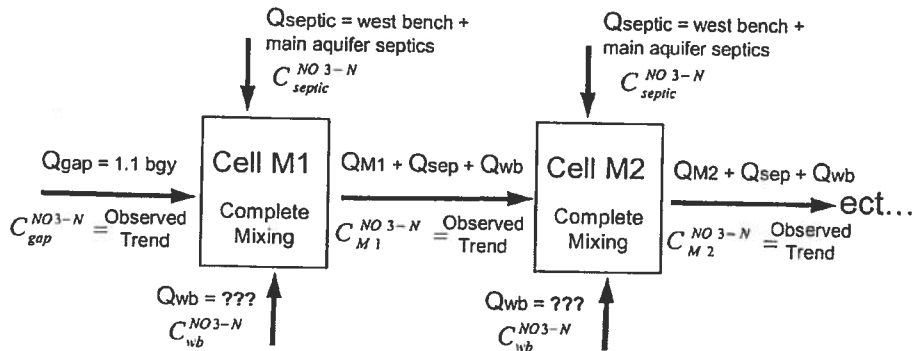
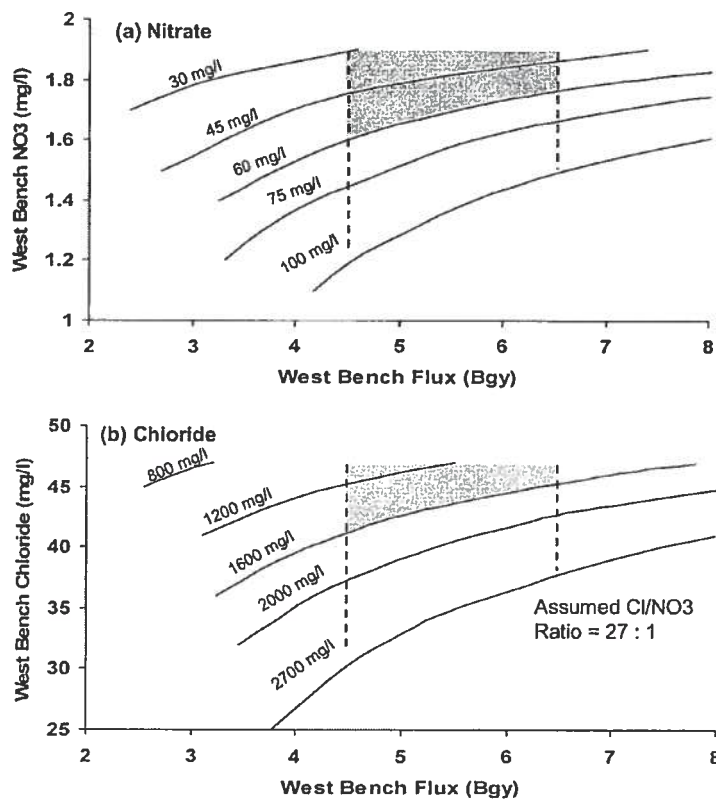


Figure 6-5: Septic mixing model B flow diagram. This modeling approach incorporated an influx of West Bench background water in addition to septic effluent to each cell. A West Bench tracer concentration (C_{wb}^{NO3-N}) was assigned and used to solve for the flux of west bench water (Q_{wb}) necessary to mix in cell M1 so that the tracer concentration in M1 (C_{M1}^{NO3-N}) fit the observed NO_3 and Cl trend in Figure 5-4. The flux of water entering cell M2 was the combined sum of the fluxes added to cell M1 ($Q_{septic} + Q_{wb} + Q_{gap}$).

was chosen to represent possible uncertainty in the water balance calculations as well as changes in West Bench fluxes between wet and dry periods that were determined in the updated water balance in Appendix C.

The shaded areas bracket the most likely ranges of West Bench fluxes and groundwater recharge concentrations corresponding to septic effluent NO_3 levels from of



Figures 6-6(a) and (b): Results of mixing calculations incorporating West Bench recharge. Solid lines reflect the relationship between West Bench flux, background recharge concentration and average concentration in septic effluent. The dashed lines identify the 5.5 ± 1 bgy of West Bench groundwater flux as constrained by the water balance in Welhan et al. (1996). The shaded areas represent the most likely ranges of West Bench background concentrations and fluxes for septic effluent NO_3 -N concentrations of 30 to 60 mg/l.

30 to 60 mg/l, the most typical range documented in septic effluent. Some studies have found NO₃ concentrations in septic effluent as high as 100 mg/l (Hantzsche and Finnemore, 1992), which represent possible, but less likely, mixing scenarios.

These model results restrict the range of possible West Bench background NO₃ concentrations to less than 1.9 mg/l and Cl concentrations less than 47 mg/l. These are the concentrations observed at the Portneuf Gap as determined by the linear regression model in Figures 5-4(a) and (c) and that are required to ensure that West Bench groundwater is able to dilute the contribution from septic effluent.

The model indicates that background NO₃ concentrations in West Bench waters most likely range from 1.7 to 1.9 mg/l at a 5.5 bgy flux. If septic effluent were as high as 100 mg/l NO₃, West Bench recharge would have to be as low as 1.2 – 1.4 mg/l NO₃-N. Corresponding Cl concentrations in West Bench recharge (Figure 6-6(b)) could range from 44 to 47 mg/l for typical effluent and as low as 33 mg/l if septic effluent Cl concentrations were substantially higher.

For the range of West Bench fluxes considered most plausible (5.5 ± 1 Bgy, Appendix C), concentrations of NO₃ in West Bench recharge are constrained to better than 10% for the most likely concentrations expected in septic effluent.

To test the model's sensitivity to the magnitude of septic discharge flux, Model B was rerun assuming fluxes of 150 gpd and 300 gpd per septic tank at a NO₃ effluent concentration of 45 mg/l. At 150 gpd, the relationship between West Bench flux and West Bench NO₃ concentrations would be very similar to the 30 mg/l effluent model shown in Figure 6-6(a). At 300 gpd, results were very similar to the 60 mg/l effluent

model in Figure 6-6(a). These results indicate that uncertainty in the assumed average septic discharge flux does not change the model predictions substantially.

6.2.3 Model C: Portneuf River Contributions

The third and most complex mixing scenario considered the impact of recharge from the Portneuf River in addition to West Bench water (see Figure 6-7). Welhan et al. (1996) assumed that the seepage contribution from the Portneuf River was a minor component of the southern aquifer's water balance (see section 1.4.2). However, same-day discharge measurements along various reaches of the Portneuf River and its tributaries in the LPRV indicate that there may be some loss of river water to the aquifer.

Stream flow data from the reach of the Portneuf River that extends from the Portneuf Gap to the Carson Street-USGS gauging station were evaluated to estimate river losses to the southern aquifer (Figure 6-8). Table 6-2 shows stream discharge measurements in this reach of the river made by the ISU Biology Dept. (Inouye, R.,

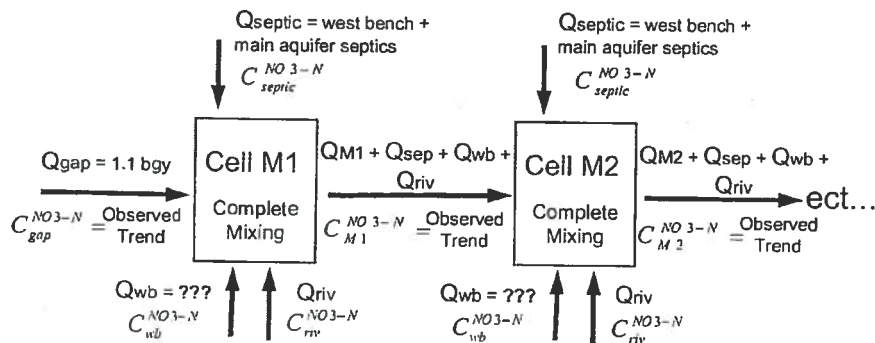


Figure 6-7: Septic mixing model C flow diagram. This modeling scenario considered the impact of recharge from the Portneuf River in addition to West Bench water. An estimate of 1 bgy was used to constrain the river recharge. Concentrations of NO_3 and Cl in the Portneuf River were determined from U.S. Geological Survey analyses made at the in Pocatello gauging station from 1966 through 2002 (Figures 6-9(a) and (b)).

Table 6-2: Portneuf River and tributary discharge measurements in the Pocatello Valley. Measurements made in this study ^a, by Chen (2001) ^b, and by the ISU Biology Dept. ^c. Values designated with an (*) indicate tributary discharges that were estimated from previous tributary measurements ratio to Portneuf River discharge.

Station	Distance (km)	Stream Discharge Measurements (cubic feet per second)				
		July 1994 ^a	August 2000 ^b	August 2002 ^b	Sept 2002 ^c	Sept 2004 ^c
Portneuf Gap	0	17.0				
Fort Hall Rd.	0.7		36.6	38.2	54.3	44.3
Mink Creek		0.7	5.2	2.7	2.3	2
Century Well	2.25			32.6	64.6	
Gibson Jack		0.4	2.5	1.5	1.5	1
Edson Fichter	6.68			45.8	52.0	38.6
Cheyenne Rd.	7.07	13.4	44.0			
Rainey Park	11.35			23.2	39.1	31.9
City Creek			0.6	0.2	0.2	0.2
Carson USGS	14.26		36.3	44.2	48.6	37.0

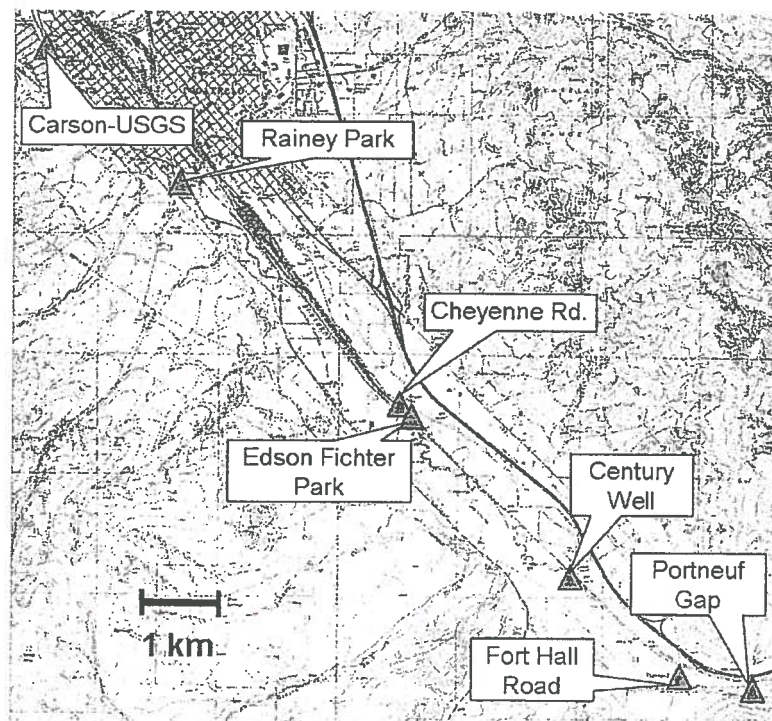


Figure 6-8: Locations of stream discharge measurements along the Portneuf River.

written comm., 2004), Chen (2001), and this study. Relative flows for the Mink Creek, Gibson Jack, and City Creek tributaries were estimated based on measurements made in 1994 in those streams.

Several discrepancies are apparent in Table 6-2. Measurements collected by Chen in August 2000 indicated that the river was losing a large amount of water between Cheyenne Road and Carson USGS gauging station (-8.4 cfs). Measurements collected by the ISU Biology Dept. in 2002 and 2004 at Rainey Park and Carson Street suggest that part of this section is actually gaining water (+4.9 to +20.79 cfs). This reach of the Portneuf has been modified with riprap and concrete, which would make such a transfer of water difficult. Furthermore, groundwater levels are below the river level in this part of the valley, which would make gains from groundwater impossible. Stream measurements in the reach from Fort Hall Road to the Century Well in August 2002 indicate a loss of river water, but in September 2002 there was a large gain for the same reach.

Such discrepancies indicate that large errors may be associated with the discharge measurements. Assuming the errors are normally distributed, the apparent gains for all possible pairs of stream flow measurements were calculated and divided by the corresponding reach distances to yield the average apparent gain per kilometer of reach (Table 6-3). The average gain for all pairs of flow measurements was -0.48 cfs/km. This value suggests that the river may be a losing stream in this part of the valley; however, its 95% confidence interval (-1.44 cfs/km to .48 cfs/km) brackets both gains and losses.

The reach of the Portneuf River between Portneuf Gap and Red Hill is approximately 9.1 km. At an infiltration rate of 0.48 cfs/km, the river would contribute a

Table 6-3: Reach gains and losses for the Portneuf River in the Pocatello Valley.
 Negative values of gain represent reaches where the river is losing water via seepage to groundwater. The amount of gain/loss per kilometer of stream reach was calculated for all sections measured. The average gain over all reaches was -0.48 cfs per kilometer.

Stream Reach		Reach Gains (cfs)	Reach Distance (km)	Gain (cfs/km)
July 1994				
Bonneville Gap	Cheyenne Rd.	4.67	7.07	-0.66
August 2000				
Fort Hall Rd.	Cheyenne Rd.	0.32	6.37	-0.05
Fort Hall Rd.	Carson-USGS	8.72	13.56	-0.64
Cheyenne Rd.	Carson-USGS	8.4	7.19	-1.17
August 2002				
Fort Hall Rd.	Century	8.47	1.55	-5.46
Fort Hall Rd.	Edson Fichter	3.23	5.96	0.54
Fort Hall Rd.	Rainey park	19.36	10.65	-1.82
Fort Hall Rd.	Carson-USGS	1.43	13.56	0.11
Century	Edson Fichter	11.7	4.41	2.65
Century	Rainey park	10.89	9.10	-1.20
Century	Carson-USGS	9.9	12.01	0.82
Edson Fichter	Rainey park	22.59	4.69	-4.82
Edson Fichter	Carson-USGS	1.8	7.60	-0.24
Rainey park	Carson-USGS	20.79	2.91	7.14
September 2002				
Fort Hall Rd.	Century	7.98	1.55	5.15
Fort Hall Rd.	Edson Fichter	6.1	5.96	-1.02
Fort Hall Rd.	Rainey park	18.97	10.65	-1.78
Fort Hall Rd.	Carson-USGS	9.69	13.56	-0.71
Century	Edson Fichter	14.08	4.41	-3.19
Century	Rainey park	26.95	9.1	-2.96
Century	Carson-USGS	17.67	12.01	-1.47
Edson Fichter	Rainey park	12.87	4.69	-2.74
Edson Fichter	Carson-USGS	3.59	7.6	-0.47
Rainey park	Carson-USGS	9.28	2.91	3.19
September 2004				
Fort Hall Rd.	Edson Fichter	8.7	5.96	-1.46
Fort Hall Rd.	Rainey park	15.4	10.65	-1.45
Fort Hall Rd.	Carson-USGS	10.5	13.56	-0.77
Edson Fichter	Rainey park	6.7	4.69	-1.43
Edson Fichter	Carson-USGS	1.8	7.6	-0.24
Rainey park	Carson-USGS	4.9	2.91	1.68
Average Gain				-0.48 cfs/km

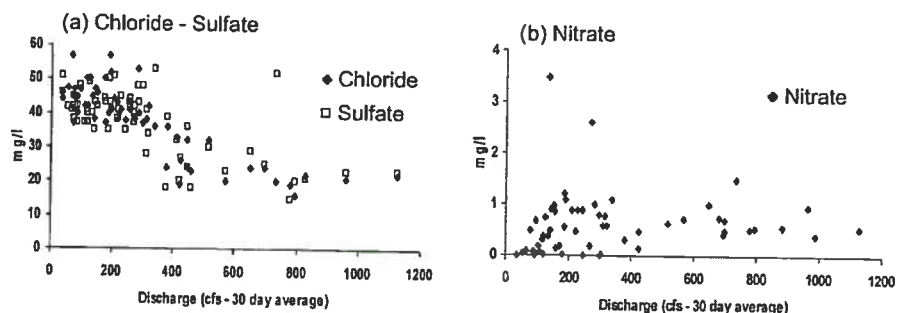
little more than 1 Bgy to groundwater recharge. This estimate was used to constrain the maximum possible river recharge in the third mixing model (Figure 6-7). The amount of river flux contributing to each mixing cell was determined by the reach distance covered by that cell.

Concentrations of NO_3 and Cl in the Portneuf River were determined from U.S. Geological Survey analyses made at the Pocatello gauging station from 1966 through 2002. Figures 6-9(a) and (b) show the relationship between river water concentrations and river discharge. Chloride and sulfate show clear trends of decreasing concentrations with greater discharges, whereas nitrate concentrations appear to be independent of river discharge (Figure 6-9(b)).

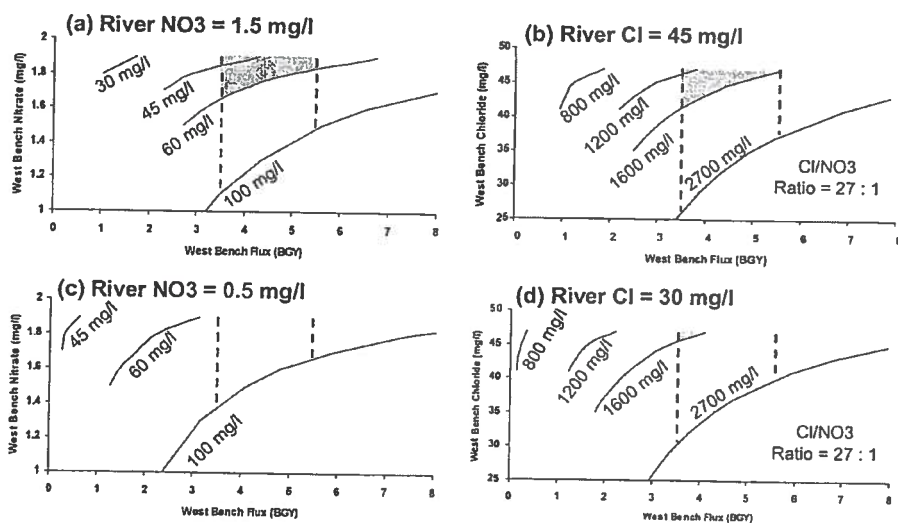
The annual mean flow for the river at Pocatello usually ranges between 150 and 400 cfs (USGS, 2005). Chloride concentrations at these flow rates would range between 30 and 45 mg/l, which were the concentrations used for modeling along with nitrate concentrations of 0.5 and 1.5 mg/l.

Figures 6-10(a) through (d) summarize the results of the mixing calculations. The model was run with septic effluent NO_3 concentrations of 30, 45, 60 and 100 mg/l and corresponding septic Cl concentrations as in previous models. The dashed lines in the figures represent the 4.5 ± 1 Bgy range of West Bench recharge necessary to maintain the water balance (after accounting for one Bgy of contributions from the river). The shaded areas represent the most likely range of West Bench background concentrations and fluxes for the range of NO_3 concentrations that are most typical of septic effluent.

