Farm Well Water Quality in Alberta

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On-farm groundwater supplies in Alberta were evaluated for chemical (routine chemistry, trace metals), herbicides and microbiological (total and fecal coliforms) parameters to determine the suitability of domestic drinking water usage based on the Guidelines for Canadian Drinking Water Quality (GCDWQ). The sampling program was conducted between May and October of 1995 and 1996. Thirty-two percent of the 816 farm water wells surveyed (depth range 2 to 284 m) exceeded the GCDWQ for maximum acceptable concentration (MAC) or interim maximum acceptable concentration (IMAC) of at least one parameter. In addition, the water from 92% of the sites exceeded the GCDWQ for at least one of the aesthetic objectives (AO). The chemicals were ranked from most to least frequently exceeding the GCDWQ MAC, in the following order: F >> NO₃–N + NO₂–N > As > Se > Pb > B > U > Cr (13, 6, 3, 3, 2, 0.9, 0.4 and 0.2% of all samples, respectively). The parameters ranked from most to least frequently, exceeding the AO, in the following order: TDS > Na > Fe > Mn > pH > SO₄ > Cl > Al > Zn > Cu (85, 64, 36, 34, 23, 19, 6, 2, 1 and 0.1 % of the samples, respectively). The majority of the higher concentrations of most inorganic parameters are due to natural geological conditions defined by source aquifer mineralogy. The effects of primary agriculture are likely limited to the 3% herbicide detections and to some nitrate and microbiological contaminations observed; this water should not be used for human consumption without some form of site-specific treatment. Some rural residents may be “mistreating” their water, and a general lack of water testing among rural residents was noted.

Key words: groundwater, nitrates, trace metals, coliforms, herbicides, aquifers

Introduction

The majority of Albertans live in large towns and cities where water quality is supported by millions of dollars in infrastructure. The water provided by municipal treatment systems is required by Alberta Environment regulations to meet federal guidelines (Guidelines for Canadian Drinking Water Quality (GCDWQ) (Health Canada 1996). Operating licences

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require ongoing and stringent monitoring programs to ensure the safety of urban residents. In Alberta, groundwater is used by approximately 26% of the population, of which roughly 440,000 private domestic users make up 72% of the total (Pupp et al. 1989). The majority of Alberta’s farm population accesses groundwater supplies for domestic use with a smaller portion of the population using hauled or treated pipeline supplies and privately treated or untreated surface water supplies. The task of developing, maintaining, treating and monitoring these private water supplies is primarily left to the users. The quality of these on-farm sources can be influenced by natural or anthropogenic conditions (Craun 1985; Steichen et al. 1988; Ecobichon et al. 1990 and Geldreich 1996).

Several sources of information on farm/rural water quality in Alberta exist. In the past, rural water supply records in Alberta have been collected through local regional health units. Since 1992, all provincial Public Health chemical water quality testing records from rural water supplies have been kept in a central database (currently over 100,000 records) but they are not publicly available due to confidentiality requirements. The Public Health microbiological records for rural water supplies reside on two separate databases that are also not publicly available. Chemical and microbiological water analyses performed through private-sector laboratories are unavailable as client confidentiality is a concern. A database of 10,000 water supply records maintained by the provincial ministry of Alberta Agriculture, Food and Rural Development (AAFRD) is likely biased as these records may reflect problem water supplies that have been tested to answer specific undocumented treatment or health-related concerns.

The Canada-Alberta Environmentally Sustainable Agriculture (CAESA) Agreement, a 5-year federal-provincial research initiative, funded the Farmstead Water Quality Study (FWQS) to conduct a statistically based comprehensive survey of water quality of on-farm water wells in Alberta. The goal of the FWQS, as part of the CAESA Water Quality Monitoring Program, was to test and report on-farm groundwater supplies on a provincial basis to determine the suitability of domestic drinking water usage based on the GCDWQ.

**Materials and Methods**

**Farmstead Water Quality Survey Site Selection**

Site selection for the FWQS was based on a statistical study of the number of farms (Statistics Canada 1991) located in 64 municipalities, with a forced minimum sample size of 10 sites and a maximum of 20 sites from any one municipality. These sample sizes were based on a 95% confidence level (Allison and Neilson 1994) and the constraints of the study budget. The FWQS examined 816 sites with the understanding that in areas of the province where water wells are not commonly used, fewer sites would be available for sampling. In some situations, data from Alberta Environmental Protection’s Groundwater Information Database
were used to help identify farm sites within municipalities with small populations.

Site selection lists that provided potential participant names, legal land locations, addresses, and phone numbers on a municipality basis were supplied to sampling teams from AAFRD and Prairie Farm Rehabilitation Administration (PFRA). Each list contained approximately three times as many sites as required in order to compensate for anticipated poor participant response or inappropriate site characteristics. The sampling teams contacted the participants by telephone and arranged to collect water samples and administer a questionnaire. To enhance confidence in the sampling technique, a one-day workshop was held to train sample teams on proper sampling protocol, as outlined by public health officials. The sample teams, as a group, completed the questionnaire during the workshop. It was believed that this process would heighten the staff’s understanding of the questions while standardizing the method of administering the questionnaire, thereby reducing the potential bias of the sample teams. It was also believed that water sampling and questionnaire administration by a select group of trained individuals (sample team members) would reduce sample errors.

**Questionnaire**

The seven-page questionnaire, consisting of 94 individual questions with space for a sketch of the farm site, was developed from a pilot questionnaire tested the previous year, along with information provided through staff of AAFRD, PFRA and Alberta Health. Information related to type of operation, building and operational setbacks from the water well, some construction details such as depth, diameter and age of the well, water-treatment related information on usage and performance and personal perceptions on environmental issues was gathered through the questionnaire. The distances and locations of buildings, septic systems, fuel storage, farm operations and general topography relative to the water well were recorded on a sketch of the site to provide additional information on the site if water quality problems were encountered.