Figures 6-10(a) and (b) indicate that for higher river concentrations of NO_3 (1.5 mg/l) and Cl (45 mg/l), the most likely septic effluent are > 45 mg/l $\text{NO}_3\text{-N}$ and > 1200



Figures 6-9(a) and (b): Relationship between Portneuf River water concentrations of Cl , SO_4 , and NO_3 and river discharge in Pocatello (USGS, 2005). Chloride and sulfate show clear trends of decreasing concentrations with greater discharges. Nitrate concentrations appear to be independent of river discharge.



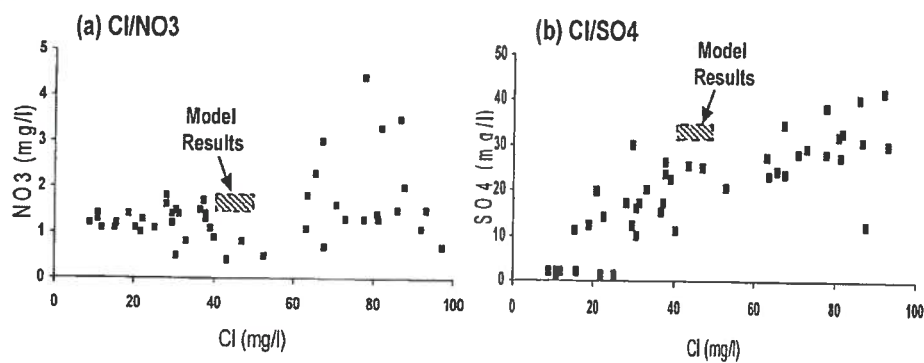
Figures 6-10(a) through (d): Septic mixing model results considering Portneuf River contributions. Solid lines represent septic effluent NO_3 concentrations of 30, 45, and 60 mg/l and corresponding septic Cl concentrations as in previous models. The dashed lines in the figures represent 4.5 ± 1 bgy of West Bench recharge after accounting for one bgy of contributions from the river. The shaded areas represent the most likely range of West Bench background concentrations and fluxes that explain the septic tracer data.

mg/l Cl. Models with lower concentrations of NO_3 and Cl (corresponding to high flow conditions) in the river do not meet the constraints of the mass balance.

6.2.4 Discussion of Main Aquifer Septic Mixing Models

Mixing models B and C indicate that if NO_3 in septic effluent is 30 to 60 mg/l, which is the range typically found, the average NO_3 and Cl concentrations in West Bench groundwater would have to be greater than 1.6 mg/l and 40 mg/l, respectively. All models show that the constraints of the water balance can be achieved with an adequate flux of Bannock Range groundwater with NO_3 concentrations less than 2 mg/l, which is within the natural background range.

The concentrations of Cl and NO_3 in West Bench water calculated in the mixing models are summarized in Figure 6-11(a) and compared to the observed concentrations in West Bench groundwater. The modeled results represent the average concentration in West Bench groundwater; as Figure 6-11(a) shows, these are within the range of observed values.



Figures 6-11(a) and (b): Comparisons of West Bench Cl, NO_3 , and SO_4 groundwater concentrations calculated by the models (hatched areas) and observed concentrations. These figures show that the model predicted values fall within the range of observed concentrations in West Bench groundwater.

The same modeling approach used for NO_3 and Cl (Model B) was used to test if septic effluent could explain the observed trend of SO_4 increases shown in Figure 5-4(b). Septic effluent was assumed to have a Cl/ SO_4 ratio of 2.5, as observed in LPRV groundwater (Figure 4-2(b)). Figure 6-11(b) summarizes the model calculated concentrations of SO_4 and Cl in average background West Bench groundwater compared to the observed concentrations in West Bench groundwater. As for NO_3 and Cl, the mixing model also reproduces observed SO_4 trends from a septic source, with background concentrations that fall within those observed in West Bench groundwater.

In contrast, river mixing models do not reproduce as coherent results. The fact that the river mixing models (Model C) require higher septic effluent concentrations to account for a river contribution also reinforces the hypothesis that the Portneuf River is not a significant component of the southern aquifer's water balance (see section 1.4.2). For example, if most river recharge occurs at high discharge (and hence lower Cl concentrations than shown in Figure 6-10 (b) and (d)), then no reasonable solution to the mass balance equations exists within the model constraints.

6.3 Mink Creek Septic Mixing Models

Septic mixing models were also used to test if elevated groundwater concentrations of NO_3 and Cl in the Mink Creek drainage can be attributed to septic effluent (see Figures 4-5(a) and (c)). One model was created for the upper part of the drainage where NO_3 concentrations rose steadily from 1.1 just below the Forest Service boundary to 6.6 just above the poultry farm (see Figure 4-5(c)). Another model evaluated if groundwater concentrations observed at the mouth of the drainage could be attributed to impacts from all septic tanks upstream. These models considered only that

groundwater which flows through the alluvium and not the bedrock.

Table 6-4 shows the parameters used to estimate the valley groundwater underflow. Cross sectional areas were estimated from surface topography and geology; gravity-derived profiles from Reid (1997) were used to define the cross-sectional geometry of alluvium at the mouth of the drainage. The hydraulic gradient (i) was assumed to be similar to the topographic gradient just above each cross sectional area being modeled. Hydraulic conductivities (K) were estimated from specific capacity measurements obtained from domestic wells completed in the alluvium in Mink Creek using methods for partial penetration correction as described in Walton (1987). Table 6-4 shows the mean K and maximum K estimated for the aquifer thickness at each cross section.

It was assumed that each septic tank upstream of each cross section was contributing 225 gpd of effluent. Models were run with effluent NO₃ concentrations of

Mink Creek Groundwater Flux Estimates

		K (ft/day)	i	Area (ft ²)	Flux (ft ³ /day)	Flux (Mgy)
Upper Mink Creek Drainage	Mean K	4.4	0.025	11,752	1,293	3.5
	Maximum K	17.5	0.025	11,752	5,142	14
Mouth of Mink Creek Drainage	Mean K	2.5	0.02	155,086	7,799	21.3
	Maximum K	10	0.02	155,086	31,017	84.7

Table 6-4: Parameters used to estimate annual groundwater fluxes in Mink Creek mixing models. Cross sectional areas were estimated from surface topography and geology, in the case of the mouth of the drainage, gravity profiles in Reid (1997) were also used. The hydraulic gradient (i) was assumed to be similar to the topographic gradient just above each cross sectional area being modeled. Mean and maximum hydraulic conductivities (mean k and max k) were calculated from specific capacity measurements from domestic wells completed in the alluvium in Mink Creek.

30, 45, and 60 mg/l. Chloride effluent concentrations were, respectively, 800, 1200, and 1600 mg/l, as constrained by their linear relationship to NO_3 in groundwater.

Background concentrations were taken from a domestic well (MC-130) sampled just below the Forest Service boundary.

Tables 6-5(a) and (b) display the results of septic mixing models for the upper section of the Mink Creek drainage above the poultry farm. The first models run with the mean hydraulic conductivity value predicted that levels of NO_3 and Cl in groundwater would be much higher than the observed concentrations of 6.6 mg/l and 180 mg/l. When the maximum hydraulic conductivity values were used, with concentrations of 60 mg/l of NO_3 and 1600 mg/l of Cl, the predicted concentrations matched more closely with observed concentrations.

In the adjusted flux model, the ground water flux was adjusted to force the predicted concentration to equal the observed concentration, assuming an average septic concentration of 45 mg/l NO_3 . This yielded a groundwater flux estimate of 9.8 million gallons per year. This flux estimate was then used in the chloride mixing model with a septic effluent Cl concentration of 1200 mg/l, based on the Cl/ NO_3 ratio observed in groundwater of 27 to 1. This gave a modeled Cl concentration (184 mg/l) that was also very close to the observed (180 mg/l).

The mouth of Mink Creek drainage was modeled in a similar way (Table 6-6). In these scenarios, model results were compared to the range of NO_3 (1.8 to 5.9 mg/l) and Cl (63 to 140 mg/l) concentrations observed in groundwater at the mouth of the drainage as displayed in Figures 4-5(a) and (c).

Upper Mink Creek Drainage

Septic Flux = 17 septic x 225 gpd = 1.4 mgy; background NO₃ = 1.1 mg/l,
Observed NO₃ = 6.6 mg/l

(a) Nitrate Models

Model Type	Ground Water Flux (Mgy)	Septic NO ₃ (mg/l)	Predicted Concentration (mg/l)	Observed (mg/l)
Mean K (4.4 ft/day)	3.5	30	9.3	6.6
	3.5	45	13.6	
	3.5	60	17.8	
Maximum K (17.5 ft/day)	14	30	3.7	6.6
	14	45	5.1	
	14	60	6.4	
Groundwater Flux Adjusted for Observed Concentration of 6.6 mg/l	9.8	45	6.6	6.6

Septic Flux = 17 septic x 225 gpd = 1.4 mgy; background NO₃ = 1.1 mg/l,
Observed NO₃ = 6.6 mg/l

(b) Chloride Models

Model Type	Ground Water Flux (mgy)	Septic Cl (mg/l)	Predicted Concentration (mg/l)	Observed (mg/l)
Mean K (4.4 ft/day)	3.5	800	254	180
	3.5	1200	368	
	3.5	1600	482	
Maximum K (17.5 ft/day)	14	800	107	180
	14	1200	143	
	14	1600	180	
Groundwater Flux Adjusted for Observed Concentration of 6.6 mg/l	9.8	1200	184	180

Tables 6-5(a) and (b): Results of septic mixing models for the upper section of the Mink Creek drainage above the poultry farm. Models using the mean hydraulic conductivity estimate predicted much higher levels of NO₃ and Cl in groundwater than were observed. Predicted values matched very closely with observed concentrations when maximum K values were used. An adjusted groundwater flux of 9.8 million gallons per year was required to match the observed NO₃ concentrations for a septic effluent concentration of 45 mg/l. This lies within the range of flux values (3.5 to 14 million gallons per year) estimated from the mean and maximum hydraulic conductivity estimates.

Mouth of Mink Creek Drainage

Septic Flux = 199 septic x 225 gpd = 16.3 mgy; background NO₃ = 1.1 mg/l,
Observed NO₃ = 1.8 – 5.9 mg/l (median = 3.4)

(a) Nitrate Models

Model Type	Ground Water Flux (Mgy)	Septic NO ₃ (mg/l)	Predicted Concentration (mg/l)	Observed (mg/l)
Mean K (2.5 ft/day)	21.3	30	13.6	1.8 - 5.9
	21.3	45	20.2	
	21.3	60	26.7	
Maximum K (10 ft/day)	84.7	30	5.8	1.8 - 5.9
	84.7	45	8.2	
	84.7	60	10.6	
Groundwater Flux Adjusted for Observed Median Concentration of 3.4 mg/l				
	300	45	32	34

Septic Flux = 199 septic x 225 gpd = 16.3 mgy; background NO₃ = 1.1 mg/l,
Observed NO₃ = 1.8 – 5.9 mg/l (median = 88)

(b) Chloride Models

Model Type	Ground Water Flux (Mgy)	Septic Cl (mg/l)	Predicted Concentration (mg/l)	Observed (mg/l)
Mean k (2.5 ft/day)	21.3	800	369	63 - 140
	21.3	1200	543	
	21.3	1600	716	
Maximum k (10 ft/day)	84.7	800	161	63 - 140
	84.7	1200	226	
	84.7	1600	291	
Groundwater Flux Adjusted for Observed Median Concentration of 88 mg/l	360	1200	88	88

Tables 6-6(a) and (b): Results of septic mixing models for the mouth of the Mink Creek drainage. The mean conductivity model results had higher concentrations of both NO₃ and Cl than was observed in groundwater. Maximum conductivity models also tended to have higher than observed concentrations, with the exception of NO₃ in septic effluent at 30 mg/l, which was close to the observed high range. The adjusted flux models for NO₃ and Cl were comparable to each other (300 and 360 mgy), but indicated that the groundwater flux would be much greater than those estimated from the mean and maximum hydraulic conductivities.

The mean hydraulic conductivity estimate produced higher concentrations of both NO_3 and Cl than were observed in groundwater, as did maximum hydraulic conductivity estimates, with the exception of NO_3 in septic effluent at 30 mg/l, which was close to the observed high range.

The adjusted ground water flux models were run with the median NO_3 (45 mg/l) and Cl (1200 mg/l) septic effluent concentrations. These models show the range of groundwater fluxes necessary for modeled results to match the median concentrations of NO_3 and Cl observed in groundwater (3.4 and 88 mg/l, respectively). Both adjusted groundwater flux models show comparable results with 300 to 360 million gallons per year necessary to match the median concentrations.

6.3.1 Discussion of Mink Creek Models

Results of mixing models in upper Mink Creek suggest that septic effluent is likely responsible for the observed trend of increasing Cl and NO_3 groundwater concentrations in this part of the drainage (see Figure 4-5(a) and (c)).

The adjusted flux models for NO_3 and Cl at the mouth of the drainage indicated that the groundwater flux would be much greater than those estimated from the mean and maximum hydraulic conductivities. There may be several possibilities for these discrepancies. Conductivity estimates from specific capacity tend to underestimate aquifer transmissivities due to the effects of well loss and partial penetration. Also, these models considered only groundwater flowing through the alluvium, and not bedrock. As discussed in section 4.3.1, fracture flow in Mink Creek is likely important. The higher flow estimates are more comparable to the water balance estimate of about 5.5 billion gallons a year from the southern Bannock Range determined in Welhan et al. (1996).

Mink Creek is the largest drainage in the LPRV and likely makes up a large portion of recharge from the Bannock Range.

The main aquifer septic mixing models in Section 6.2 also suggest that recharge from the Mink Creek drainage would be greater. The flux of West Bench recharge into cell M2, the cell at the mouth of Mink Creek, was generally greater than two billion gallons per year for models that met the constraints of the water balance. To achieve a cross sectional area along the western boundary of Cell M2 that was large enough to accommodate a West Bench recharge of approximately 2 billion gallons per year and a flux that is areally proportional to the 300 to 360 million gallons a year calculated for the cross sectional area of alluvium used in Table 6-6, the depth of the main aquifer would have to be approximately 150 feet. This depth is within the range of aquifer thickness in the southern LPRV, and indicates that the model flux calculations can be reconciled with physical constraints as well as chemical mass balance.

Chapter 7: Summary and Conclusions

7.1 Summary

The Lower Portneuf River Valley (LPRV) aquifer supplies the entire water needs to rural and urban residents of Pocatello and Chubbuck, Idaho. This aquifer system is comprised of two very different subsystems: the northern aquifer system and the southern aquifer system (Welhan et al., 1996) (Figure 1-3). This study focused primarily on the geochemistry of water in the southern aquifer and the adjacent recharge areas of the system in the context of what sources may be responsible for impacting water quality.

Several localized areas of groundwater have been affected by nitrate and salts such as sulfate, chloride, sodium, and calcium, which are usually associated with non-point sources like septic leachate, road salting, and runoff from agriculture and ranching operations (Panno et al., 2002; Wilhelm et al., 1996). Of the order of 672 septic drain fields overlie portions of the southern aquifer and its recharge areas in the Bannock Range and upper Portneuf watershed, and there has previously been no broad assessment of their impact on LPRV groundwater. This thesis attempted to address this question and to identify and quantify the sources of inorganic salts contributing to the southern LPRV aquifer by summarizing and evaluating the best available information on the inorganic geochemistry of the southern aquifer and its recharge sources.

Groundwater and surface water samples were collected from the southern aquifer and its recharge areas during several sampling events over a year-long period from June of 1993 through July of 1994. Many of these locations were re-sampled in the summer of 2004 to evaluate changes in aquifer chemistry that may be due to anthropogenic sources. Historical data from government agencies was incorporated in this analysis to form a

more complete picture of the spatial and temporal trends observed in groundwater chemistry.

Groundwater recharge to the southern LPRV aquifer originates from the upper Portneuf River basin, the eastern aquifer system, and the southern Bannock Range. The chemistry of groundwater originating from the upper basin through the Portneuf Gap is very similar to the chemistry of groundwater in the southern aquifer. Groundwater in the eastern aquifer is elevated in salts and nitrates, and is believed to be impacted by septic effluent (Winter et al., 1994). Groundwaters in the Bannock Range are the most chemically diverse, with ionic concentrations that range from the lowest to some of the highest concentrations found in the LPRV.

At a vast majority of sample sites, all major ions in groundwater, including NO_3 , SO_4 , hardness (CaCO_3), Ca, Mg, Na, and Si, show strong positive correlations with Cl, suggesting that mixing with a common source is responsible for adding these salts to groundwater. At some sample locations, $\text{NO}_3\text{-N}$ concentrations remained at near-background levels ($\approx 2 \text{ mg/l}$), independent of Cl and other salts, possibly reflecting depletion of nitrate in anaerobic groundwaters by denitrification reactions.

In this work, an emphasis was placed on evaluating Cl, SO_4 , and NO_3 mixing relationships, due to the conservative nature of these anions in groundwater, and because they are commonly associated with the contaminant sources described above. While concentrations of these constituents at most sample locations in the southern aquifer cluster relatively tightly around their respective mean concentrations, a number of sample locations show concentrations at the high and low ranges, indicating mixtures of water representing recharge and contaminant sources. Mixing trends in the southern aquifer

provide evidence for three types of groundwater found in the West Bench drainages of the Bannock Range and eastern aquifer. These have been described as high-TDS/high- NO_3 water, high-TDS/low- NO_3 water, and low-TDS/low- NO_3 water.

Most wells completed in the Tertiary Starlight Formation that forms the basement of the southern aquifer, exhibit low-TDS/low- NO_3 type waters. Two wells recently drilled in the southern aquifer and completed in the Tertiary sediments are flowing-artesian wells and have low-TDS type waters, providing strong evidence that low-TDS Bannock Range waters are recharging the aquifer from beneath the Upper Gravels, as well as laterally along the western margin of the aquifer.

Sample locations with elevated salt concentrations (high-TDS type waters) in the southern aquifer tend to occur near the margins of the aquifer, possibly reflecting influence from lateral recharge. Salt contamination observed along the north-eastern boundary of the southern aquifer indicates impacts from the eastern aquifer, as reflected in coupled increases of Cl , SO_4 , and $\text{NO}_3\text{-N}$. The area of contamination is most pronounced at three municipal wells in Ross Park where large pumping withdrawals may be drawing contaminated waters from the eastern aquifer.

Small plumes of high-TDS/Low- NO_3 type water near the western margin of the aquifer may result from localized sources of contamination originating near the surface, or possibly from waters in the West Bench area that are already elevated in salts.

Aside from a few areas of clustered similar water types, there is a fairly high degree of spatial variability in water composition over the study area, with sharp changes occurring over short distances. In the Mink Creek drainage, this variability may be related in part to the heterogeneous nature of groundwater transport in fracture-dominated

flow, but in general, groundwater chemistry in the LPRV is the result of a combination of factors which include: depth of well completion, proximity to near-surface sources of contamination, and the lithology of the water bearing unit(s).