**Sampling Protocol**

The sampling program occurred between May and October of 1995 and 1996. At each site, water samples were collected for chemical, trace metal, herbicide and microbiological analyses. The water samples collected at each site (one well per site) were individually documented and labelled with unique identification numbers and shipped by courier within 24 h to laboratories for analysis. Water samples were collected from the kitchen tap or in cases where treatment was present, at a tap that bypassed treatment. Thus, these water samples obtained at the end of the distribution system do not reflect the true conditions of the groundwater because the well, pipes, connections and faucet may affect the chemical composition of the water. All samples were collected after running the
water for 3 to 5 minutes to ensure a representative sample was collected. The samples were then stored in coolers with ice packs. At several locations, duplicate samples were collected to assess variability due to sampling protocol and laboratory techniques.

Four groups of tests were performed on the water samples: routine chemistry, trace metal, microbiological and selective herbicides. The first three are standard analytical suites for rural water supplies under Alberta Health and Wellness’s Public Health chemical and microbiological water quality services provided at the Trace Elements/Environmental Toxicology Laboratory and the Northern Alberta Provincial Laboratory of Public Health, respectively. The routine chemical analysis was used to determine if the water was drinkable (potable); the trace metals analysis was used to determine if there were any trace metals contaminating the water; the microbiological analysis was used to determine the presence of potentially harmful bacteria; and the herbicide analysis was used to detect the presence of eight selected herbicides.

The results obtained from the laboratories were returned to the participants via their local Public Health Inspector, who also provided participants with specific information related to the water quality parameters analysed. For these sites with initial results that indicated water problems, resampling was conducted to minimize errors during sample collection or laboratory testing.

Routine Chemical Analyses

The routine chemistry analyses used in this study measured the concentration of total alkalinity, carbonate, bicarbonate, chloride, fluoride, nitrate-N + nitrite-N, sulphate, calcium, magnesium, sodium, potassium and iron, in addition to electrical conductivity, hardness, pH and total dissolved solids.

Water samples were collected in 500-mL PET500R (polyethylene terephthalate) sample bottles (System Plus, New Hamburg, ON). All samples were labelled for routine chemical analysis and shipped to the Provincial Laboratory of Public Health for delivery to the Capital Health Authority’s Trace Element/Environmental Toxicology Laboratory at the University of Alberta Hospital in Edmonton, Alberta. The samples were thoroughly mixed prior to aliquots being taken for the routine analyses. All chemical parameters were determined using modified American Public Health Association (APHA 1992) Standard Methods for the Examination of Water and Wastewater protocols.

Electrical conductivity, pH and fluoride concentrations were determined using Radiometer® conductivity specific ion electrodes employing a Radiometer® VIT90 autotitrator system equipped with a 40-sample SAC90 auto-sampler, a CDM 80 conductivity meter, a ABU93 25-mL triburette (for the TISAB delivery) and the TitraLab® software.

All cation analyses were performed using air/acetylene flame atomic absorption spectroscopy protocols on a Perkin Elmer® Z5000 flame atomic absorption spectrophotometer equipped with a flow spoiler burn-
er chamber and an AS40 auto-sampler. Samples were analysed for sodium, potassium, calcium and magnesium. The calcium and magnesium aliquots contained a final concentration of 0.24 N HCl and 2% La (as La(NO₃)₃) to eliminate interferences from varying concentrations of carbonate, bicarbonate and sulphate. Samples aliquoted for iron analysis were acidified with concentrated trace metal free HNO₃ to a final concentration of 1% HNO₃ to dissolve any precipitated iron.

The anion quantitations were performed on a Roche COBAS FARA II® autoanalyzer for chloride (ferricyanide method), nitrate-N + nitrite-N (hydrazine reduction method), sulphate (turbidimetric method) and alkalinity (autotitration method). Samples with pH >8.3 were manually titrated to a pH of 8.3 to assist in the proper calculation of carbonate, bicarbonate and hydroxide concentrations. Total dissolved solids and ion balances were calculated from the analytical results.

**Trace Metal Analyses**

The trace metal analyses included aluminum, antimony, arsenic, barium, beryllium, boron, cadmium, chromium, cobalt, copper, iron, lead, manganese, molybdenum, nickel, selenium, silver, strontium, titanium, thallium, uranium, vanadium and zinc.

Water samples were collected in 500-mL PET500® (polyethylene terephthalate) trace metal free sample bottles (System Plus, New Hamburg, ON). After 500 mL of water was collected, the samples were preserved with 5 mL of trace metal free 70% nitric acid (Eagle Picher® 5 mL ampule NA-6166-1EP2). All samples were properly labelled for trace metal analysis and shipped to the Provincial Laboratory of Public Health for delivery to the Capital Health Authority’s Trace Element/Environmental Toxicology Laboratory at the University of Alberta Hospital in Edmonton.

Samples were stored at 4°C prior to analysis. Samples and standards were prepared in a trace metal free (TMF), positive pressure, hepafiltered room employing Eppendorf® pipettes/tips and Sarsted® sample tubes. SPEX® certified ultra-pure single element standards were employed for the preparation of all mixed aqueous calibration standards, internal standards and quality control samples. All standards were prepared by weight in TMF Nalgene bottles, acidified with TMF HNO₃ acid and diluted to appropriate weight with Barnstead® 18 Megohm-cm TMF water. SEASTAR® TMF HNO₃ acid was used throughout.