Wells producing water closer to the water table tend to have higher concentrations of nitrate and salts than deeper wells. Potential sources include landfills, chemical fertilizers, animal waste, road salt, road aggregate, water softeners, natural sources, and sewer and septic tank effluent. The spatial relation between septic tanks and groundwater with high TDS and $\text{NO}_3\text{-N}$ concentrations suggests that septic effluent may be the principal factor impairing groundwater quality in the LPRV.

A comparison of more than 80 locations sampled in 1994 and 2004 showed no discernable temporal trends of $\text{NO}_3\text{-N}$ concentrations in groundwater. Concentrations of Cl and SO_4 appear to have decreased as much as 10 mg/l in the southern aquifer at many locations, possibly due to increased river losses in the upper basin due to drought conditions.

Spatial trends along the groundwater flow path of the southern aquifer were identified, where statistically significant increases in NO_3 and Cl were observed along the 12 km length of the aquifer (the increase in SO_4 concentration was not statistically significant). A five-compartment box model was used to evaluate the mixing relationships in the southern aquifer between septic effluent, recharge water, river leakage, and aquifer throughflow to determine if septic effluent could account for the observed concentration increases along the flow path.

Reasonable solutions to the mixing models were constrained by the water balance for the LPRV (Welhan et al. (1996) and updated in Appendix C). At typical

septic effluent $\text{NO}_3\text{-N}$ concentrations (30 to 60 mg/l), and corresponding Cl in septic effluent (constrained by the 27:1 ratio observed in groundwater), the cumulative septic impacts in the LPRV match the observed concentration increases along the aquifer flow path.

The mixing models demonstrate that constraints on the water balance can be met with an appropriate flux of Bannock Range recharge having $\text{NO}_3\text{-N}$ concentrations less than 2 mg/l, which is within the natural background range. Models that incorporated contributions from the Portneuf River required higher septic effluent concentrations, and reinforced the hypothesis that the river is not a significant contributor to the southern aquifer's water balance.

Similar mixing models were completed for two reaches of Mink Creek, a tributary drainage having no potential sources of ground water contamination other than septic effluent. The Mink Creek models showed comparable results and verified that impacts from septic fields account for the trends observed in groundwater within the range of expected fluxes of recharge water.

7.2 Conclusions

Approximately 672 septic fields are located in the southern LPRV, annually contributing about 50 million gallons of effluent to groundwater, 6 to 11 metric tons of $\text{NO}_3\text{-N}$ and more than 80 metric tons of Cl. The spatial relation between septic tanks and groundwater impacted by salts and NO_3 , and the similarity in the chemical signatures of septic effluent to that of impacted groundwater, indicates that septic impacts are cumulative and that septic effluent is the principal factor impairing groundwater quality in the southern LPRV.

Mixing models demonstrate that the cumulative impacts of septic tanks in the LPRV account for the observed trends of $\text{NO}_3\text{-N}$ and Cl in groundwater in the southern LPRV and indicate that such models could be used to predict the impacts on wastes of future septic-based growth and/or management policies.

7.3 Recommendations for Further Study

The results of this study provide the strongest evidence to date of the magnitude of the cumulative effects of septic effluent on groundwater in the LPRV, and may be used to predict future water quality trends in the management and development of water resources in this aquifer.

Future work on the impact of septic effluent in the LPRV could involve evaluating tracers not available in this thesis investigation. Nitrogen isotopes ($^{15}\text{N} / ^{14}\text{N}$) can be useful diagnostic parameters of septic source investigations. Groundwater impacted by septic effluent is typically enriched in ^{15}N due to biological fractionation, while NO_3 originating from chemical fertilizers is depleted in ^{15}N .

Other tracers used to fingerprint septic effluent in groundwater include volatile organic compounds, pharmaceuticals, phosphorous, caffeine, and bromide, and could be evaluated with mixing models similar to those developed in this study, to refine current and future estimates of septic impact and principal pathways of groundwater recharge to the system.

The box models presented in Chapter 6 may also be refined to further investigate the spatial pattern of principal recharge zones in the southern aquifer. The models presented in this study relied on the regression analyses that revealed linear increasing trends of Cl , NO_3 , and SO_4 along the flow path of the southern aquifer. Future modeling

work could include segregating the southern aquifer longitudinally, in parallel flow tubes, which may reveal non-linear trends in water chemistry along flow paths that could be used to define spatial variations in West Bench recharge and contaminant fluxes.

REFERENCES CITED

- Armstrong, F.C., and Oriel, S.S., 1965, Tectonic development of the Idaho-Wyoming thrust belt: American Association of Petroleum Geologists Bulletin, v.49, p.1847-1866.
- Bannock County, 1993, Letter of Intent, Fugitive road dust control strategies, Bannock County highway Department, November 8, 1993, 2 pp.
- Bannock County, 2004, Remediation monitoring summary report, Fort Hall Canyon landfill, Bannock County, Idaho, prepared by Maxim Technologies, 32 pp.
- Barton, G.J., 2004, Surface and ground-water relations on the Portneuf River, and temporal changes in ground-water levels in the Portneuf Valley, Caribou and Bannock Counties, 2001-2, U.S. Geological Survey, Scientific Investigations report 2004-5170, 34 pp.
- Barton, L. and Colmer, T., 2004, Irrigation and fertilizer strategies for minimizing nitrogen leaching from turfgrass, Fourth International Crop Science Congress, 15 pp.
- Bauman, B.J. and W.M. Schafer, 1985, Estimating ground-water quality impacts from on-site sewage treatment systems, in on-site wastewater treatment: Proceedings of the seventh international symposium on individual and small community sewage systems, American Society of Agricultural Engineers, St. Joseph, MI.
- BBC Research and Consulting, 2001, Economic impacts of enhanced aquifer protection for the lower Portneuf river valley, prepared for the Idaho Department of Environmental Quality, 68pp.
- Billings, G.K. and Williams, H.H., 1967, Distribution of chlorine in terrestrial rocks – A discussion: *Geochemica et Cosmochimica Acta*, v. 31, p. 2247
- Brady, K., Kania, T., Smith, M.W., and Hornberger, R.J., 2000, Coal mine drainage prediction and pollution prevention in Pennsylvania, Pennsylvania Department of Environmental Protection, Harrisburg, PA, 305 pp.
- Brown and Caldwell, 1994, Phase II Hydrogeologic assessment draft report, Bannock County, Idaho
- Boyce, J.S., Muir, J., Edwards, A.P., Seim, E.C., and Olson, R.A., 1976, Geologic nitrogen in Pleistocene loess of Nebraska, *Journal of Environmental Quality*, vol. 5, no. 1, p. 93-96.
- CH₂M-Hill 1994a, Final site assessment report, Pocatello Creek landfill hydrogeologic investigation, Pocatello, Idaho, 85 pp.

- CH₂M-Hill 1994b, Hydrogeology and assessment of TCE contamination in the southern portion of the Pocatello aquifer - Phase I Aquifer Management Plan final report, 100 pp.
- CH₂M-Hill 1994c, Results of the test drilling program for the City of Chubbuck, Idaho – Final Report, 26 pp.
- Chen, S., 2001, Hydrogeochemical Studies of the Portneuf River, Southeastern Idaho, [M.S. thesis]: Pocatello, Idaho State University, 134pp.
- Christiansen, R.L., and McKee, E.H., 1978, Late Cenozoic volcanic and tectonic evolution of the Great Basin and Columbia intermontane regions, *in* Smith, R.B., and Eaton, G.P., eds., *Cenozoic Tectonics and Regional Geophysics of the Western Cordillera*: Geological Society of America Memoir 152, p. 283-311.
- City of Pocatello, 1993, Letter of Intent, Fugitive road dust control strategies, Particulate matter (PM₁₀) reduction agreement, November 23, 1993, 2 pp.
- Corbett, M.K., Anderson, J.E. and Mitchell, J.C. 1980. An evaluation of thermal water occurrences in the Tyhee Area, Bannock County, Idaho; Idaho Department of Water Resources, Water Information Bulletin N. 30, Geothermal Investigations in Idaho, Part 10, 67 pp.
- Davis, J.G., Swinker, A.M., 2004, Horse manure management, University Cooperative extension, Colorado State University, <http://www.ext.colostate.edu/pubs/livestk/01219.html>, 6pp.
- Ecochem, 2005, Manure-an excellent fertilizer, http://www.ecochem.com/t_manure_fert.html
- Freeze, R.A. and Cherry, J.A., 1979, *Groundwater*; Prentice-Hall, New Jersey; 604 pp.
- Gellenbeck, D.J., 1994, Isotopic compositions and sources of nitrate in ground water from western salt river valley, Arizona, U.S. Geological Survey, Water-Resources Investigations report 94-4063, 53 pp.
- HACH, 1988, Digital titrator manual, model 16900-01, HACH company, 121pp.
- HACH, 1991, DR/2000 Spectrophotometer instrument manual, HACH company, 521pp.
- Ham, J.M., Reddi L.N., and Rice, C.W., 1999, Animal waste lagoon water quality study, A research report by Kansas State University, 12 pp.
- Hantzsche, N.N. and Finnemore, J.E., 1992, Predicting ground-water nitrate-nitrogen impacts, *Ground Water*, Vol. 30, No. 4, p. 490-499.

- Harman, J., Robertson, W.D., Cherry, J.A., and Zanini, L., 1996, Impacts on a sand aquifer from an old septic system: nitrate and phosphate, *Ground Water*, vol. 34, p. 1105-1114.
- Hem, J.D., 1985, Study and interpretation of the chemical characteristics of natural water, U.S. Geological Survey, Water Supply Paper 2254, 263 pp.
- Hendry, M.J., McCreedy, R.G., and Gould, W.D., 1984, Distribution, source and evolution of nitrate in a glacial till of southern Alberta, Canada, *Journal of Hydrology*, vol. 70, p. 177-198.
- Holloway, J.M., Dahlgren, R.A., Hansen, B., and Casey, W.H., 1998, Contribution of bedrock nitrogen to high nitrate concentrations in stream water, *Nature*, vol. 395, p. 785-788
- ITD, 1993, Letter of Intent, Fugitive road dust control strategies, Idaho Transportation Department, December 1, 1993, 2 pp.
- IDWR, 2003, Memorandum: Evaluation of ground water conditions at Wildhorse Ridge PWS wells, Idaho Dept. of Water resources, from Joe Baldwin to Barbara Jones, IDEQ, 9 pp.
- Isaaks, E.H. and Srivastava, R.M., 1989, *Applied Geostatistics*, Oxford University Press, New York, New York.
- ISWS, 2004, Nitrogen cycles project, Illinois State Water Survey, <http://www.sws.uiuc.edu/nitro/detail.asp?lpg=areas&type=geosphere>
- ITRC, 2000, Emerging technologies for enhanced in situ biodenitrification of nitrate contaminated ground water, Interstate Technology and Regulatory Cooperation work group, 73 pp.
- Kellogg, K.S., 1992, Cretaceous thrusting and Neogene block rotation in the northern Portneuf Range region, southeastern Idaho, *in* Link, P.K., Kuntz, M.A., and Platt, L.B., eds., *Regional Geology of Eastern Idaho and Western Wyoming: Geological Society of America Memoir 179*, p. 95-113.
- Kindel, G., Ore, H.T. and Welhan J.A., 1991, Geological and hydrogeological characterization of the Fort Hall Canyon landfill site, Bannock County, Idaho; *Proc., 27th Engineering Geology and Geological Engineering Symp.*, Logan, Utah, pp. 33-1 to 33-7.
- Kuntz J.E., McBride, A., 2004, Salt in well water, Is softener in the background?, Department of health and Social Services, City of Stamford, CT, 9 pp.

- Libra, R.D., Quade, D.J., and Seigley, L.S., 1998, Ground water monitoring at earthen manure-storage structures in Iowa, Managing Manure with the Environment and Society Conference, Ames, Iowa, February 1998, 14pp.
- Link, P.K., and Stanford, L.R., 1999, Geologic map compilation of the Pocatello 30x60 minute Quadrangle, Idaho: Idaho Geological Survey Technical Report 99-02, scale 1:100,000.
- Link, P.K., Jansen, S.T., Halimdihardja, P., Lande, A., Zahn, P., 1987, Stratigraphy of the Brigham Group (Late Proterozoic-Cambrian), Bannock, Portneuf, and Bear River Ranges, southeastern Idaho, *in* Miller, W.R., ed., The Thrust Belt Revisited: Wyoming Geological Association 38th Annual Field Conference Guidebook, p.133-148.
- Long, A.E., Brown, J.G., Gellenbeck, D.J., 1997, Statistical analysis of nitrate in ground water, West Salt River Valley, Arizona, USGS Water-resource investigations report 97-4185, 38 pp.
- Lowe, M., Wallace, J., Bishop, C.E., and Hurlow, H.A., 2004, Ground water quality classification and recommended septic tank soil-adsorption-system density maps, Castle County, Utah, Utah Geological Survey, Special Study 113, 22 pp.
- McQuillan, D., 2004, Ground water quality impacts from on-site septic systems, National Onsite Wastewater Recycling Association, 13th Annual Conference, Albuquerque, N.M., November 7-10, 2004, 13 pp.
- Meehan, C. and Welhan, J.A. 1994, Impact of leachable sulfate on the quality of groundwater in the Pocatello aquifer; Proc. 30th Symp. Engineering Geology and Geotechnical Engineering, Boise, ID; pp.19-35.
- Minnesota Pollution Control Agency, 2000, Ground water quality under three unsewered subdivisions in Minnesota, Ground Water Monitoring and Assessment Program, 47 pp.
- Molenat, J., Durand, P., Gascuel-Oudou, C., Davy, P., Gruau, G., 2002, Mechanisms of nitrate transfer from soil to stream in an agricultural watershed of French Brittany, Water Air and Soil Pollution, vol. 133, p. 161-183.
- Montgomery Watson, 1996, City of Pocatello sewer extension facilities plan, 23pp.
- NDSU Extension Service, 2004, Forms of sulfur fertilizer, North Dakota State University, <http://www.ag.ndsu.nodak.edu/aginfo/procrop/fer/ammsu104.htm>.
- Norvitch, R.F. and Larson, A.L. 1970, A reconnaissance of the water resources in the Portneuf River Basin; Idaho Dept. Water Resources, Water Information Bull 16, 58pp.

- O'Connor, J.E., 1990, Hydrology, hydraulics, and sediment transport of Pleistocene Lake Bonneville Flooding on the Snake River, Idaho [PhD Dissertation]: University of Arizona, 192p.
- Osier, T.A., 2004, The Neogene-Quaternary Topographic Development of the Portneuf Valley, Bannock County, Southeast Idaho [M.S. thesis]: Pocatello, Idaho State University, 106pp.
- Panno, S.V., Hackley, K.C., Hwang, H.H., Greenberg, S., Krapac, I.G., Landsberger, S., and O'Kelley, D.J., 2002, Source identification of sodium and chloride contamination in natural waters: Preliminary results. In Proceedings of the 12th Annual Conference of the Illinois Groundwater Consortium.
- Parliman, D.J., 2002, Analysis of nitrate (NO₃-N) concentration trends in 25 ground-water-quality management areas, Idaho, 1961-2001: USGS, Water resources Investigations report 02-4056, 18pp.
- Queensland Government, 2004, Feedlot waste management manure production data, Environmental Management, Department of primary industries and Fisheries, <http://www.dpi.qld.gov.au/environment/5166.html>, 5pp.
- Reid, T.V., 1997, Gravity-derived subsurface paleomorphology and structure of the Lower Portneuf River Valley, Pocatello, Idaho [M.S. thesis]: Pocatello, Idaho State University, 72p.
- Robertson, F.N., 1991, Geochemistry of ground water in alluvial basins and Arizona and adjacent parts of Nevada, New Mexico, and California, U.S Geological Survey Professional Paper 1406-C, 90 pp.
- Robertson, W.D., Cherry, J.A., and Sudwicky, E.A., 1991, Ground-water contamination from two small septic systems on sand aquifers, Ground Water, vol 36, p. 82-92.
- Robertson, W.D., and Blowes, D.W., 1995, Major ion and trace metal geochemistry of an acidic septic-system plume in silt, Ground Water, vol. 33, p. 275-283.
- Rodgers, D.W., and Janecke, S.U., 1992, Tertiary paleogeologic maps of the western Idaho-Wyoming-Montana thrust belt, in Link, P.K., Kuntz, M.A., and Platt, L.B., eds., Regional Geology of Eastern Idaho and Western Wyoming: Geological Society of America Memoir 179, p. 83-94.
- Rodgers, D.W., and K.L., Othberg, 1999, Geologic map of the Pocatello South Quadrangle, Bannock and Power Counties, Idaho: Idaho Geological Survey Technical Report, GM-26, scale 1:24,000.
- SCVJSS, 2002, Chloride Source Report, Santa Clarita Valley Joint Sewerage System, http://www.lacsd.org/chloride/images/Chloride_Report.PDF

- Scott, W.E., Pierce, K.L., Bradbury, J.P., and Forester, R.M., 1982, Revised Quaternary stratigraphy and chronology in the American Falls area, southeastern Idaho, *in* Bonnichsen, B., and Breckenridge, R.M., eds., *Cenozoic Geology of Idaho*: Idaho Bureau of Mines and Geology Bulletin 26, p. 581-595.
- Stark, J., Brown, B., and Shewmaker, G., 2002, Southern Idaho fertilizer guide, *Irrigated Alfalfa*, University of Idaho, College of Agricultural and Life Sciences
- Stark, J., Westerman, and Hopkins, B., 2004, Nutrient management guidelines for Russet Burbank Potatoes, University of Idaho, College of Agricultural and Life Sciences
- Steinheimer, T.R., and Scoggin, K.D., 1998, Agricultural chemical movement through a field-size watershed in Iowa: Subsurface hydrology and distribution of nitrate in groundwater, *Environmental Science and Technology*, vol. 32, no. 8, p. 1039-1047.
- Stewart, J.H., 1972, Initial deposits in the Cordillera geosyncline: Evidence of a Late Precambrian (850 m.y.) continental separation: *Geological Society of America Bulletin*, v. 83, p. 1345-1360.
- Stewart, J.H., 1998, Regional characteristics, tilt domains, and extensional history of the late Cenozoic Basin and Range province, western North America, *in* Faults, J.E., and Stewart, J.H., eds., *Accommodation Zones and Transfer Zones: The Regional Segmentation of the Basin and Range Province*: Geological Society of America Special Paper 323, p.47-74.
- Thomas, J.M., Thodal, C.E., Seiler, R.L., 1999, Identification of nitrate sources contributing to ground water in the Indian Hills area of Douglas County, Nevada, *USGS Water-Resources Investigations report 99-4042*, 22p.
- Till, R., 1974, *Statistical methods for the earth scientists*, Halsted Press, 148 pp.
- Titus, F.B., 1980, Groundwater in the Sandia and northern Manzano Mountains, New Mexico, *New Mexico Bureau of Mines and Mineral Resources, Hydrologic report 5*, 23 pp.
- Trimble, D.E., 1976, *Geology of the Michaud and Pocatello quadrangle, Bannock and Power counties, Idaho*: U.S. Geological Survey Bulletin 1440, 88p.
- U.S. EPA, 2003, IsoSource: stable isotope mixing model for partitioning an excess number of sources, U.S. Environmental Protection Agency, National Health and Environmental Effects Research Laboratory, download from <http://www.epa.gov/wed/pages/models/isosource/isosource.htm>
- USGS, 2005, Streamflow statistics for Idaho for the Portneuf River at Pocatello, Idaho, U.S. Geological Survey, <http://nwis.waterdata.usgs.gov/id/nwis/>