The analyses were conducted in a separate TMF, positive pressure, hepafiltered room employing a PE-SCIEX® Elan 5000A or Elan 6000 inductively coupled plasma-mass spectrometer (ICP-MS) operating in the quantitative mode with internal standardization. All samples contained a mixed internal standard consisting of ⁴⁵Sc, ⁸⁹Y, ¹⁰³Rh and ¹⁸¹Ta. The analyses for 23 trace metals were performed using a modified Ontario Ministry of Environment DWATER protocol (determination of trace metals in potable waters by ICP-MS). The ICP-MS method employed was also consistent with the principles outlined in the U.S. EPA method 200.8, revision
5.4, 1994 protocol (determination of trace elements in waters and wastes by inductively coupled plasma-mass spectrometry [Creed et al. 1994]). All routine chemistry and trace metal analytical procedures adhered to stringent in-house quality control (QC) protocols (typically 15–25% QC samples/unknowns). The coefficient of variation (SD/mean * 100) for the routine analytes was between 0.2 and 4% while the coefficient of variation for the trace metals analytes was between 1 and 3%. In addition, the laboratory successfully passed all rounds of two external proficiency testing (PT) programs for all the parameters during the study. The first PT program was the bi-annual Alberta Water Analyst Committee PT program (between 20 to 30 laboratories) and the second was the Analytical Product Group’s (APG) WP proficiency testing (PT) program (>250 North American laboratories) on a quarterly basis. Both PT programs meet the ISO Guide 43, 1996 Proficiency Testing Program protocols and employ z-scores for performance evaluations. The APG program, the largest PT provider of water samples in North America, is ISO 9001 certified and is accredited by the new NAVLAP/NIST Proficiency Testing Program.

Microbiological Analysis

The microbiological analysis, performed by the Northern Alberta Provincial Laboratory for Public Health, tested the water samples for both total and fecal coliform bacteria, which are an indicator organism for more harmful organisms.

Because it is not practical to monitor all pathogens in drinking water, microbiological guidelines are based on detection of “indicator organisms”. The coliform group of bacteria is used as an indicator of the suitability of water for drinking or other purposes (Health Canada 1996). While these bacteria serve as a good indicator of the presence of pathogenic bacteria, they are not reliable in indicating the presence of some protozoa such as *Giardia* or *Cryptosporidium* or the presence of some viruses.

The presence of fecal coliform, a subgroup of the coliform group, indicates the possible presence of fecal contamination. Some species of bacteria in the fecal group occur naturally in soil and on vegetation; however, species within the group *Escherichia coli* are a definite indicator of fecal contamination from a warm-blooded source. The GCDWQ MAC for human drinking water is zero fecal coliform detected per 100 mL. If their presence is confirmed, the water source should not be used until it is detection free.

The maximum acceptable concentration (MAC) for total coliform in drinking water is 10 colonies/100 mL (Health Canada 1996). However, because coliforms are not uniformly distributed in water, if up to 10 total coliform colonies/100 mL are detected, the water should be resampled. If the presence of coliform is reconfirmed, the cause should be determined and corrective action taken. Ideally, coliform bacteria should not be present in well water because these organisms require a nutrient source to survive, and are generally associated with plant and animal life processes that do not occur in deep wells (Health Canada 1996).
The presence of coliform (total and fecal) bacteria was determined by either membrane filtration or by multiple tube fermentation, followed by selective media culture. Analysis was performed using modified American Public Health Association (APHA 1992) Standard Methods for the Examination of Water and Wastewater protocols (Ashton 1996). Water samples were collected in sterile 250-mL polyethylene sample bottles containing sodium thiosulphate. The samples were shipped at approximately 5 °C and received by the Northern Alberta Laboratory of Public Health, at the University of Alberta Hospital in Edmonton within 24 h of sampling. Samples received for analyses between 24 and 48 h after collection were accepted but microbiological growth was suspect due to possible contamination. Any samples received after 48 h were retaken.

Phenoxy/Neutral Herbicide Analysis

Eight herbicides (MCPA, 2,4-D, dicamba, bromoxynil, triallate, trifluralin, diclofop-methyl and fenoxyprop) were chosen for assessment based on research by Anderson (1995) and Cotton and Bytrus (1995), who utilized provincial sales figures and chemical persistence data as well as the field plot and groundwater monitoring research of Miller et al. (1992), Rodvang et al. (1992) and Hill et al. (1996). A 1-L glass bottle was used to collect water samples which were cooled (approximately 5°C) and shipped to Enviro-Test Laboratories, Edmonton, Alberta. The water samples were paired by blending 0.5 L of one site sample with 0.5 L of another site sample and then analyzed as a composite sample to decrease project costs. If herbicides were detected in the composite samples, the individual samples were re-analysed separately. Overall the composite results compared closely with the individual re-extraction results. Some variability in the individual versus composite results is expected with this type of re-analysis. There are several factors that affect this, such as the amount of particulate or sediment found in the original sample and the variability in the quantitation at low levels. The quantitation using HP 5970 mass selective detector with a 5890A gas chromatograph (GC/MSD) has a typical coefficient of variation ranging from 10 to 25%. The instrument was run on a selected ion monitoring mode in which three ions are selected for each analyte (Bruns et al. 1991; Bruns 1996). The identification of an analyte in a sample must be confirmed for all three ions at the expected retention time and ion ratios. The variability of the results increases near the quantitation limit (0.05 ppb) due to an increase in signal to noise ratio, which may approach 50%. Tests on stability reveal that there is little degradation of herbicides in water stored at refrigerated conditions (0–4°C) (Bruns 1996).

Statistical Analyses

Statistical analyses (including descriptive statistics, stepwise regression analysis to identify significant independent variables, chi-square analysis and F-test through one-way ANOVA analysis) for the FWQS data were performed using SPSS Base 8.0 (SPSS 1998) statistical software. Values less than LOQ were set to zero. Data were not transformed.
Qualitative data, such as presence of livestock, were assigned a value of one for presence and zero for absence.

Results and Discussion

While a total of 824 farm sites were visited, sampled and surveyed during 1995 and 1996, some discrepancies in the numbers reported were encountered due to incomplete survey questionnaires or missing samples. The total number of sites sampled was roughly the same each year: 407 water wells in 1995 and 413 water wells in 1996. Of these sites, 816 were completely matched to both the questionnaire and laboratory analysis.

Water samples from 32% of the sites exceeded the GCDWQ (Health Canada 1996) maximum acceptable concentration (MAC) or interim maximum acceptable concentration (IMAC) for one or more of the parameters analysed (Tables 1 and 2). Water from approximately 92% of the sites failed to meet the aesthetic objectives (AO) set out under the GCDWQ. Although the samples were randomly collected over a 2-year period, little difference in water quality was noted between years.

Chemical and Trace Metal Analyses

Twenty-eight percent (226/816) of the FWQS well water samples exceeded either the MAC or IMAC GCDWQ for one or more of the following elements: arsenic, barium, chromium, fluoride, lead, nitrate, selenium and uranium. In addition, 92% (754/816) of the sites exceeded one or more of the aesthetic objectives (AO) set for chloride, pH, sodium, sulphate, iron, total dissolved solids (Table 1), copper, iron, manganese and zinc (Table 2). The data were not significantly different between years. Most of the samples had non-quantifiable levels of antimony, beryllium, cadmium, cobalt, silver, thallium and vanadium (Table 2).

Nitrates

The mean nitrate-N + nitrite-N (nitrate) concentration from the FWQS samples was 2.19 mg L^{-1} (Table 1), which would lend support to the assumption made by Madison and Brunett (1985) that levels above 3.0 mg L^{-1} might involve anthropogenic sources. Six percent of the samples (46) tested had concentrations of nitrate that exceeded the MAC of 10 mg L^{-1}. These 46 nitrate detections were randomly distributed throughout the province. The proportion of samples exceeding the MAC for nitrate in these Alberta data is lower than values reported in Ontario (14%) by Goss et al. (1997), in Nebraska (19%) by Gosselin et al. (1997) and Missouri (22% >45 ppm NO₃) by Sievers and Fulhage (1992); however, it is similar to those values found in Montana (6%) by Bauder et al. (1991).

Of the 816 FWQS sites, 486 (Table 1; 60%) had concentrations below the limit of detection for nitrate (0.05 mg L^{-1}). A higher mean concentration was observed in 1996 (2.53±standard error of mean 0.35 mg L^{-1}) versus 1995 (1.89±0.42 mg L^{-1}). However, there was no significant difference in nitrate concentration (p=0.251) between the year the sample was taken.
Table 1. Descriptive statistics for analyses of chemical properties (816 samples)

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<th>Parameter</th>
<th>Units</th>
<th>Mean</th>
<th>Std. dev.</th>
<th>Median</th>
<th>Min.</th>
<th>Max.</th>
<th>LOQ&lt;sup&gt;b&lt;/sup&gt;</th>
<th>No. &lt;LOQ</th>
<th>Standard</th>
<th>CDWQG (1996)&lt;sup&gt;a&lt;/sup&gt;</th>
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<td></td>
<td></td>
<td></td>
<td>No. &gt;MAC&lt;sup&gt;c&lt;/sup&gt; No. &gt;IMAC&lt;sup&gt;d&lt;/sup&gt; No. &gt;AO&lt;sup&gt;e&lt;/sup&gt;</td>
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<td>Alkalinity (total)</td>
<td>mg L&lt;sup&gt;-1&lt;/sup&gt;</td>
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<td>220</td>
<td>526</td>
<td>36</td>
<td>1490</td>
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<td>264</td>
<td>624</td>
<td>44</td>
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<td>78</td>
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<td>F</td>
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<td>342</td>
<td>128</td>
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<td>2</td>
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<td>0.14</td>
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<td>9.16</td>
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<td>729</td>
<td>905</td>
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<td>5652</td>
<td>1</td>
<td>500</td>
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<sup>a</sup> CDWQG, Canadian Drinking Water Quality Guidelines.
<sup>b</sup> LOQ, limit of quantitation as defined by the IUPAC Convention (Analytical Chem., June 1983).
<sup>c</sup> MAC, maximum acceptable concentration.
<sup>d</sup> IMAC, interim maximum acceptable concentration.
<sup>e</sup> AO, aesthetic objective.
### Table 2. Descriptive statistics for analyses of trace elements