- Walton, C.W., 1987, Groundwater Pumping Tests: Design and Analysis, Lewis publishers Inc., Chelsea, Michigan, 201 p.
- Welhan J., 2001a, Ground-water interactions near the highway Pond Gravel Pit, Pocatello, Idaho, Idaho geological Survey, 23 pp.
- Welhan J., 2001b, Water balance and total pumping capacity of the lower Portneuf river valley aquifer, Idaho Geological Survey, 27pp.
- Welhan, J., and Meehan, C., 1994, Hydrogeology of the Pocatello aquifer: Implications for wellhead protection strategies: *in* Link, P.K., ed., Proceeding of the 30th Symposium on Engineering Geology and Geotechnical Engineering Symposium 1994: Hydrogeology, Waste Disposal, Science and Politics, p. 1-18.
- Welhan, J., Meehan, C., and Reid, T., 1996, The lower Portneuf River valley aquifer: A geologic/hydrologic model and its implications for wellhead protection strategies: EPA Wellhead Protection Demonstration Project and the City of Pocatello Aquifer Geologic Characterization Project, 41p.
- West, D., 2001, Nitrates in ground water, a continuing issue for Idaho citizens, Idaho Department of Environmental Quality, 21pp.
- Wilhelm, S.R., Shiff, S.L., and Cherry, J.A., 1994, Biogeochemical evolution of domestic waste water in septic sytems: 1. conceptual model, Ground Water, Vol. 32, No. 6, p. 905-916.
- Wilhelm, S.R., Shiff, S.L., and Robertson, W.D., 1996, Biogeochemical evolution of domestic waste water in septic systems: 2. Application of conceptual model in sandy aquifers, Ground Water, Vol. 34, No. 5, p. 853-864.
- Williams, L.M., 1997, Evaluation of quality assurance/quality control data collected by the U.S. Geological Survey for water-quality activities at the Idaho National Engineering Laboratory, Idaho, 1994 through 1995: USGS Water-Resource Investigations report 97-4058, 87 p.
- Winter, G., Baldwin, J., Wicherski, B. and Richlen, S. 1994. Black Cliffs ground water quality study: nitrate impact; IDEQ, Ground Water Quality Technical Report No. 4; 23 pp.
- Younos, T., Wright, B.A., Reaves, D.W., 1999, The potential for developing mine cavity water for water supplies: Institutional and water quality issues, Virginia Water Resources Center, Virginia Tech, Blacksburg, Virginia, Special Report SR12-1999, 63 pp.

Zielinski, R.A., Otton, J.K., and Johnson, C.A., 2001, Sources of salinity near a coal mine spoil pile, north-central Colorado, *Journal of Environmental Quality*, vol. 30, p. 1237-1248

**Appendix A: Database of Inorganic Groundwater Chemistry from Wells in the
Lower Portneuf River Valley**

The database for this appendix is included on to the CD-Rom that accompanies this thesis. Data were obtained from the City of Pocatello, the Idaho Department of Environmental Quality, and the Idaho Department of Water Resources, cross-checked and verified where possible.

Appendix B: Analytical Results for Water Samples Collected in 1993-1994 and 2004 (see legend on p. 156)

Location ID	Area	Date	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	HCO3 (mg/L)	Cl (mg/L)	SO4 (mg/L)	NO3-N (mg/L)	Si (mg/L)	T Hard. (mg/L)	SC (us)
EA-01	EA	7/20/1993	170.1	98.4	59.7	5.14	203	410	172.5	23.24	23.6	836	1600
EA-01	EA	12/6/1993	179	98	57	4	190	492	165	27.09	19.1	1140	1500
EA-01	EA	2/26/1994	202	128.4			210	396	220	24.38	20.0	1040	2200
EA-01	EA	7/1/1994	130	96			195	345	170	15.89	21.0	720	1600
EA-01	EA	8/28/2004						543	192	25.8			2250
EA-02	EA	6/21/1993	128	58.8	69	6.62	298	252.5	110	8.64	22.2	592	1100
EA-02	EA	12/6/1993	154	53	80.5	8.4	297	207	105	8.58	21.5	608	1000
EA-02	EA	2/26/1994	130	61.2			293	225	140	8.13	23.0	580	1400
EA-02	EA	7/1/1994	141	60			284	280	115	7.65	22.0	600	1300
EA-02	EA	8/25/2004						291	115	9.7			1536
EA-03	EA	7/20/1993	111	69.6	48.2	3.77	188	280	117.5	12.32	21.1	568	1100
EA-03	EA	12/6/1993	139	71	57.3	3.3	195	212	125	14.90	18.3	590	1100
EA-03	EA	2/26/1994	110.8	80.4			195	200	145	14.00	20.0	612	1400
EA-03	EA	7/1/1994	112	77			205	268	135	11.22	20.0	598	1300
EA-03	EA	8/25/2004						284	150	13.2			1575
HP-01	MA	8/25/2004	78					43	36	1.9			
HP-02	MA	8/25/2004	77					44	38	2.1			
HP-03	MA	8/25/2004						47	38	2.2			
MA-01	MA	7/20/1993	94.19	32.4	37.4	9.84	405	46.75	49	2.58	38.5	371	700
MA-01	MA	11/28/1993	77	38	55	10.4	405	54.3	50	2.28	24.6	352	650
MA-01	MA	3/2/1994	98	34.8			281	54	58	2.57	27.0	390	700
MA-01	MA	7/1/1994	98	33			332	61.5	53	2.48	25.0	380	700
MA-01	MA	8/11/2004	60					46	36	1.6			648
MA-02	MA	7/20/1993	70.74	31.2	30.4	7.8	345	48	45	2.38	27.4	307	550
MA-02	MA	3/2/1994	78	28.04			320	47	45	1.59	25.0	316	550
MA-02	MA	7/1/1994	77	30			327	51.5	47	1.69	26.0	315	600
MA-02	MA	8/11/2004	56					44	38	2.4			682
MA-03	MA	7/20/1993	68.26	38.4	48.8	10.7	402	46	52	3.38	40.0	331	650
MA-03	MA	12/9/1993	78	38	69	11.1	448	49.6	60	2.88	23.0	355	650
MA-03	MA	3/2/1994	98.8	38.84			454	58	62	2.98	28.0	408	700
MA-03	MA	7/12/1994	71	37			405	59.5	50	2.28	25.0	330	500
MA-03	MA	8/11/2004	77					61	49	5.6			872
MA-04	MA	6/30/1993	68	27.6	39.6	5.13	327	39.25	42	1.59	24.8	305	460
MA-04	MA	12/9/1993	71	31	44	7.3	326	40	45	1.99	22.6	308	520
MA-04	MA	3/2/1994	79.2	26.88			320	40	44	0.79	14.4	310	490
MA-04	MA	7/1/1994	79	28			315	46.5	47	1.59	21.0	312	500
MA-04	MA	8/11/2004	53					48	34	1.8			627

Appendix B: Analytical Results for Water Samples Collected in 1993-1994 and 2004 (see legend on p. 156)

Location ID	Area	Date	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	HCO3 (mg/L)	Cl (mg/L)	SO4 (mg/L)	NO3-N (mg/L)	Si (mg/L)	T Hard. (mg/L)	SC (us)
MA-05	MA	6/30/1993	71.4	28.8	40.8	4.99	327	40.5	41	1.49	25.7	302	550
MA-05	MA	12/9/1993	73	31			330	40	45	1.89	23.0	310	500
MA-05	MA	8/11/2004						50	35	1.5			638
MA-06	MA	6/30/1993	69.7	49.2	79.6	0.77	320	90	57	6.65	15.8	383	750
MA-06	MA	11/28/1993	86	42			340	98	61	4.87	24.0	388	700
MA-06	MA	7/5/1994	74	48			335	91	58	6.06	25.0	384	700
MA-06	MA	8/25/2004	101					72	33	1.8		352	759
MA-12	MA	11/28/1993	75	28			310	49	45	2.57	24.0	303	500
MA-12	MA	3/2/1994	77.2	29.52			334	52	49	2.09	25.0	316	800
MA-12	MA	5/23/1994					60	53	53	2.28	26.0	326	600
MA-12	MA	7/1/1994	81	32			364	63	57	2.58	26.0	333	650
MA-17	MA	5/28/1994					19		1	0.99			
MA-17	MA	5/29/1994					24		2	0.99			
MA-17	MA	7/12/1994	51	10			128	20.5	12	0.99	12.0	170	200
MA-17	MA	8/11/2004	47					48	25	1.9		164	548
MA-19	MA	6/15/1993	66.4	27.84	45.1	7.36	337	43.5	44	1.39	25.25	292	406
MA-20	MA	6/4/1993	78	38.4			376	50	90	0.30	16.975	366	725
MA-20	MA	6/21/1993	68	38.4	43.4	10.36	373	45.25	100	0.70	35.65	340	725
MA-20	MA	2/26/1994	42.4	31.2			261	36	44	0.50	17	236	550
MA-20	MA	7/5/1994	64	30			271	64.5	43	1.59	19	285	500
MA-21	MA	6/4/1993	66	28.8			301	33.5	42	1.59	13.875	260	575
MA-21	MA	6/21/1993	50.86667	26.4	30	5.26	290	42.25	44	0.79	11.7	261	475
MA-21	MA	12/6/1993	86	29	44	5.7	297	33.5	38	1.59	22.3	268	550
MA-21	MA	2/26/1994	37.2	28.08			242	30.75	42	0.79	18	210	500
MA-21	MA	7/5/1994	60	27			247	57	40	1.48	21	262	450
MA-22	MA	6/4/1993	74	32.4			333	32	42	1.69	22.05	300	600
MA-22	MA	6/21/1993	73	27	36.1	6.76	328	34.5	44	1.59	23.8	306	475
MA-22	MA	12/6/1993	93	32	43.5	5.8	342	41.5	39	1.69	24.7	310	450
MA-22	MA	2/26/1994	71.2	32.64			349	31.5	48	1.49	26	314	750
MA-22	MA	7/5/1994	70	27			331	89	26	2.09	21	285	350
MA-23	MA	5/23/1994						54.5	47	1.79			550
MA-23	MA	8/11/2004	72					48	34	1.6		273	
MA-24	MA	6/15/1993	77	38.4	53.7	22.2	356	54.25	127.5	3.38	24.875	361	800
MA-24	MA	12/6/1993	71	26	46.2	15	301	45	60	1.99	27.6	279	800
MA-24	MA	2/26/1994	65.2	30.96			312	56	61	2.09	26	292	750
MA-24	MA	7/5/1994	69	31			327	92	64	2.09	27	299	850

Appendix B: Analytical Results for Water Samples Collected in 1993-1994 and 2004 (see legend on p. 156)

Location ID	Area	Date	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	HCO3 (mg/L)	Cl (mg/L)	SO4 (mg/L)	NO3-N (mg/L)	SI (mg/L)	T Hard. (mg/L)	SC (us)
MA-25	MA	5/27/1994						41.5	32	1.88			
MA-25	MA	7/8/1994	62	25			257	41.5	34	2.58	21	258	400
MA-25	MA	8/19/2004	71					45	33	2.1		253	598
MA-26	MA	5/23/1994						48	46	1.89			650
MA-26	MA	8/21/2004						53	36	2.3			883
MA-27	MA	11/28/1993	72	28			308	40	44	1.59	25	296	500
MA-27	MA	3/2/1994	71.2	27.84			342	40	45	1.68	24	294	440
MA-27	MA	5/23/1994						48	44	2.19	24	450	450
MA-27	MA	7/1/1994	79	29			288	52.5	45	1.79	25	318	450
MA-27	MA	8/12/2004						52	35	1.7			672
MA-28	MA	5/23/1994						32	30	1.39			460
MA-28	MA	7/12/1994	54	18			217	25	23	2.09	22	210	400
MA-29	MA	5/23/1994						52.5	47	1.59			1300
MA-29	MA	8/16/2004						52	36	1.8			681
MA-30	MA	6/15/1993	71	30.6	45.6	6.77	338	38.5	53	2.38	25.775	313	
MA-30	MA	12/6/1993	78	24	38.1	6.3	305	36	43	1.89	25.7	283	550
MA-30	MA	2/26/1994	68.4	26.84			303	34	44	1.59	25.4	282	500
MA-30	MA	7/5/1994	71	26			298	50	43	1.89	25	285	500
MA-31	MA	5/27/1994						46	43	1.98			500
MA-31	MA	8/11/2004						52	38	2.2			672
MA-32	MA	5/23/1994						54.5	47	2.19			550
MA-32	MA	7/1/1994	83	28			312	52.5	47	1.89	24	322	550
MA-32	MA	8/1/2004						53	37	2.5			686
MA-33	MA	6/15/1993	76	28.2	35.8	6.59	339	44.75	47	1.89	27.025	315	
MA-33	MA	12/6/1993	75	22	37	5.9	306	33	37	1.89	25.1	285	420
MA-33	MA	2/26/1994	69.6	26.4			312	33.5	41	1.20	24	284	500
MA-33	MA	7/5/1994	70	26			332	45	40	1.49	24	282	450
MA-34	MA	5/27/1994						25.5	14	0.60			290
MA-34	MA	7/12/1994	62	11			189	27	16	1.48	14	200	300
MA-34	MA	8/12/2004	35					31	13	0.7		120	397
MA-35	MA	5/27/1994						75	56	2.68			
MA-36	MA	5/27/1994						34	26	0.78			
MA-39	MA	5/27/1994						51.5	43	2.09			
MA-39	MA	8/21/2004						56	38	2.3			547
MA-40	MA	5/25/1994						53.5	45	2.19			550
MA-40	MA	8/25/2004						49	37	2.5			690
MA-42	MA	5/25/1994						47	43	2.19			550

Appendix B: Analytical Results for Water Samples Collected in 1993-1994 and 2004 (see legend on p. 156)

Location ID	Area	Date	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	HCO3 (mg/L)	Cl (mg/L)	SO4 (mg/L)	NO3-N (mg/L)	Si (mg/L)	T Hard. (mg/L)	SC (us)
MA-42	MA	8/21/2004						52	37	1.8			649

Appendix B: Analytical Results for Water Samples Collected in 1993-1994 and 2004 (see legend on p. 156)

Location ID	Area	Date	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	HCO3 (mg/L)	Cl (mg/L)	SO4 (mg/L)	NO3-N (mg/L)	Si (mg/L)	T Hard. (mg/L)	SC (us)
MA-43	MA	5/25/1994						45.5	42	1.29			500
MA-44	MA	5/27/1994						86	43	1.29			440
MA-44	MA	8/12/2004	59					93	46	1.1		206	666
MA-45	MA	5/27/1994						56.5	48	1.59			
MA-46	MA	5/23/1994						142.5	70	1.69			650
MA-46	MA	8/12/2004	92					143	72	1.9		324	921
MA-47	MA	7/1/1994		30				52.5	46	1.79	24	302	500
MA-47	MA	9/22/2004	71				332	59	40	2.3			666
MA-48	MA	5/23/1994						53.5	48	1.89			650
MA-49	MA	5/27/1994						118	65	1.99			650
MA-50	MA	5/23/1994						110	69	1.29			
MA-51	MA	5/23/1994						50.5	50	1.89			
MA-51	MA	8/25/2004						49	38	2.2			698
MA-52	MA	5/27/1994						77.5	68	2.28			
MA-53	MA	5/23/1994						76.5	55	2.19			550
MA-54	MA	5/23/1994						48.5	47	1.59			
MA-54	MA	8/21/2004						52	36	1.8			643
MA-55	MA	5/27/1994						48.5	42	1.79			600
MA-55	MA	8/21/2004						49	37	2.0			650
MA-56	MA	5/23/1994						50	46	1.89			480
MA-56	MA	8/21/2004						48	36	1.9			645
MA-57	MA	5/27/1994						47.5	42	1.39			500
MA-59	MA	5/25/1994						234	180	1.79			
MA-59	MA	8/19/2004	108					151	134	1.7		408	1162
MA-62	MA	5/23/1994						77	54	2.28			580
MA-62	MA	8/16/2004	54					58	32	1.8		193	618
MA-63	MA	5/23/1994						69.5	58	2.58			450
MA-64	MA	5/23/1994						68	51	2.19			450
MA-65	MA	5/27/1994						46	46	1.79			
MA-65	MA	8/12/2004						52	36	1.8			508
MA-66	MA	5/25/1994						54	48	4.40			
MA-67	MA	5/25/1994						70	58	2.78			600
MA-67	MA	8/16/2004	63					74	48	2.5		240	773
MA-68	MA	5/23/1994						43.5	42	1.59			550
MA-68	MA	9/8/2004						48	38	2.1			649
MA-100	MA	8/16/2004	42					32	19	0.3		161	420
MA-101	MA	8/16/2004						49	36	2.4			670

Appendix B: Analytical Results for Water Samples Collected in 1993-1994 and 2004 (see legend on p. 156)

Location ID	Area	Date	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	HCO3 (mg/L)	Cl (mg/L)	SO4 (mg/L)	NO3-N (mg/L)	Si (mg/L)	T Hard. (mg/L)	SC (us)
MA-102	MA	8/21/2004						51	35	1.8			844

Appendix B: Analytical Results for Water Samples Collected in 1993-1994 and 2004 (see legend on p. 156)