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Units</th>
<th>Mean</th>
<th>Std. dev.</th>
<th>Median</th>
<th>Min.</th>
<th>Max.</th>
<th>LOQ&lt;sub&gt;b&lt;/sub&gt;</th>
<th>No. &lt;LOQ</th>
<th>No. &gt;MAC&lt;sub&gt;c&lt;/sub&gt;</th>
<th>No. &gt;IMAC&lt;sub&gt;d&lt;/sub&gt;</th>
<th>No. &gt;AO&lt;sub&gt;e&lt;/sub&gt;</th>
<th>No. exceeding&lt;sub&gt;f&lt;/sub&gt;</th>
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<tr>
<td>Aluminum</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
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<td>69</td>
<td>4</td>
<td>&lt;1</td>
<td>1414</td>
<td>1</td>
<td>277</td>
<td>200&lt;sup&gt;f&lt;/sup&gt;</td>
<td>13</td>
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<td>Antimony</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
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<td>0.7</td>
<td>&lt;1</td>
<td>&lt;1</td>
<td>8</td>
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<td>Arsenic</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>4</td>
<td>9.4</td>
<td>1</td>
<td>&lt;1</td>
<td>119</td>
<td>1</td>
<td>406</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Barium</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>101</td>
<td>199</td>
<td>45</td>
<td>&lt;1</td>
<td>2751</td>
<td>1</td>
<td>1</td>
<td>1,000</td>
<td>7</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Beryllium</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>&lt;1</td>
<td>0.1</td>
<td>&lt;1</td>
<td>&lt;1</td>
<td>5</td>
<td>1</td>
<td>811</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Boron</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>436</td>
<td>515</td>
<td>272</td>
<td>8</td>
<td>4346</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
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<td></td>
</tr>
<tr>
<td>Cadmium</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>&lt;1</td>
<td>0.2</td>
<td>&lt;1</td>
<td>&lt;1</td>
<td>2</td>
<td>1</td>
<td>810</td>
<td></td>
<td></td>
<td></td>
<td></td>
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<tr>
<td>Chromium</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>9</td>
<td>6.4</td>
<td>7</td>
<td>&lt;1</td>
<td>71</td>
<td>1</td>
<td>24</td>
<td>5000</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cobalt</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>&lt;1</td>
<td>0.5</td>
<td>&lt;1</td>
<td>&lt;1</td>
<td>6</td>
<td>1</td>
<td>769</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Copper</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>17</td>
<td>62</td>
<td>4</td>
<td>&lt;1</td>
<td>1517</td>
<td>1</td>
<td>1,63</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Iron</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>892</td>
<td>2,620</td>
<td>106</td>
<td>&lt;20</td>
<td>31,460</td>
<td>20</td>
<td>289</td>
<td>300</td>
<td>282</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lead</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>1</td>
<td>27.2</td>
<td>&lt;1</td>
<td>&lt;1</td>
<td>137</td>
<td>1</td>
<td>628</td>
<td>10</td>
<td>14</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Manganese</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>135</td>
<td>334</td>
<td>19</td>
<td>&lt;1</td>
<td>5334</td>
<td>1</td>
<td>43</td>
<td>50</td>
<td>281</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Molybdenum</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>4</td>
<td>10.6</td>
<td>1</td>
<td>&lt;1</td>
<td>309</td>
<td>1</td>
<td>301</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Nickel</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>2</td>
<td>8.0</td>
<td>&lt;1</td>
<td>&lt;1</td>
<td>237</td>
<td>1</td>
<td>578</td>
<td></td>
<td></td>
<td></td>
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</tr>
<tr>
<td>Selenium</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>2</td>
<td>6.0</td>
<td>&lt;1</td>
<td>&lt;1</td>
<td>94</td>
<td>1</td>
<td>594</td>
<td>10</td>
<td>28</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Silver</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>&lt;1</td>
<td>0.2</td>
<td>&lt;1</td>
<td>&lt;1</td>
<td>2</td>
<td>1</td>
<td>803</td>
<td></td>
<td></td>
<td></td>
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</tr>
<tr>
<td>Strontium</td>
<td>µg L&lt;sup&gt;-1&lt;/sup&gt;</td>
<td>605</td>
<td>657</td>
<td>383</td>
<td>11</td>
<td>6,446</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Element</td>
<td>µg L⁻¹</td>
<td>5</td>
<td>28.6</td>
<td>&lt;1</td>
<td>&lt;1</td>
<td>749</td>
<td>1</td>
<td>452</td>
<td></td>
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<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Titanium</td>
<td>µg L⁻¹</td>
<td>&lt;1</td>
<td>0.2</td>
<td>&lt;1</td>
<td>&lt;1</td>
<td>4</td>
<td>1</td>
<td>805</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Thallium</td>
<td>µg L⁻¹</td>
<td>5</td>
<td>21.5</td>
<td>&lt;1</td>
<td>&lt;1</td>
<td>585</td>
<td>1</td>
<td>479</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Uranium</td>
<td>µg L⁻¹</td>
<td>&lt;1</td>
<td>&lt;1</td>
<td>&lt;1</td>
<td>&lt;1</td>
<td>7</td>
<td>1</td>
<td>758</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Vanadium</td>
<td>µg L⁻¹</td>
<td>329</td>
<td>1600</td>
<td>32</td>
<td>&lt;1</td>
<td>17606</td>
<td>1</td>
<td>30</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Zinc</td>
<td>µg L⁻¹</td>
<td>11</td>
<td>1492</td>
<td>749</td>
<td>1</td>
<td>17606</td>
<td>1</td>
<td>30</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

*CDWQG, Canadian Drinking Water Quality Guidelines.*

*LOQ, limit of quantitation as defined by the IUPAC Convention (Analytical Chem., June 1983).*

*MAC, maximum acceptable concentration.*

*MAC, interim maximum acceptable concentration.*

*AO, aesthetic objective.*

*200 µg L⁻¹ proposed guideline for aluminum in water treatment plants (Health Canada 1999).*
Nitrate contamination is often associated with nitrogen fertilizers, barnyard manure, and septic systems used on farms and by rural residents. However, some aquifers in Alberta contain water with nitrate concentrations as high as 1800 mg L\(^{-1}\) (Hendry et al. 1984; Pupp et al. 1989). Madison and Bruett (1985) found in a U.S. water quality survey that nitrate levels greater than 3 mg L\(^{-1}\) were generally anthropogenic in nature.

Baker (1990), Kross et al. (1993) and Dasika and Atwater (1995) found that sites with high levels of nitrates appeared to be most prevalent in shallow wells within unconfined aquifers. Forty four of the 46 FWQS samples exceeding the 10 mg L\(^{-1}\) nitrate MAC were from water wells less than 30 m deep. Such an observation was also made by Rudolph and Goss (1993) in the Ontario Farm Groundwater Quality Survey, where depth of well along with well age and construction type were significantly correlated with nitrate concentrations.

The physical conditions of a well, including casing, seal or pit, affect the possible entry of nitrates from a surface or near-surface source rather than the groundwater supply. Exner and Spalding (1985), Sievers and Fulhage (1992), Sharpe et al. (1993) and Goss and Barry (1995) provided this rationale for the large percentage of elevated nitrate detections. Exner and Spalding (1985) also concluded that wide ranges in nitrate concentration and a high degree of aquifer heterogeneity over a short distance are indicative of point source contamination. Similar results are likely for Alberta given similar aquifer conditions. However, Hallberg (1987), in an Iowa study, found that nitrate contamination in less than 2% of the wells sampled could be attributed to well placement or construction problems.

Commercial fertilizers were used within 50 m of the wells in 8.7% (4/46) of the cases where nitrate exceeded the 10 mg L\(^{-1}\) MAC, compared to 8.4% (65/770) of the sites where the MAC for NO\(_3\) was not exceeded. Hallberg (1987) stated that groundwater contamination may not be evident for many years after excess fertilizer-N application begins in areas with a thick, medium to fine textured soil mantle and that not enough time had elapsed for the surficially applied chemicals over the past 10 to 20 years to reach the deeper portions of the Iowa groundwater system.

Other factors that can influence the NO\(_3\) concentration in groundwater, as previously indicated, include human septic systems and livestock and their associated manure. Seventy-eight percent (36) of the 46 sites exceeding the MAC for NO\(_3\) had livestock while 75% (576/770) of the FWQS sites that did not exceed the MAC for nitrate also had livestock. From the FWQS data, 33% (16/46) of the NO\(_3\) contaminated sites had a septic system within 50 m of the well as compared to 21.6% (159/770) of the sites with nitrate less than the MAC, which had a septic system within this distance. Overall, in the FWQS, septic systems within 50 m of the wellhead were not significantly related to NO\(_3\) concentrations (p = 0.098).

The nitrate results in the FWQS were similar to those of Pupp et al. (1989), Henry and Meneley (1993) and Goss et al. (1997). The influence of livestock and their associated corrals and feedlot areas as well as the storage of manure and proximity of septic systems to the wellhead did not significantly affect nitrate concentrations observed, a trend similar to
those of Hallberg (1987) and Goss et al. (1997). However, manure spreading activities were found to have a significant ($p=0.043$) but negative relationship to the proximity of the wellhead.

**Coliform**

In the FWQS study, $3.1\%$ (25/816) of the sites tested positive for fecal coliform, while $7.6\%$ (62/816) of the samples exceeded the MAC (>10 colonies per 100 mL) for total coliform and $13.8\%$ (113/816) of the samples tested positive for total coliform greater than 1 colony per 100 mL (Table 3). Coliform bacteria were detected in $20.1\%$ (66/329) of the wells with a depth less than 30 m, of which 53 were dug wells. In comparison, $9.6\%$ (47/487) of the wells deeper than 30 m tested positive for the presence of coliform bacteria.

Some of the obvious sources of fecal contamination include near surface entry into the well from septic systems, manure storage facilities, farmyard fecal sources, vermin gaining entry into the well, livestock and pets living too close to the well head or the presence of a compromised well seal. Rudolph et al. (1992) found 31% of the 1300 wells sampled in Ontario exceeded total coliform levels, and 20% exceeded the fecal coliform levels. Detected concentrations in Ontario were not significantly correlated to presence of animals, proximity of septic system, or manure storage.

**Table 3.** Farmyard practices that can potentially affect total coliform (TC) bacteria detection levels

<table>
<thead>
<tr>
<th>Practices</th>
<th>Number of wells with TC (113)</th>
<th>Percent of total sites with TC</th>
<th>Number of wells without TC (703)</th>
<th>Percent of total sites without TC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Unused wells</td>
<td>44</td>
<td>39</td>
<td>313</td>
<td>45</td>
</tr>
<tr>
<td>Water pooling at well</td>
<td>9</td>
<td>8</td>
<td>39</td>
<td>6</td>
</tr>
<tr>
<td>Well pit used</td>
<td>38</td>
<td>34</td>
<td>219</td>
<td>31</td>
</tr>
<tr>
<td>Septic system &lt;50 m from well</td>
<td>25</td>
<td>22</td>
<td>151</td>
<td>21</td>
</tr>
<tr>
<td>Livestock present</td>
<td>73</td>
<td>65</td>
<td>614</td>
<td>87</td>
</tr>
<tr>
<td>Livestock at code levels&lt;sup&gt;a&lt;/sup&gt;</td>
<td>9</td>
<td>8</td>
<td>74</td>
<td>11</td>
</tr>
<tr>
<td>Corral &lt;50 m from well</td>
<td>23</td>
<td>20</td>
<td>200</td>
<td>28</td>
</tr>
<tr>
<td>Feedlot &lt;50 m from well</td>
<td>9</td>
<td>8</td>
<td>58</td>
<td>8</td>
</tr>
<tr>
<td>Manure storage &lt;50 m from well</td>
<td>4</td>
<td>3.5</td>
<td>31</td>
<td>4</td>
</tr>
<tr>
<td>Manure applied &lt;50 m from well</td>
<td>1</td>
<td>0.8</td>
<td>39</td>
<td>6</td>
</tr>
</tbody>
</table>

<sup>a</sup> 1995 Code of Practice for the Safe and Economical Handling of Animal Manure.
Presence of livestock was found to have a significant but negative relationship to the presence of total coliform bacteria (Table 3, p=0.005). It was found that 11.9% (73/613) of the sites with livestock had total coliform detections as opposed to 19.6% (40/203) of those sites without livestock having total coliform detections. Other factors which may affect the presence of bacteria at the well include the presence of old, unused wells (p=0.313), use of well pits (p=0.597) and pooling of surface water (p=0.312) at the wellhead. Factors such as proximity of the well to corrals, feedlots, manure storage, manure application and septic systems can potentially increase bacteria levels in wells; however, relationships from the FWQS data were not statistically significant.