Location ID	Area	Date	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	HCO ₃ (mg/L)	Cl (mg/L)	SO ₄ (mg/L)	NO ₃ -N (mg/L)	Si (mg/L)	T Hard. (mg/L)	SC (us)
MA-103	MA	8/21/2004						82	54	2.8			786
MA-104	MA	8/21/2004						63	40	2.0			676
MA-105	MA	9/8/2004						102	80	0.7			
MA-106	MA	8/31/2004						43	36	2.1			647
MA-107	MA	8/31/2004						58	38	2.8			723
MA-108	MA	9/22/2004						54	35	1.7			665
MA-109	MA	9/22/2004						39	28	1.9			571
MA-110	MA	9/22/2004						36	25	2.2			580
MA-111	MA	9/22/2004						75	25	1.9			611
MA-112	MA	9/27/2004						49	36	1.9			631
MA-113	MA	9/27/2004						50	36	1.9			625
MA-114	MA	9/27/2004						49	36	1.8			610
MA-115	MA	9/27/2004						35	20	0.8			402
PA-1	MA	5/29/1994						47	49	1.59			
PA-1	MA	7/12/1994	78	33			362	58.5	61	1.89	26	330	600
PA-1	MA	8/25/2004						46	37	2.2			
PA-10	MA	5/29/1994						42	39	1.39			
PA-10	MA	7/12/1994	76	27			275	44	38	1.79	23	300	450
PA-10	MA	8/25/2004						43	36	1.9			
PA-2	MA	5/29/1994						49.5	22	1.59			
PA-2	MA	7/12/1994	51	19			225	56	24	1.59	12	205	350
PA-2	MA	8/25/2004	66					62	22	2.0		220	
PA-3	MA	5/29/1994						52	48	2.19			
PA-3	MA	7/12/1994	77	32			317	55	48	2.28	22	325	580
PA-3	MA	8/25/2004						46	35	2.2			
PA-4	MA	5/29/1994						56.5	52	2.09			
PA-4	MA	5/29/1994						56.5	52	1.79			
PA-4	MA	5/29/1994						56.5	51	1.79			
PA-4	MA	7/12/1994	85	36			353	60.5	52	1.99	24	360	550
PA-4	MA	8/25/2004	81					58	38	2.3		307	
PA-5	MA	5/29/1994						44.5	45	2.28			
PA-5	MA	8/31/2004						43	36	2.0			643
PA-6	MA	5/29/1994						47	47	1.89			
PA-6	MA	5/29/1994						44	48	1.99			
PA-6	MA	7/12/1994	76	31			339	55.5	48	1.69	28	317	525
PA-6	MA	8/31/2004						42	35	1.9			639

Appendix B: Analytical Results for Water Samples Collected in 1993-1994 and 2004 (see legend on p. 156)

Location ID	Area	Date	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	HCO3 (mg/L)	Cl (mg/L)	SO4 (mg/L)	NO3-N (mg/L)	Si (mg/L)	T Hard. (mg/L)	SC (us)
PA-7	MA	5/29/1994						33	2	2.48			
PA-7	MA	7/12/1994	29	8			122	38.5	0	2.09	13	105	250
PA-7	MA	8/31/2004	98					54	30	2.3		247	537
PA-8	MA	5/29/1994						47.5	47	1.69			
PA-8	MA	5/29/1994						45.5	47	2.09			
PA-8	MA	5/29/1994						49	47	2.09			
PA-8	MA	8/31/2004						43	37	2.2			643
PA-9	MA	5/23/1994						39	35	0.99			460
PA-9	MA	5/29/1994						40	34	1.29			
PA-9	MA	7/12/1994	84	22			258	41	35	1.59	23	300	400
PA-9	MA	8/25/2004	76					43	36	1.8		285	
PMW-1	MA	5/27/1994						45.5	45	2.28			
PMW-1	MA	7/12/1994	75	31			322	55	48	1.59	26	316	550
PMW-1	MA	9/8/2004	66					43	33	2.5		250	570
PMW-3	MA	7/12/1994	209	76			386	185	160	5.16	24	835	
Well 02	MA	7/11/1993	81.6	37.2	51.7	6.13	376	61	75	4.17	24.65	375	890
Well 02	MA	3/2/1994	138.8	47.28			344	118	66	4.88	25	544	700
Well 02	MA	7/5/1994	70	44			367	75	75	2.38	24	356	750
Well 03	MA	7/11/1993	86.7	37.2	52.2	6.4	364	58.5	75	4.17	25.875	376	890
Well 03	MA	7/5/1994	78	41			373	92	90	2.78	24	364	750
Well 10	MA	7/5/1994	77	33			359	56	54	2.38	25	328	600
Well 12	MA	7/11/1993	76.5	33.6	46.9	5.2	368	53.75	48	2.48	26.125	329	550
Well 12	MA	7/5/1994	74	34			367	62	52	2.09	27	324	600
Well 13	MA	7/11/1993	81.6	36	38.3	5.52	342	50.5	53	2.38	25.9	337	470
Well 13	MA	7/5/1994	80	30			312	57	48	2.19	26	324	500
Well 14	MA	7/11/1993	68.7	32.4	36.8	5.27	351	45.25	43	2.98	25.5	315	470
Well 16	MA	7/11/1993	81.6	32.4	48	5.54	342	49.75	53	1.99	27.425	327	550
Well 16	MA	7/5/1994	69	35			359	56	57	2.09	27	316	600
Well 28	MA	6/29/1993	86.7	27.6	50.4	5.74	360	55.25	58	2.19	19.55	342	750
Well 28	MA	12/10/1993	69	36	49	8.4	339	58.5	64	2.57	21.9	323.7	750
Well 28	MA	7/5/1994	81	41			325	100	80	3.48	22	372	750
Well 30	MA	7/11/1993	71.4	27.6	38.3	4.33	312	40.75	48	2.58	24.7	295	455
Well 30	MA	12/10/1993	75	32	47	6.4	322	48.3	53	2.98	22.6	321.9	500
Well 30	MA	3/2/1994	83.6	28.08			320	53	59	2.89	24	326	520
Well 36	MA	12/6/1993	76.7	32	43.2	7.5	336	48.8	45	2.89	20	326.15	550
Well 36	MA	3/2/1994	81.2	29.52			330	44	52	2.39	25	326	390
Well 36	MA	5/23/1994						64.5	47	1.99		322	600

Appendix B: Analytical Results for Water Samples Collected in 1993-1994 and 2004 (see legend on p. 156)

Location ID	Area	Date	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	HCO3 (mg/L)	Cl (mg/L)	SO4 (mg/L)	NO3-N (mg/L)	Si (mg/L)	T Hard. (mg/L)	SC (us)
Well 36	MA	7/5/1994	83	28			310	56.5	50	2.09	28	328	600

Appendix B: Analytical Results for Water Samples Collected in 1993-1994 and 2004 (see legend on p. 156)

Location ID	Area	Date	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	HCO ₃ (mg/L)	Cl (mg/L)	SO ₄ (mg/L)	NO ₃ -N (mg/L)	SI (mg/L)	T Hard. (mg/L)	SC (us)
Well 372-001	MA	8/31/2004						55	40	2.1			700
Well 381-001	MA	8/31/2004						45	37	2.1			668
River-01	PR	6/4/1993	48	16.2			217	18	22	1.39	16.55	187.5	332
River-01	PR	6/21/1993	61.33333	20.8	24	5.03	253	27.75	27	0.60	21.125	216	375
River-01	PR	12/6/1993	96	39	48.4	8.3	364	36	45	1.59	26	330	500
River-01	PR	2/26/1994	70	36.72			351	47.75	53	1.20	27	328	480
River-01	PR	7/5/1994	63	30			317	53	44	1.49	27	280	500
River-01	PR	9/8/2004						62	40	1.20			
RC-01	RC	8/31/2004						10	14	0.9			305
RC-02	RC	8/31/2004						25	14	1.7			512
RC-03	RC	8/31/2004						28	25	1.4			485
SG-01	SG	7/20/1993	67.69	27.6	30	7.12	330	53	48	2.08	26.825	284.317	550
SG-01	SG	11/28/1993	103	26	40	5.9	316	49.5	42	1.89	23.6	290	500
SG-01	SG	3/2/1994	73.2	26.64			308	41	46	1.49	24	294	420
SG-01	SG	7/1/1994	72	28			317	48.5	46	1.59	25	294	500
SG-01	SG	8/1/2004	53					47	37	1.7		202	659
SG-02	SG	6/12/1993	80	29.6	48.2	7.86	477	39.5	44	2.28	27.6	340	580
SG-02	SG	11/28/1993	104	31	44.5	6.3	347	40	42	1.99	27.1	316	500
SG-02	SG	3/2/1994	78.8	28.08			330	47.6	44	1.59	27	314	450
SG-02	SG	7/1/1994	75	29			308	57	45	2.88	27	308	500
SG-02	SG	8/25/2004	78					44	34	1.9		293	533
SG-03	SG	7/20/1993	82.39	32.4	34.1	7.85	334	54.25	46	2.88	27.125	341.083	550
SG-03	SG	11/28/1993	139	32	49.7	6.6	337	40.5	41	2.38	26.8	302	500
SG-03	SG	3/2/1994	76.4	28.08			347	41.5	43	1.48	27	308	500
SG-03	SG	7/1/1994	74	28			273	52.5	43	1.89	26	299	450
SG-03	SG	8/1/2004	49					46	33	1.9		188	617
SH-01	SI	8/31/2004	11					73	66	2.4		392	898
SH-02	SI	8/31/2004	67					54	38	2.0		267	737
SH-03	SI	8/31/2004	71					53	37	1.7		288	720
MC-01	WB, MC	11/28/1993	69	21	16.8	2.3	211	64.5	30	1.89	22.6	275	400
MC-01	WB, MC	2/28/1994	80.8	21.12			292	45.8	40	2.57	23	290	450
MC-02	WB, MC	11/28/1993	75	17	33.5	1.4	265	64	31	3.28	18.25	258	400
MC-02	WB, MC	2/28/1994	70	19.44			260	74	36	2.78	19	256	450
MC-02	WB, MC	7/1/1994	70	19			273	80.5	35	3.08	19	254	500
MC-02	WB, MC	8/18/2004						87	31	3.5			712

Appendix B: Analytical Results for Water Samples Collected in 1993-1994 and 2004 (see legend on p. 156)

Location ID	Area	Date	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	HCO ₃ (mg/L)	Cl (mg/L)	SO ₄ (mg/L)	NO ₃ -N (mg/L)	SI (mg/L)	T Hard. (mg/L)	SC (us)
MC-03	WB, MC	6/4/1993	58	8.4			186	15.5	2	0.60	10.725	158	275
MC-03	WB, MC	11/28/1993	56	5	0	0.51	179	21.5	0	0.79	12.4	170	250
MC-03	WB, MC	2/28/1994	43.6	11.04			196	13	3	1.35	14	155	240
MC-03	WB, MC	7/1/1994	54	5			156	19	2	0.70	13	156	250
MC-04	WB, MC	8/19/2004						31	10	0.5			
MC-04	WB, MC	2/28/1994	90.4	35.04			317	105	53	4.27	26	372	700
MC-04	WB, MC	8/19/2004						127	47	4.7			
MC-05	WB, MC	7/26/1993	72.89	22.8	31.9	3.48	327	52	27	0.30	21.65	277.301	944
MC-05	WB, MC	11/28/1993	89	22	24.1	1.9	322	53.5	23	0.70	21.1	294	450
MC-05	WB, MC	2/28/1994	73.6	23.04			301	52.8	28	0.79	21	280	370
MC-05	WB, MC	7/1/1994	82	23			325	65	27	0.79	22	300	450
MC-05	WB, MC	8/19/2004						68	24	0.7			
MC-06	WB, MC	6/4/1993	55	12			217	15.5	17	1.39	12.925	173	310
MC-06	WB, MC	11/28/1993	75	19	9.6	2.9	275	27.5	25	1.49	20.5	252	300
MC-06	WB, MC	2/28/1994	61.2	17.52			249	49.5	27	0.79	19	226	320
MC-06	WB, MC	7/1/1994	65	21			250	43	30	0.99	20	248	350
MC-06	WB, MC	9/8/2004						38	24	1.4			
MC-07	WB, MC	2/28/1994	51.6	16.08			204	56.3	30	1.40	18	196	320
MC-07	WB, MC	7/12/1994	57	21			208	95	42	1.39	22	230	450
MC-07	WB, MC	8/19/2004						115	42	1.8			
MC-08	WB, MC	6/4/1993	53	11.4			216	16.4	3	0.79	11.225	182	325
MC-08	WB, MC	11/28/1993	53	8	5.1	0.97	210	24	0	1.09	13.3	171	250
MC-08	WB, MC	2/28/1994	44.4	11.52			223	14.5	3	1.99	13	159	220
MC-08	WB, MC	7/1/1994	45	15			217	24.5	11	0.70	16	174	250
MC-08	WB, MC	8/19/2004						40	11	0.9			
MC-100	WB, MC	8/19/2004						81	27	1.3			
MC-101	WB, MC	8/19/2004	89					92	42	1.1			
MC-102	WB, MC	8/19/2004	259					828	111	5.2		314	813
MC-103	WB, MC	8/19/2004						63	23	1.8		1084	2020
MC-104	WB, MC	8/19/2004						105	31	2.4			
MC-105	WB, MC	8/19/2004	108					140	48	5.9		418	665
MC-106	WB, MC	8/19/2004						88	12	2.0		1128	
MC-107	WB, MC	9/8/2004	81					38	26	1.3		270	531
MC-108	WB, MC	9/8/2004	79					93	30	1.5		275	657
MC-109	WB, MC	8/28/2004						217	105	1.8		1087	
MC-110	WB, MC	8/28/2004						25	1	1.1		356	
MC-111	WB, MC	8/27/2004						71	28	1.6		513	

Appendix B: Analytical Results for Water Samples Collected in 1993-1994 and 2004 (see legend on p. 155)

Location ID	Area	Date	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	HCO3 (mg/L)	Cl (mg/L)	SO4 (mg/L)	NO3-N (mg/L)	SI (mg/L)	T Hard. (mg/L)	SC (us)
MC-112	WB, MC	8/28/2004					133	47		1.5			855

Appendix B: Analytical Results for Water Samples Collected in 1993-1994 and 2004 (see legend on p. 156)

Location ID	Area	Date	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	HCO3 (mg/L)	Cl (mg/L)	SO4 (mg/L)	NO3-N (mg/L)	SI (mg/L)	T Hard. (mg/L)	SC (us)
MC-113	WB, MC	8/28/2004						82	33	3.3			686
MC-114	WB, MC	8/28/2004						109	38	3.5			784
MC-115	WB, MC	8/28/2004						176	80	6.8			1068
MC-116	WB, MC	9/8/2004	77					132	50	1.6		291	818
MC-117	WB, MC	8/28/2004						165	51	2.8			920
MC-118	WB, MC	8/28/2004						180	31	6.6			1239
MC-119	WB, MC	9/8/2004	154					263	95	2.6		554	1190
MC-120	WB, MC	8/28/2004						97	60	0.7			762
MC-121	WB, MC	8/28/2004						319	95	2.6			1339
MC-122	WB, MC	8/28/2004						118	36	1.8			720
MC-123	WB, MC	8/28/2004						170	80	2.4			948
MC-124	WB, MC	8/28/2004						67	34	3.0			717
MC-125	WB, MC	8/28/2004						280	102	10.7			1395
MC-126	WB, MC	8/28/2004						81	32	1.4			624
MC-127	WB, MC	8/27/2004						242	45	4.3			1154
MC-128	WB, MC	8/27/2004						22	1	1.0			354
MC-129	WB, MC	8/27/2004						65	24	2.3			634
MC-130	WB, MC	9/8/2004						39	22	1.1			530
MC-131	WB, MC	9/8/2004						22	14	1.3			389
MC-132	WB, MC	8/22/2004						47	25	0.8			481
MW-1	WB, JC	9/27/2004						119	54	111.0			
MW-2	WB, JC	10/1/2004						115	47	89.0			
MW-3	WB, JC	10/1/2004						124	45	57.0			
MW-4	WB, JC	10/1/2004						121	47	48.0			
MW-5	WB, JC	8/31/2004						280	140	2.1			1567
WB-01	WB, GJ	7/27/1993	41.35	5.4	7.2	1.57	138	7.5	0	0.79	14.875	125.893	160
WB-01	WB, GJ	12/9/1993	74	13	19.5	0.8	261	7.9	0	0.70	12.7	239.5	150
WB-01	WB, GJ	3/4/1994	58.6	11.28			254	5.25	0	0.60	14	196	320
WB-01	WB, GJ	7/8/1994	61	11			245	8.5	1	1.09	12	199	250
WB-01	WB, GJ	9/9/2004						12	2	1.1			386
WB-02	WB, GJ	7/27/1993	34.85	4.2	10.2	2	78	7.5	0	0.50	15.175	104.638	110
WB-02	WB, GJ	12/9/1993	13	3	19	0.8	61	5.7	0	0.99	7.8	45.1	100
WB-02	WB, GJ	3/4/1994	12.8	2.4			59	3.25	1	0.60	12	42	70
WB-02	WB, GJ	7/8/1994	13	4			74	7	1	1.39	13	48	100
WB-02	WB, GJ	9/9/2004						11	2	1.3			89.6
WB-03	WB, GJ	3/4/1994	32	4.8			115	6.1	0	0.70	12	100	180

Appendix B: Analytical Results for Water Samples Collected in 1993-1994 and 2004 (see legend on p. 156)

Location ID	Area	Date	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	HCO3 (mg/L)	Cl (mg/L)	SO4 (mg/L)	NO3-N (mg/L)	Si (mg/L)	T Hard. (mg/L)	SC (us)
WB-04	WB, GJ	12/9/1993	39	18	31	1.8	203	11.3	21	0.70	10.3	173.1	300
WB-04	WB, GJ	3/4/1994	40.4	17.52			205	13.5	23	0.70	14	174	310
WB-04	WB, GJ	7/8/1994	44	17			120	18.5	23	1.09	13	179	350
WB-04	WB, GJ	8/25/2004						21	20	1.1			361
WB-05	WB, JC	12/9/1993	20	5	18	0.5	70	6.9	0	1.20	10.1	71	150
WB-06	WB, JC	7/27/1993	68.74	20.4	42.7	3.15	283	73	36	0.60	30.225	256.918	450
WB-06	WB, JC	12/9/1993	77	24	53	2.4	273	69	38	0.60	20	283.3	550
WB-06	WB, JC	3/4/1994	71.2	21.6			254	67.5	36	0.60	25	268	650
WB-07	WB, GJ	2/28/1994	106	51.12			266	246	100	2.78	24	478	1550
WB-07	WB, GJ	7/1/1994	98	53			271	248	100	2.68	23	464	1400
WB-07	WB, GJ	8/12/2004						33	20	0.8			553
WB-08	WB, GJ	5/23/1994						25.5	17	1.19			300
WB-08	WB, GJ	8/12/2004						37	15	1.5			518
WB-09	WB, GJ	5/23/1994						45.5	27	3.08			350
WB-09	WB, GJ	8/12/2004	67					28	17	1.8		200	403
WB-10	WB, GJ	6/14/1993	36.8	6.72	19.5	2.03	144	16.2	12	0.89	13.375	129	206
WB-10	WB, GJ	7/27/1993	70.11	11.4	17.5	1.79		28	19	0.80	16.725	222.813	310
WB-10	WB, GJ	12/9/1993	63	8			149	14	0	0.60	16	192	250
WB-10	WB, GJ	3/4/1994	63.2	7.2			161	10.75	0	0.50	14	188	240
WB-10	WB, GJ	7/8/1994	67	8			156	14	1	1.49	17	201	300
WB-10	WB, GJ	9/9/2004						16	2	1.2			252
WB-11	WB, GJ	3/4/1994	56.8	12			210	33.5	23	0.79	16	192	250
WB-11	WB, GJ	7/12/1994	52	14			193	43	23	1.29	16	189	240
WB-11	WB, GJ	8/25/2004						78	28	1.3			564
WB-12	WB, GJ	5/23/1994						38.5	24	1.69			360
WB-12	WB, GJ	8/12/2004						37	17	1.7			434
WB-13	WB, JC	5/27/1994						51	43	1.69			
WB-14	WB, JC	6/14/1993	59	9.6	28.8	0.03	187	38.5	27	0.70	16.775	183	305
WB-14	WB, JC	7/27/1993	77.47	12.6	23.5	1.67		38.5	25	0.50	17.275	246	315
WB-14	WB, JC	12/9/1993	81	15			242	74	40	0.50	18	264	350
WB-14	WB, JC	3/4/1994	70.8	19.92			247	78.5	41	0.50	23	260	650
WB-14	WB, JC	9/2/2004					110	110	45	1.1			791
WB-15	WB, CC	7/27/1993	24.12	4.8	7.09	2.14	67	7.5	0	0.50	12.875	80.316	90
WB-15	WB, CC	12/9/1993	24.4	8.4	21.3	1.8	78	12	5	0.79	11.1	96.28	100
WB-15	WB, CC	3/4/1994	17.2	6.48			9	9	5	0.50	13	70	125
WB-15	WB, CC	7/8/1994	24	6			129	8.5	3	0.79	12	86	100
WB-15	WB, CC	9/10/2004					9	9	2	1.2			58.2