The mean age of wells containing total coliform was 20.4±1.6 years and without total coliform 19.5±0.58 years. The mean age of the septic system at sites with total coliform was 17.7 ± 1.3 years and those without total coliform was 17.3 ± 0.4 years. While at many of the sites some form of water treatment was used, water was disinfected before it was used at only a few sites. Approximately 49 % of the sites that had a total coliform detection had either no previous testing (15/112) or had only initial testing (40/112) when the well was first completed. Hallberg (1987) noted in a review of over 16,000 water quality analyses from northeastern Iowa that bacterial problems were randomly distributed in relation to geologic setting and well depth. This observation was also made for the FWQS data from Alberta.

There was a significant correlation (p=0.006) between all total coliform detections (13.8% of the 816 sites sampled) and higher nitrate levels (>10 mg/L). The shallower wells (<30 m) sampled were also significantly related to total coliform detection (p <0.001); however, this was not the case for older wells and total coliform detection (p=0.649).

**Herbicides**

Twenty-seven (3.3%) of the 816 samples from the FWQS had a measurable herbicide detection (limit of 0.05 µg L⁻¹ or greater except for fenoxaprop at a detection level of 0.10 µg L⁻¹ or greater). Of those sites with a detection, 29.6% had more than one herbicide present.

Five samples had measurable amounts of dicamba (Table 4). While none exceeded the guideline of 120 µg L⁻¹, the largest concentration detected was 9.5 µg L⁻¹. Seventeen of the samples contained measurable amounts of MCPA; the highest concentration was 3.10 µg L⁻¹. Currently, MCPA is under review (Caux et al. 1995; Health Canada 1996) with a possible guideline of 10 µg L⁻¹, which is in keeping with the U.S. EPA. Two of the 12 samples containing measurable amounts of 2, 4-D exceeded the guidelines (100 µg L⁻¹), with the highest concentration being 206 µg L⁻¹. However, eight samples contained less than 1 µg L⁻¹ of 2,4-D.

Four samples contained levels of bromoxynil, two at less than 1 µg L⁻¹ and one at a concentration of 29 µg L⁻¹, which exceed the IMAC guideline of 5 µg L⁻¹ (Table 4). Fenoxaprop at a concentration of 3.40 µg L⁻¹ was found in one sample. However, there is no guideline for fenoxaprop.
Table 4. Number of detection and statistics for herbicides

<table>
<thead>
<tr>
<th>Name of herbicide</th>
<th>LOQ (µg L(^{-1}))</th>
<th>Number of detections</th>
<th>Maximum concentration (µg L(^{-1}))</th>
<th>GCDWQ MAC (µg L(^{-1}))</th>
<th>GCDWQ IMAC (µg L(^{-1}))</th>
<th>No. MAC or IMAC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dicamba</td>
<td>0.05</td>
<td>5</td>
<td>9.5</td>
<td>120</td>
<td>—</td>
<td>0</td>
</tr>
<tr>
<td>MCPA</td>
<td>0.05</td>
<td>17</td>
<td>3.1</td>
<td>—</td>
<td>10 (^b)</td>
<td>2</td>
</tr>
<tr>
<td>2,4-D</td>
<td>0.05</td>
<td>12</td>
<td>206.0</td>
<td>100</td>
<td>—</td>
<td>1</td>
</tr>
<tr>
<td>Bromoxynil</td>
<td>0.05</td>
<td>4</td>
<td>29.0</td>
<td>5</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>Fenoxaprop(^a)</td>
<td>0.10</td>
<td>1</td>
<td>3.4</td>
<td>—</td>
<td>—</td>
<td>0</td>
</tr>
<tr>
<td>Diclofop-methyl</td>
<td>0.05</td>
<td>2</td>
<td>3.0</td>
<td>9</td>
<td>—</td>
<td>0</td>
</tr>
<tr>
<td>Trifluralin</td>
<td>0.05</td>
<td>1</td>
<td>0.8</td>
<td>—</td>
<td>45</td>
<td>0</td>
</tr>
<tr>
<td>Triallate</td>
<td>0.05</td>
<td>1</td>
<td>3.1</td>
<td>230</td>
<td>—</td>
<td>0</td>
</tr>
</tbody>
</table>

\(^a\) No standard set in GCDWQ.

\(^b\) Under review.

GCDWQ, Guidelines for Canadian Drinking Water Quality; MAC, maximum acceptable concentration; IMAC, interim maximum acceptable concentration; LOQ, limit of quantitation.
under the current GCDWQ; therefore, it is not known at what level human health may be compromised. Two samples contained detectable levels of diclofop-methyl; one sample had 0.094 µg L\(^{-1}\), and the other had a concentration of 3 µg L\(^{-1}\). The CDWQG for diclofop-methyl is 9 µg L\(^{-1}\). One sample had a detectable concentration of 0.80 µg L\(^{-1}\) of trifluralin and one sample had a detected triallate concentration of 3.10 µg L\(^{-1}\) (MAC = 230 µg L\(^{-1}\)).