Appendix B: Analytical Results for Water Samples Collected in 1993-1994 and 2004 (see legend on p. 156)

Location ID	Area	Date	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	HCO3 (mg/L)	Cl (mg/L)	SO4 (mg/L)	NO3-N (mg/L)	Si (mg/L)	T Hard. (mg/L)	SC (us)
WB-16	WB, JC	5/27/1994						68.5	34	0.89			350
WB-16	WB, JC	8/21/2004						73	29	1.3			511
WB-17	WB, JC	5/27/1994						36.5	24	0.79			
WB-17	WB, JC	8/21/2004						86	40	1.5			554
WB-18	WB, JC	5/23/1994						52	72.5	4.37			900
WB-18	WB, JC	8/12/2004	66					78	38	4.4		233	835
WB-19	WB, JC	5/23/1994						20	15	0.89			210
WB-19	WB, JC	7/5/1994	37	12			135	19	15	1.09	13	146	250
WB-19	WB, JC	8/21/2004						30	12	1.4			262
WB-20	WB, JC	6/12/1993	29	9	26	2.19	123	18	16	0.89	16.525	119	225
WB-20	WB, JC	12/9/1993	36	12	25	1.4	128	23	13	1.29	14	140.4	250
WB-20	WB, JC	3/4/1994	28.4	10.32			127	19	9	1.08	18	114	40
WB-20	WB, JC	7/8/1994	32	11			128	23.5	16	1.39	15	126	150
WB-20	WB, JC	9/8/2004						28	17	1.6			277
WB-21	WB, JC	6/12/1993	33.5	9	24.1	2.24	135	23.5	16	0.99	17.525	127	238
WB-21	WB, JC	12/9/1993	40	12	26.1	1.1	137	24.5	13	1.29	13.7	150.4	250
WB-21	WB, JC	3/4/1994	33.6	9.6			142	25.5	12	1.09	17	124	30
WB-21	WB, JC	7/8/1994	40	11			135	28	16	1.79	14	145	180
WB-21	WB, JC	9/8/2004						31	17	1.4			294
WB-22	WB, CK	6/4/1993	15.5	4.2			85	4.7	1	0.70	13.95	60	118
WB-23	WB, CK	6/4/1993	13	7.5			76	5	1	0.79	9.775	61.2	115
WB-23	WB, CK	11/28/1993	15	8.3	16.2	1.8	61	4.3	0	1.29	8.9	72.36	150
WB-23	WB, CK	3/4/1994	19.2	7.2			78	5.4	0	0.89	12	78	140
WB-23	WB, CK	7/8/1994	12	8			68	6.5	9	1.49	11	62	150
WB-23	WB, CK	9/8/2004						11	1	1.4			130
WB-24	WB, CK	6/4/1993	9.5	4.2			43	5	2	0.60	9.7	35.2	70
WB-24	WB, CK	3/4/1994	21.2	1.68			78	10.8	1	0.70	14	60	130
WB-25	WB, CC	3/4/1994	43.6	13.68			166	39.5	25	0.70	20	168	350
WB-26	WB, CC	6/14/1993	15.5	6.3	26	0.85	84	11.5	15	0.50	13.7	74	123
WB-26	WB, CC	7/27/1993	30.25	7	0.45	1.84		15.5	12	0.50	13.85	104.815	130
WB-26	WB, CC	12/9/1993	26	5			88	17.5	13	1.09	15	86	150
WB-26	WB, CC	3/4/1994	20	7.2				13.5	16	0.70	15	80	200
WB-26	WB, CC	7/8/1994	22	8			123	14	15	0.79	16	88	150
WB-26	WB, CC	9/10/2004						19	12	1.4			182
WB-27	WB, JC	5/25/1994						32.5	25	0.60			320
WB-27	WB, CC	8/11/2004	42				43	25	0.4			160	453
WB-100	WB, JC	8/12/2004					53	20	0.5				

Appendix B: Analytical Results for Water Samples Collected in 1993-1994 and 2004 (see legend on p. 156)

Location ID	Area	Date	Ca (mg/L)	Mg (mg/L)	Na (mg/L)	K (mg/L)	HCO ₃ (mg/L)	Cl (mg/L)	SO ₄ (mg/L)	NO ₃ -N (mg/L)	Si (mg/L)	T Hard. (mg/L)	SC (us)
WB-101	WB, GJ	8/30/2004	204					345	111	2.1		627	1417
WB-102	WB, GJ	8/30/2004	388					897	262	4.3		1528	3300
WB-103	WB, GJ	8/30/2004	75					304	127	1.8		289	1555
WB-104	WB, GJ	8/30/2004						30	30	1.2			424
WB-105	WB, JC	9/2/2004						15	11	1.1			350
WB-106	WB, JC	9/2/2004						63	27	1.1			591
WB-107	WB, JC	9/9/2004						31	16	1.5			293

Note: Shaded location IDs represent samples collected in 2004

Abbreviations

MA : Main Aquifer
EA : East Aquifer
PR : Portneuf River
RC : Rapid Creek
SG : South Gap
SI : South of Inkorn
WB, MC : West Bench, Mink Creek drainage
WB, JC : West Bench, Johnny Creek drainage
WB, GJ : West Bench, Gibson Jack Creek drainage
WB, CK : West Bench, Cusick Creek drainage
WB, CK : West Bench, City Creek drainage
SC : Specific Conductance
T Hard. : Total Hardness as CaCO₃
HCO₃ : Alkalinity as bicarbonate

**Appendix C: Updated Southern Pocatello Aquifer Water Balance, 1994
through 2003**

A detailed water balance of the southern Pocatello aquifer was completed in 1993-94 by Welhan et. al. (1996) (see section 1.5.3). This appendix includes data and calculations used to update the water balance for a period from June 1994 through October 2003.

The water balance in Welhan et al. (1996) covered a 510 day period in 1993 and 1994 (grey area in Figure C-1). This period spanned above- and below-normal water years so that the normalized average was considered to be a good approximation of the water balance for average conditions.

Table C-1 shows the data and calculations used to solve for the water balance equation:

$$Q_{\text{residual}} = \Delta S - Q_{\text{gap}} + Q_{\text{out}} + Q_{\text{pumping}},$$

where ΔS is the change in aquifer storage, Q_{gap} is underflow through the Portneuf Gap, Q_{out} is underflow out of the aquifer at cross section C-C near Red Hill, and Q_{pumping} represents pumping withdrawals from the southern aquifer. Welhan et al. (1996) assumed that 90% of Q_{residual} was Bannock Range recharge and that the Portneuf River was a negligible source of recharge.

Figure C-1 shows static water levels for four wells in the southern Pocatello aquifer through October 2003. In the first period shown, June 1994 - November 1997, static water levels rose about 7.1 feet, in the second period, November 1997 - October 2003, water levels declined an average of 21.2 feet. The third period, from June 1994 - October 2003, saw a net decrease in water levels of about 14 feet. These static water levels were used to calculate changes in aquifer storage (ΔS) for each time period.

In recalculating the underflow through the Portneuf Gap and outflow from the aquifer, it was assumed that the hydraulic gradients and transmissivity estimates from the original water balance were the same. The areas of saturated flow for each time period were adjusted based on the difference in the average change in static water levels for each period and the water level in June of 1994, which is when water levels for the original water balance were at their midpoint.

Pumping withdrawals were calculated for all of the municipal wells in the southern aquifer. Pumping withdrawals from domestic wells were also estimated for the 120 residences that overlie the aquifer; however, their combined pumpage is only about 10 million gallons per year, which is insignificant compared to the 1.5 billion gallons a year withdrawn by municipal wells (Table C-1).

Results were normalized to 365 days to compare to the water balance of Welhan et. al. (1996)(Table C-1). These normalized annual results show that for each of the three time periods, underflow through the Portneuf Gap did not vary considerably, ranging from about 1 to 1.1 bgy. Similarly, underflow from the southern aquifer (Qout) varied less than 15%, ranging from 5 to 5.7 bgy. Pumping withdrawals were also steady for all three time periods at about 1.5 bgy.

Since pumping withdrawals, underflow through the Portneuf Gap, and underflow out of the aquifer showed little variation, the model results indicate that larger fluctuations in Qresidual are required to account for the observed changes in aquifer storage. The normalized Qresidual values ranged from 5 to 6.5 bgy.

Welhan et al. (1996) assumed that the recharge from the Bannock Range comprised of more than 90% of the Qresidual component of the water balance. Figure C-

2 shows snowpack in the Wildhorse Divide area of the Bannock Range in relation to the three periods for the updated water balance calculations. Comparing Figures C-1 and C-2 shows that there is a strong correlation between static water levels and snowpack, providing further evidence that recharge from the Bannock Range is a the major component of recharge to the southern aquifer. Figures C-1 and C-2 also show that pumping withdrawals are in a delicate balance with aquifer recharge potential and are currently depleting the aquifer due to drought conditions over the past seven years.

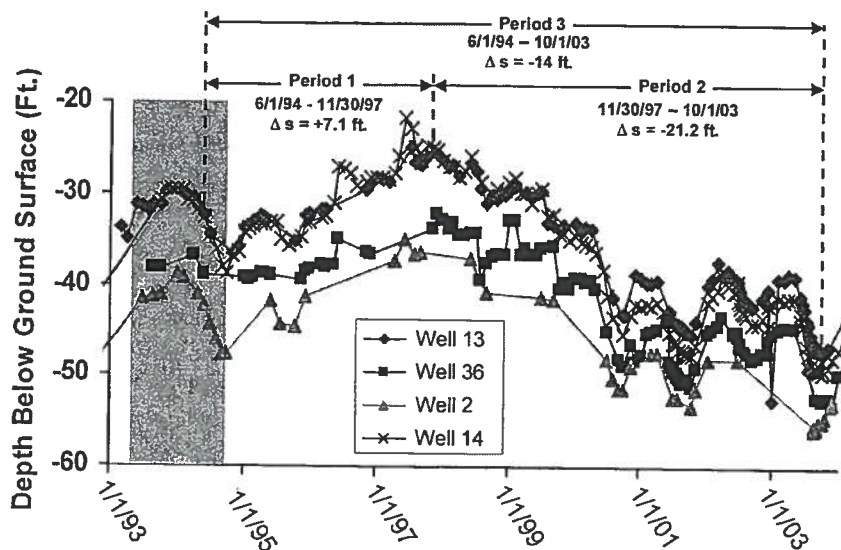


Figure C-1: Static water levels for four wells in the southern Pocatello aquifer for the three time periods in the updated water balance. The grey area represents the 510 day period covered in the water balance in Welhan et. al. (1996).

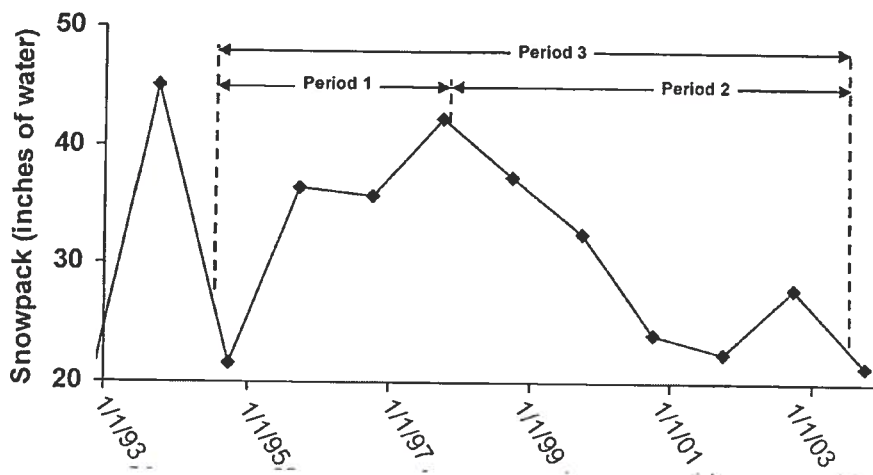


Figure C-2: Snowpack in the Wildhorse Divide area of the Bannock Range in relation to the three periods in the updated water balance calculations. This figure shows that there is a strong correlation between snowpack and static water levels in Figure C-1, providing further evidence that recharge from the Bannock Range is a significant component of recharge to the southern aquifer.

Table C-1: Data and calculations used for the southern Pocatello aquifer water balance. The water balance was updated for three periods from June 1994 through October 2003. Results were normalized to 365 days to compare to the water balance in Welhan et. al. (1996) completed in 1993-94 (bg = billion gallons, bgy = billion gallons per year, SWL = static water level, i = hydraulic gradient, K = hydraulic conductivity).

Water Balance Equation

$$Q_{\text{residual}} = \Delta S - Q_{\text{gap}} + Q_{\text{out}} + Q_{\text{pumping}}$$

Time Periods

	Total Time
1993 - 1994 =	365 days
Period 1 6/94 to 11/97 =	1278 days
Period 2 11/97 to 10/03 =	2130 days
Period 3 6/94 - 10/03 =	3408 days

Change in Aquifer Storage (ΔS)

Aquifer Area = 8.80E+07 ft²
 Porosity = 0.3

	Δ SWL (feet)	ΔS (ft ³)	ΔS (bg)	Normalized Annual ΔS (bgy)
1993-94			-0.34	-0.34
Period 1	7.1	1.87E+08	1.40	0.40
Period 2	-21.2	-5.6E+08	-4.19	-0.72
Period 3	-14	-3.7E+08	-2.76	-0.30

Portneuf Gap Underflow (Q_{gap})

Area Calculations - as function of a triangle (.5 base x height)

	Base (ft.)	Height (ft.)	Area (ft ²)
1993-94	600	100	30,000
Period 1	600	103.6	31,065
Period 2	600	96.6	28,980
Period 3	600	93	27,900

Flux Calculation

	K (ft/day)	i	area (ft ²)	Q (ft ³ /day)	Total Q (bg)	Normalized Annual Total Q (bgy)
1993-94	10,000	0.0013	30,000	390,000	1.06	1.06
Period 1	10,000	0.0013	31,065	403,845	3.86	1.10
Period 2	10,000	0.0013	28,980	376,740	6.00	1.03
Period 3	10,000	0.0013	27,900	362,700	9.25	0.99

Table C-1 (continued): Water Balance Calculations for the Southern Pocatello Aquifer

Underflow Out of Aquifer (Qout)

Area Calculations for xsection C-C

	Average Δ Height from 1994(ft)	Aquifer Width (ft)	Δ Area	Area (Ft2)
1993-94	0	3,100	0	250,000
Period 1	3.55	3,100	11,005	261,005
Period 2	-3.45	3,100	-10,695	239,305
Period 3	-7	3,100	-21,700	228,300

	K (ft/day)	i	area (ft2)	Q (ft3/day)	Total Q (bg)	Normalized Annual Total Q (bgy)
1993-94	8,000	0.001	250,000	2,000,000	5.46	5.46
Period 1	8,000	0.001	261,005	2,088,040	19.96	5.70
Period 2	8,000	0.001	239,305	1,914,440	30.50	5.23
Period 3	8,000	0.001	228,300	1,826,400	46.56	4.99

Pumping Withdrawal (Qpumping)

Municipal Withdrawals

	(bg)
1994	1.51
1995	1.3
1996	1.56
1997	1.47
1998	1.06
1999	1.32
2000	1.72
2001	1.93
2002	1.44
2003	1.52

	bg	Normalized Annual Qpumping (bgy)
1993-94	1.26	1.26
Period 1	5.30	1.51
Period 2	8.77	1.50
Period 3	14.10	1.51

Domestic Withdrawals

120 Households=

$$(300 \text{ gpd} \times 7 \text{ months}) + (150 \text{ gpd} \times 5 \text{ months}) = .01 \text{ bgy}$$

Table C-1 (continued): Water Balance Calculations for the Southern Pocatello Aquifer

Water Balance Results

$$Q_{\text{residual}} = \Delta S - Q_{\text{gap}} + Q_{\text{out}} + Q_{\text{pumping}}$$

	ΔS (bg)	Q_{gap} (bg)	Q_{out} (bg)	Q_{pumping} (bg)	Q_{residual} (bg)	Normalized Annual Q_{residual} (bg/y)
1993-94	-0.34	1.06	5.46	1.26	5.32	5.32
Period 1	1.40	3.86	19.96	5.30	22.80	6.51
Period 2	-4.17	6.00	30.50	8.77	29.10	4.99
Period 3	-2.76	9.25	46.56	14.10	48.65	5.21

**Appendix E: Autocorrelation of Mink Creek and Southern Aquifer
Groundwater Chemistry**

Figure E-1: Southern Aquifer Groundwater: Chloride

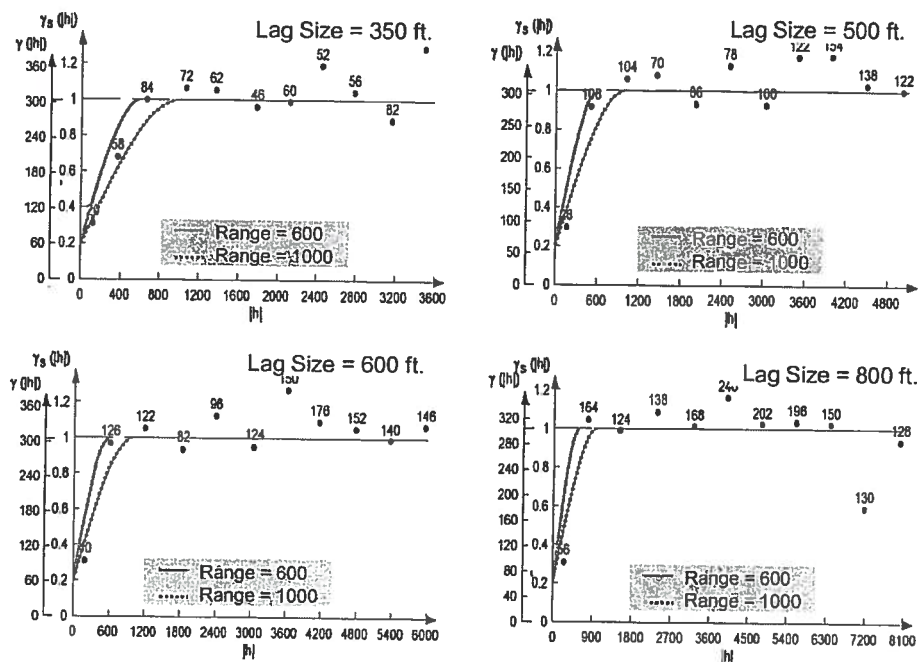


Figure E-1: Variograms displaying the spatial autocorrelation of chloride concentrations in the southern aquifer in 2004. Variograms statistics are displayed using the standardized semivariogram estimator at lag spacings of 350, 500, 600, and 800 feet. Semivariogram, covariance, and correlogram statistics were also calculated to ensure the autocorrelation structure was adequately inferred. The semivariogram scale is shown for comparison. A spherical model with a nugget of 0.2 was used to characterize the autocorrelation structure, with range values between 600 and 1000 feet.

Figure E-2: Mink Creek Groundwater: Chloride

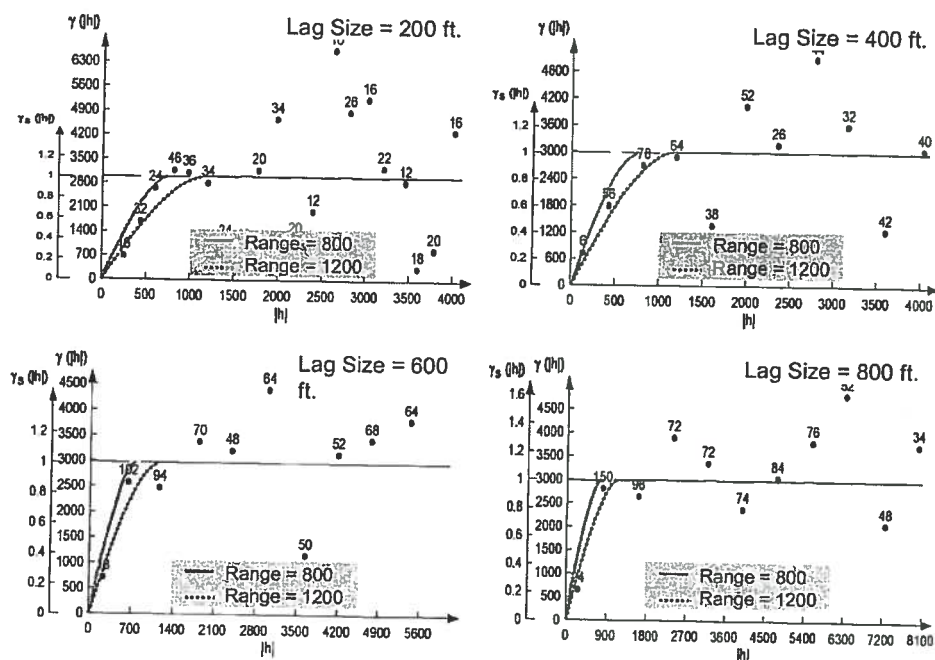


Figure E-2: Variograms displaying the spatial autocorrelation of chloride concentrations in Mink Creek groundwater in 2004. Variograms statistics are displayed using the semivariogram estimator at lag spacings of 200, 400, 600, and 800 feet. Standardized variogram, covariance, and correlogram statistics were also calculated to ensure the autocorrelation structure was adequately inferred. The standardized variogram scale is shown for comparison. A spherical model with a nugget of 0 was used to characterize the autocorrelation structure, with range values between 800 and 1200 feet.

Figure E-3: Southern Aquifer Groundwater: Sulfate

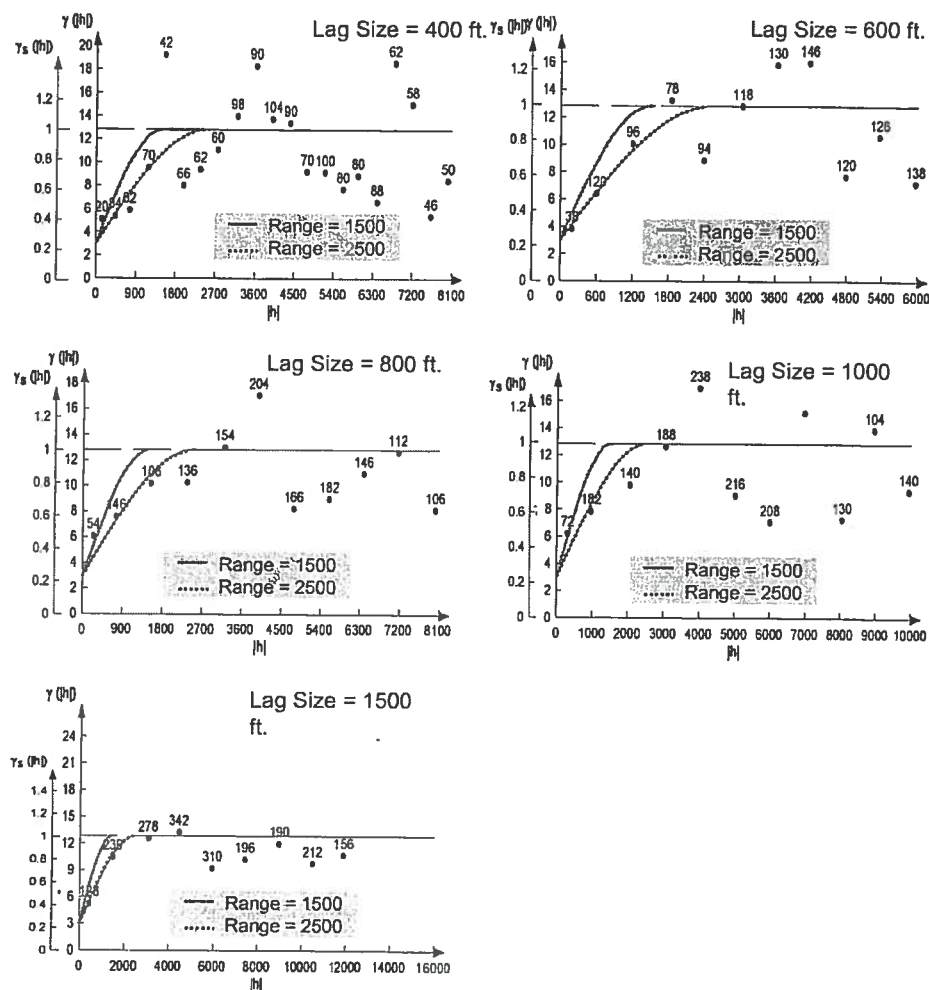


Figure E-3: Variograms displaying the spatial autocorrelation of sulfate concentrations in the southern aquifer in 2004. Variograms statistics are displayed using the semivariogram estimator at lag spacings of 400, 600, 800, 1000, and 1500 feet. Standardized variogram, covariance, and correlogram statistics were also calculated to ensure the autocorrelation structure was adequately inferred. The standardized variogram scale is shown for comparison. A spherical model with a nugget of 3 was used to characterize the autocorrelation structure, with range values between 1500 and 2500 feet.

Figure E-4: Mink Creek Groundwater: Sulfate

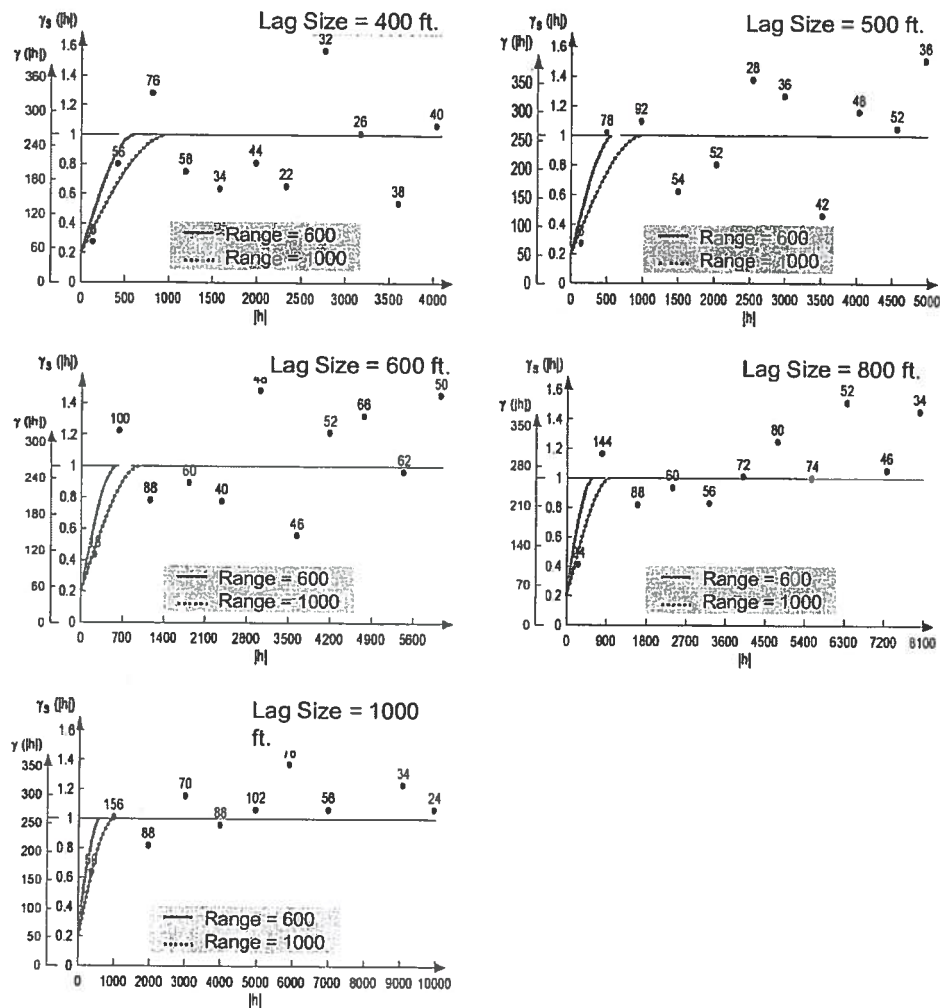


Figure E-4: Variograms displaying the spatial autocorrelation of sulfate concentrations in Mink Creek groundwater in 2004. Variograms statistics are displayed using the standardized semivariogram estimator at lag spacings of 400, 500, 600, 800, and 1000 feet. Semivariogram, covariance, and correlogram statistics were also calculated to ensure the autocorrelation structure was adequately inferred. The semivariogram scale is shown for comparison. A spherical model with a nugget of 0.2 was used to characterize the autocorrelation structure, with range values between 600 and 1000 feet.

Figure E-5: Southern Aquifer Groundwater: Nitrate

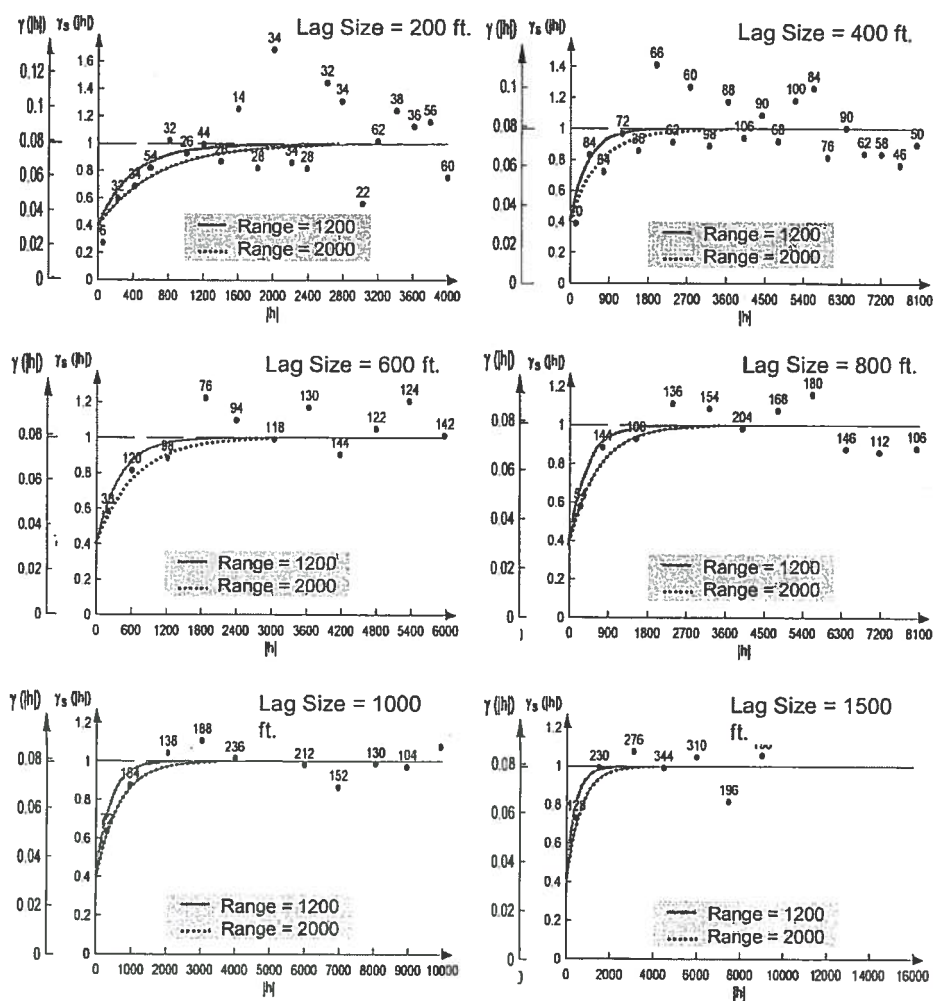


Figure E-5: Variograms displaying the spatial autocorrelation of nitrate concentrations in the southern aquifer in 2004. Variograms statistics are displayed using the standardized semivariogram estimator at lag spacings of 200, 400, 600, 800, 1000, and 1500 feet. Semivariogram, covariance, and correlogram statistics were also calculated to ensure the autocorrelation structure was adequately inferred. The semivariogram scale is shown for comparison. An exponential model with a nugget of 0.4 was used to characterize the autocorrelation structure, with range values between 1200 and 2000 feet.

Figure E-6: Mink Creek Groundwater: Nitrate

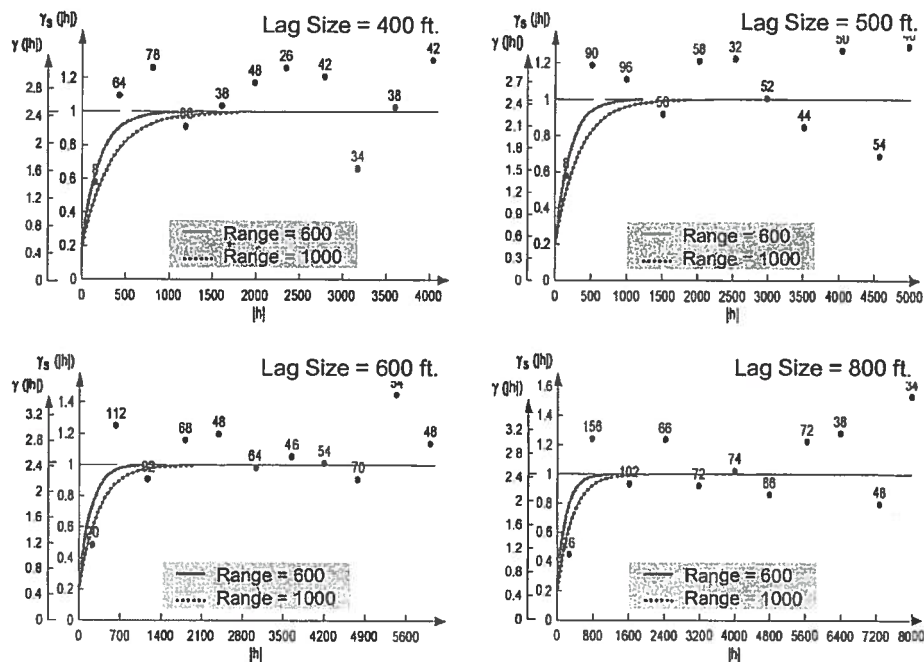


Figure E-6: Variograms displaying the spatial autocorrelation of nitrate concentrations in Mink Creek groundwater in 2004. Variograms statistics are displayed using the standardized semivariogram estimator at lag spacings of 400, 500, 600, and 800 feet. Semivariogram, covariance, and correlogram statistics were also calculated to ensure the autocorrelation structure was adequately inferred. The semivariogram scale is shown for comparison. An exponential model with a nugget of 0.2 was used to characterize the autocorrelation structure, with range values between 600 and 1000 feet.

Appendix F: Potential Sources of Salt Contamination to Groundwater

This appendix presents an overview of potential sources of inorganic salts and nutrients impacting groundwater. Emphasis is placed on chloride, sulfate, and nitrate chemistry. As presented in Chapters 3 and 4, these chemical constituents have strong linear relationships that indicate impacts from a single source. Due to their conservative nature they are also good tracers for evaluating mixing relationships between natural recharge and contaminant fluxes to groundwater.

Chapter 5, Section 5.3.2 discusses the most likely of these sources that may be responsible for salt and nitrate contamination.

F.1 Landfill Leachate

Precipitation falling on landfills tends to infiltrate readily due to the relatively porous nature of solid waste. Infiltrating water will mix with landfill fluids and solid waste resulting in contaminated leachate. Leachate from municipal landfills is typically enriched in all inorganic salts including nitrate (Panno et. al., 2002). The chemical composition of landfill leachate can vary greatly from site to site, and within a particular site. Table F-1 (from Lee and Jones, 1991) presents a compilation of chemical concentrations of several constituents in leachate from 83 municipal landfills in the U.S.

Table F-1: Concentration ranges for several constituents in leachate from 83 municipal landfills (from Lee and Jones, 1991).

Parameter	"Typical" Concentration	
	Range (mg/l)	Average (mg/l)
Chloride	100 – 2,000	980
Sulfate	10 – 1,000	380
Nitrate-N	0.1 - 10	4
Calcium	100 – 3,000	1,000
Sodium	200 – 1,500	700

F.2 Chemical Fertilizers

Chemical fertilizers are commonly used in agricultural practices and maintaining turfgrass in residential areas. Salts and nutrients can infiltrate beyond the root zone when fertilizers are over-applied or crops are over-irrigated (Minnesota Pollution Control Agency, 2000). Chloride, sulfate, and nitrate are commonly found in fertilizers and amendments applied to crops and turfgrass (Barton and Colmer, 2004; NDSU Extension Service, 2004; Panno et. al., 2002).

Chloride concentrations in triple phosphate fertilizers can be as high as 6.18% by weight (Panno et. al., 2002). Potash (KCl) is also commonly used to amend crop fields. The application of KCl for Potato crops in Idaho can be as high as 600 pounds per acre per year (Stark et al. 2004).

To correct or prevent sulfur deficiencies, amendments added to soils include ammonium sulfate, elemental sulfur, calcium sulfate, polysulfides and thiosulfates (NDSU Extension Service, 2004). Potassium sulfate is also commonly used for potato crops grown here in Idaho (NDSU Extension Service, 2004).

Nitrogen is the most common nutrient found in fertilizers and is sufficiently mobile in the subsurface when in the form of NO_3 (ITRC, 2000). Typical annual nitrogen application rates for turfgrass range from 100 to 300 pounds per acre (Barton and Colmer, 2004). Nitrogen losses tend to be low (< 5% of applied fertilizer) from established turfgrass that is not over-irrigated (Barton and Colmer, 2004). Nitrogen losses to groundwater under irrigated agriculture range from 70 to 150 pounds per acre per year, depending on cropping practices and climate (Minnesota Pollution Control Agency, 2000).

F.3 Animal Waste

Dairies, poultry farms, feedlots, stockyards and other facilities for raising and holding animals can produce significant volumes of animal manure enriched in nutrients and inorganic salts. Table 5-2 displays concentrations of salts and nutrients in lagoon waste water from swine and cattle feedlots (Ham et. al., 1999; Libra et. al., 1998) and from cattle manure leachate (Gellenbeck, D.J., 1994). This table shows that much of the nitrogen in animal waste is in the form of ammonia (NH_4). Ammonia may be oxidized to nitrate in a process called nitrification. This oxidation process is described in more detail in the discussion of septic tank effluent in section F.7.

Table F-2: Concentrations of salts and nutrients in lagoon waste water from swine and cattle feedlots (Ham et. al., 1999; Libra et. al., 1998) and from cattle manure leachate (Gellenbeck, D.J., 1994).

Parameter	Cattle Manure Leachate ^(a) (mg/l)	Lagoon Wastewater (mg/l)		
		Swine ^(b)	Swine ^(c)	Cattle ^(b)
Calcium	130	80	—	145
Sodium	110	270	—	148
Chloride	460	276	490-1,100	569
Sulfate	130	—	5-190	37
Nitrate-N	71	1	0.1 – 1.7	0.5
Ammonia-N	66	673	1,600-4,400	98

(a) Gellenbeck, D.J., 1994

(b) Ham et. al., 1999

(c) Libra et. al., 1998

F.4 Road Salt

Bannock County, the City of Pocatello, and the Idaho State Transportation Department mix sodium chloride (NaCl) and calcium chloride (CaCl) into their sand stockpiles used for anti-skid and road-deicing. Sand mixtures typically contain about five

percent NaCl and two to four percent CaCl (City of Pocatello, 1993; Bannock County, 1993; IDTP, 1993).

Soil analyses from two sanding stockpiles at the Bannock County maintenance facility revealed that the molar ratio of Cl to SO₄ in the coarse fractions was about 1100:1 and in the fine fractions was about 1700:1 (CH2MHill, 1994(a)).

During this thesis research, samples were collected from road salt stock piles at the City of Pocatello maintenance facility. These samples were dissolved into deionized water for a period of forty eight hours. The resulting waters were analyzed for Cl, SO₄ and NO₃. Molar ratios of chloride to sulfate in these samples ranged from 41:1 to 126:1. Nitrate in all samples was less than 0.8 mg/l.

These results indicate that deicing salts could potentially provide large amounts of leachable Cl to groundwater, while sulfate and nitrate additions would be insignificant.

F.5 Road Aggregate

Groundwater samples collected in 1993 revealed that a stockpile of road aggregate was leaching sulfate to groundwater in the southern aquifer (see section 3.2). A detailed description on the investigation and potential implications on the fate and transport of leachable sulfate in road building materials can be found in Meehan and Welhan (1994), which has been attached as Appendix D of this thesis.

Leach tests performed on the road aggregate revealed that SO₄, Ca, and K were the major soluble components. Geochemical modeling indicated that ion-exchange of sodium with Ca and Mg in clays could explain the deficit of sodium concentrations compared to Cl (see Section 3.2), and the observed coupled increases of Mg with other salts.

Meehan and Welhan (1994) hypothesized that leachable sulfate from road aggregate and chloride from road salt may be responsible for the coupled trends of Cl and SO₄ observed in local groundwater. However, it is not known to what extent road aggregate of this type (containing leachable sulfate) has been used in the LPRV.

F.6 Water Softeners

Water softeners remove hardness (calcium and magnesium) from potable water via an ion exchange resin. The resin beads are periodically regenerated through immersion in a sodium chloride (NaCl) brine, which is discharged to a sewer or septic tank (SCVJSS, 2002).

The average volume of brine waste discharged during each regeneration cycle is typically 47 gallons with Cl concentrations from 7,000 to 13,000 mg/l (SCVJSS, 2002). A family of three or four using a water softening system (per manufacturer's recommended use) can discharge more than 390 kg/year of Cl in brine waste (Panno et. al., 2002). In Santa Clarita Valley, California, water softeners were responsible for 70% of the Cl contributions in residential waste water (28% was from laundry soaps and human waste) (SCVJSS, 2002).

F.7 Natural Sources

All inorganic salts and nutrients found in ground water can be derived from naturally occurring minerals in rocks and soils. This section discusses the natural sources typically associated with high concentrations of Cl, SO₄, and NO₃ in ground water.

F.7.1 Natural Chloride Sources

High levels of Cl in groundwater are typically associated with sedimentary rocks, particularly evaporites. Chloride salts are generally in the form of NaCl (halite)

with lesser amounts of KCl and CaCl (Hem, 1985).

Chloride can also be incorporated in sedimentary and metasedimentary rocks formed in a sea or enclosed basin as a result of inclusion of connate brine and in the cementing material (Hem, 1985). Porous rocks submerged by seawater may also become impregnated with soluble chloride salts. Some of this chloride may persist in fine-grained marine shale for long periods of time (Hem, 1985). Billings and Williams (1967) computed an average Cl content for deeply buried shales as 1,466 ppm.

F.7.2 Natural Sulfate Sources

Natural sources of sulfate can occur in sedimentary environments in the form of gypsum ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$) and anhydrite (CaSO_4). Evaporite deposits of these minerals can produce high concentrations ($> 10,000 \text{ mg/l}$) of sulfate in ground water (Hem, 1985).

Sulfate is also produced from the oxidation of pyrite and other sulfide minerals in sedimentary and igneous rocks. When sulfide minerals are in contact with oxygenated water, sulfur is oxidized to sulfate ions. Hydrogen ions are also produced in this process, which decreases the pH of the waters and can increase the solubility of other metals and salts. Pyrite is common in sedimentary deposits formed under strong reducing conditions, like shale and coal (Hem, 1985).

F.7.3 Natural Nitrate Sources

Naturally occurring concentrations of NO_3 in groundwater are typically less than 3 mg/l, resulting mostly from atmospheric additions and decaying organic material in soils. However, some geologic formations can contribute large amounts of nitrate to groundwater from various nitrate-containing minerals (Hem, 1985).

The most common nitrate minerals are very soluble and rarely preserved, only

forming as evaporites in very arid environments. Soda niter (NaNO_3) and niter (KNO_3) are the most common nitrate minerals, (ISWS, 2004).

Nitrogen can be present in large quantities in sedimentary rocks (mostly in shale) in the form of ammonia (NH_4^+) and organic-N derived from the decay of buried organic material. If rocks containing significant reduced nitrogen are exposed to oxygen, oxidation may release NO_3 . For example, native fixed ammonium in loess deposits in southwestern and central Nebraska is responsible for soil nitrate concentrations as high as 87 mg/kg (Boyce et. al., 1976).

Ammonium oxidation in glacial tills derived from shales in Alberta, Canada have produced NO_3 concentrations of 100 to 300 mg/l in groundwater (Hendry et. al., 1984). Leachate from exchangeable ammonium in a shale spoils pile has led to groundwater NO_3 concentrations in excess of 500 mg/l (Zielinski et. al, 2001), and ammonium oxidation in shales and phyllites in the Sierra Nevada Range of California have impaired surface water quality with NO_3 concentrations up to 37 mg/l (Holloway et. al., 1998).

In New Mexico, two limestone formations have been identified with naturally occurring groundwater nitrate concentrations between 12 and 15 mg/l (Titus, 1980). High nitrate concentrations (60 to 90 mg/l) in ground water have also been identified in several undeveloped basins in Arizona and New Mexico with unknown geologic sources (Robertson, F.N., 1991).

F.8 Sewer and Septic Tank Effluent

Domestic waste water contains relatively high concentrations of organic compounds (0.2 to 0.6 g/l) containing reduced N and C (Wilhelm et. al., 1994; Wilhelm et. al., 1996). Organic nitrogen is usually found at concentrations from 20 to 70 mg/l, but

can be greater than 100 mg/l (Robertson, W.D. et al., 1991). Lesser amounts of organic S are also present in a molar ratio of approximately 1 S:35 N (Wilhelm et. al., 1994). Levels of S and SO_4 can vary widely in domestic waste due to additions from detergents, solvents, and other household products. Chloride in domestic waste water is primarily from urea, but also from household products and water softeners (see section F.6) (Wilhelm et. al., 1994; Wilhelm et. al., 1996).

Domestic sewerage waste is typically treated at a municipal waste water facility and should not impair groundwater unless a sewer line breaks. In contrast, all domestic waste distributed to septic systems is discharged in the sub-surface and can impact groundwater quality. An average household discharges between 150 to 300 gallons of waste water per day to a septic system (Bauman and Schafer, 1985; BBC Research and Consulting, 2001; Hantzsche and Finnemore, 1992; Panno, 2002; Thomas et. al., 1999; Wilhelm et. al., 1994).

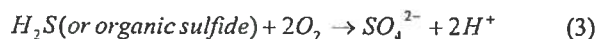
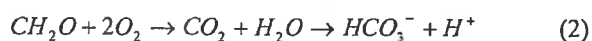
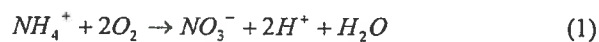
The evolution of domestic waste water in a septic system is driven by microbially catalyzed redox reactions in as many as three different redox zones. The septic tank, the first redox zone, is an anaerobic setting in which microorganisms use electron acceptors (C , H^+) to oxidize organic species of C, N, and S to produce CO_2 , CH_4 (methane), H_2S (hydrogen sulfide), and ammonia (NH_4). Most large particles are removed in the septic tank and the total N content of waste water may be reduced ten to thirty percent due to storage in sludge (Wilhelm et. al., 1994; Wilhelm et. al., 1996).

The second redox zone occurs in aerobic, unsaturated sediments of the drain field. In this zone, microorganisms use O_2 as the electron acceptor in the oxidation of organic C to CO_2 and NH_4 to NO_3 . In properly functioning drain fields, almost complete oxidation

of reduced waste water components occurs because adequate O₂ diffuses into the unsaturated zone (Wilhelm et. al., 1994; Wilhelm et. al., 1996). Complete NH₄ oxidation can occur within a few tens of centimeters of the distribution drain field pipes and within a few hours of exposure to O₂ (Robertson and Blowes, 1995).

Another 20% of inorganic N may be lost from effluent in the vadose zone, possibly due to denitrification on microsites and adsorption of NH₄ on solid surfaces (Wilhelm et. al., 1994; Wilhelm et. al., 1996). Nitrate concentrations in septic effluent after complete oxidation of NH₄ typically range from 30 to 60 mg/l (Bauman and Schafer, 1985; Hantzsche and Finnemore, 1992; Wilhelm et. al., 1994; Wilhelm et. al., 1996; ITRC, 2000).

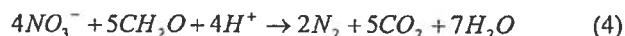
The pH and alkalinity of effluent are altered by these aerobic reactions. The oxidation of NH₄ releases 2 moles of H⁺ for every mole of N oxidized and may cause significant decreases in pH (Equation 1). Oxidation of organic C produces CO₂ which will also decrease the pH if it remains in solution (Equation 2). Sulfate concentrations have been observed to increase as effluent travels through the vadose zone, probably due to oxidation of reduced S which would also release H⁺ ions (Equation 3; Wilhelm et. al., 1994; Wilhelm et. al., 1996).



Calcium carbonate in drain field sediment often dissolves to buffer this acidity, resulting in increased Ca and total hardness in the septic effluent. Other major ions may be released during buffering reactions involving mineral dissolution and cation exchange

(Wilhelm et. al., 1994; Wilhelm et. al., 1996).

Some septic drain field settings also include a third redox zone in which denitrification occurs (Equation 4). Denitrification requires both anaerobic conditions and an electron donor such as organic labile C and usually occurs in saturated or near saturated conditions. This reaction removes NO_3^- from waste water and produces N_2 , CO_2 , and alkalinity (Wilhelm et. al., 1994; Wilhelm et. al., 1996).



A lack of labile organic C in groundwater settings is the most common limitation to denitrification (Bauman et. al., 1985). Most organic C in waste water is consumed by aerobic oxidation at the same time NO_3^- is created, so that very little waste-water organic C remains for consumption in denitrification (Wilhelm et. al., 1994; Wilhelm et. al., 1996).

Another mechanism for attenuation of NO_3^- is by the oxidation of reduced sulfur present in soils or rocks (Equation 5) (Molénat et. al., 2002; Robertson and Blowes, 1995). The consumption of sulfide minerals releases SO_4^{2-} and H^+ ions to solution. Steinheimer et al. (1998) attributed nitrate loss to autotrophic denitrification in a loess soil, with ferrous iron being the electron donors.

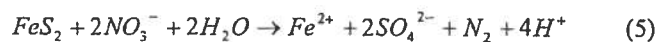


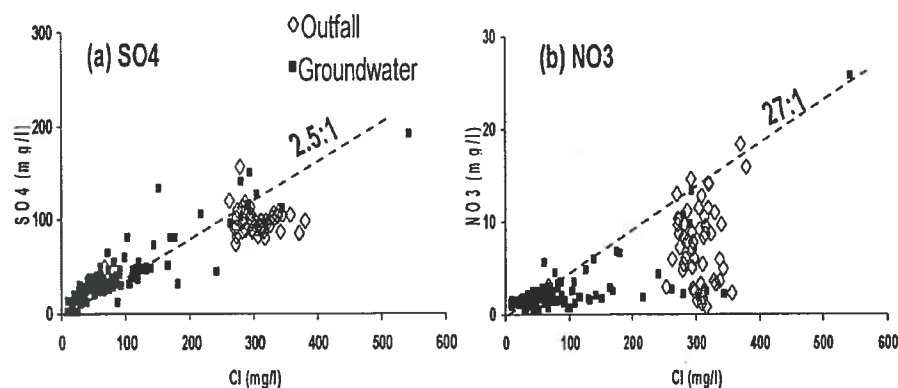
Table F-3 compares the chemical concentrations in septic tank effluent from five studies (Robertson et al., 1991; Harman et al., 1996; Wilhelm et. al., 1996; Lowe et al., 2004; McQuillan, 2004). This table shows that most of the N in septic tanks is in the form of ammonia (NH_4).

Figures F-1(a) and (b) compare concentrations of chloride with sulfate and nitrate

Table F-3: Comparisons of chemical concentrations in septic tank effluent from five studies. All results given in mg/l.

Parameter	Robertson et al., 1991	Harman et al., 1996	Wilhelm et. al., 1996	Lowe et al., 2004	McQuillan, 2004
Ca	40	137	14	-	1 - 50
Na	98	107	85	96 - 110	8 - 101
Cl	45	207	55	86 - 128	6 - 200
SO ₄	27	59	35	23 - 48	12 - 57
NO ₃ -N	1	<0.10	0.08	<1	0.1 - 2
NH ₄	30	128	58	6 - 160	0 - 36
TKN	-	152	-	35 - 170	26 - 75

TKN = Total Kjeldahl Nitrogen: a measure of the concentration of reduced forms of nitrogen in water.



Figures F-1(a) and (b): Comparison of chloride with sulfate and nitrate in samples from the outfall at the Pocatello Water Waste Treatment Facility and groundwater in the LPRV. Figure F-1(a) shows that the ratio of Cl and SO₄ in domestic waste is identical to that found in groundwater elevated in these constituents. Figure F-1(b) shows that the concentration of NO₃ varies in the outfall, but levels remain between concentrations found in groundwater suspected to be contaminated with septic effluent and groundwater where denitrification may be taking place. The dashed lines represent the 2.5:1 Cl to SO₄ ratio and the 27:1 Cl to NO₃ ratio observed in LPRV groundwater.

in outfall at the Pocatello Water Waste Treatment Facility and groundwater in the LPRV. A study of influent (waste stream) chemistry and effluent (outfall) chemistry at the Pocatello Water Waste Treatment facility revealed that calcium, sodium, potassium, chloride, and sulfate chemistry do not change during the treatment process of the sewage waste stream, while ammonium abatement typically reduces the concentration of nitrate in the outfall (Hurt, C., personal communication).

Figure F-1(a) shows that the ratio of Cl and SO₄ in treated sewage (outfall) is similar to groundwater that is suspected to be impacted by septic effluent. Figure F-1(b) shows that the outfall concentrations of NO₃ vary but remain between those found in groundwater suspected to be contaminated with septic effluent and groundwaters where denitrification may be occurring (see section 5.3.2).

Figure F-1 strongly suggests that elevated concentrations of NO₃, SO₄, and Cl found in groundwater in the LPRV reflect the influence of domestic waste from septic effluent. The higher concentrations of these salts in septic effluent may be due to differential water usage rates among municipal (sewer serviced) and rural (septic-serviced) residents.

A study conducted by the City of Pocatello revealed that single family dwellings in the city use about 500 gallons per month per household (Orr, J., personal communication). This is one and a half to three times as much as the 150 to 300 gallons per day that is cited as typical septic throughput in literature reviews (Bauman and Schafer, 1985; BBC Research and Consulting, 2001; Hantzsche and Finnemore, ITRC, 2000; 1992; Panno, 2002; Thomas et. al., 1999; Wilhelm et. al., 1994). Municipal residents appear to use more water than rural residents with private wells, thus diluting

the concentration of salts in the municipal domestic waste stream. Applying a dilution correction of 3:1 to the Cl concentrations in Figure F-1 brings these Cl concentrations to between 900 and 1200 mg/l, which falls within the range used in the septic source mixing models in Chapter 6.