Contamination of water wells by herbicides is often linked to site specific practices (herbicide handling) and is generally not indicative of widespread contamination. Kross et al. (1992) found that 25% of the wells that had herbicide levels which exceeded the Lifetime Health Advisory Levels in Iowa were due to point source contamination from back-siphoning or spills. It was noted by FWQS sample team members that at two of the 27 sites, back-siphoning of herbicides into the well had occurred. Frank (1981) noted that in a 10-year period, between 1969 and 1978, in Ontario, 19% of 159 contaminated wells were contaminated due to accidental back-siphoning, overfilling, or rinsing spray equipment, while 31% of the pesticide contamination cases were caused by spraying weeds or crops close to the well. Survey data for the FWQS indicated that only 37% (10/27) of the sites with a herbicide detection made use of the water well as a spray water source. This relationship to the non-spray water use detections was not pursued any further under the FWQS. At only one of the 27 sites did the respondents indicate that general herbicide application activities had occurred within 50 m of the wellhead.

Well owners were asked in the survey questionnaire if herbicides applied on fields (47/816 responded yes) or herbicide tank mixing/washing operations (150/816 responded yes) were a potential source of contamination to their water supply. Of those who responded “yes” to the above question, nine had a herbicide detection. Respondents at 8 of the 27 locations with a detection believed field applied herbicides had more of a contaminant potential than 2 of 27 for tank mixing, which could be directly related to management practice at the wellhead.

Twenty-seven sites had detectable herbicide concentrations. However, at 63% (17/27) of these sites, the respondents indicated that the water source tested had not been used for herbicide spray water and, therefore, no herbicides were mixed at the wellhead or farmyard. The average depth and age of the well, and activities occurring around the wells with herbicide detections were similar to those of wells where detection did not occur.

The number of herbicide detections was similar or lower than those found in other areas of North America. Of the 27 sites with detections, all but one had no detection or a significant decrease in the herbicide levels when the sites were resampled. This observation is in keeping with the findings of others that seasonal fluctuations based on local agronomic practices have the potential to affect groundwater quality as do the direct timing and handling of the herbicides. This temporal effect related to agronomic practice needs to be addressed to better identify cause and
effect relationships which the FWQS data cannot. The FWQS was a one-time survey to establish benchmark conditions; therefore, it did not include measurements of the movement of the herbicides within the specific aquifer. These measurements may have identified other sources or farm operations which could be a contaminant source.

Parameters Exceeding the MAC or IMAC

The chemicals were ranked from most to least frequently exceeding the GCDWQ MAC or IMAC, in the following order: F>>NO$_3$-N+ NO$_2$-N> As>Se>Pb>B>U>Cr (13, 6, 3, 3, 2, 0.9, 0.4, 0.2% of all samples, respectively).

Approximately 13% of the samples had fluoride levels that exceeded the MAC of 1.5 mg L$^{-1}$ (Table 1); 71 samples had concentrations between 1.5 and 2.5 mg L$^{-1}$, 22 had concentrations between 2.5 and 3.5 mg L$^{-1}$, and 3 samples had concentrations between 3.5 and 4.0 mg L$^{-1}$. Nine samples had concentrations that exceeded 4 mg L$^{-1}$ and one sample contained 6.4 mg L$^{-1}$ of fluoride. Thirty-one percent of the samples had concentrations of fluoride that were low enough (<0.5 mg L$^{-1}$) that the daily intake of fluoride would require supplementation to ensure adequate intake for proper bone and tooth development.

Only 3% of the samples tested had arsenic concentrations that exceeded the IMAC levels (Table 2; 25 µg L$^{-1}$); 50% (406) of the samples tested had levels below the 1 µg L$^{-1}$ limit of quantitation (LOQ). In addition, 47% (386) of the samples had concentrations that were measurable but were below the IMAC level. Of the 21 sites (3%) that had concentrations of arsenic that exceeded the IMAC level, two samples had concentrations greater than 100 µg L$^{-1}$.

Miller et al. (1996) measured concentrations of arsenic as high as 5340 µg L$^{-1}$ in southern Alberta in shallow (2.18 to 10.8 m) saline well sites which were hillside seep or artesian discharge in nature. Welch et al. (1988) found that aquifers underlain by sandstone or carbonate rocks were low in arsenic. However, higher concentrations in the Western United States were associated with basin-fill deposits of alluvial-lacustrine origin, mainly in semi-arid areas or in volcanic deposits, or under the effects of geothermal systems or in uranium and gold mining areas. Arsenic can be derived from arseno-sulfide minerals, phosphate rock, fertilizers or pesticides (Welch et al. 1988; Miller et al. 1996). In Alberta, the high arsenic concentrations in well waters are generally associated with sampling points in areas that are underlain by bedrock geological formations having marine shale, which frequently contain pyrite. Pyrites in such deposits are known to be enriched by arsenic. As well, the mobility of arsenic in sedimentary aquifers may be, in part, a result of changes in the geochemical environment. At greater depths, high concentrations are associated with compaction caused by groundwater withdrawals (Welch et al. 1988). It is therefore concluded that generally the elevated arsenic levels are the result of natural aquifer conditions; however, without further investigation, anthropogenic factors cannot be discounted.
One percent of the FWQS samples obtained in each of the 2 years exceeded the MAC of 1000 µg L\(^{-1}\) for barium (Table 2). Only two samples (0.2%) had levels of chromium that exceeded the MAC of 50 µg L\(^{-1}\).

The majority of the samples had very low concentrations of lead (Table 2; mean=1 µg L\(^{-1}\) with 628 <LOQ). Most samples had concentrations lower than the MAC of 10 µg L\(^{-1}\). Thirteen samples had concentrations between 10 and 100 µg L\(^{-1}\) and one sample had a concentration of 137 µg L\(^{-1}\). Lead poisoning has been associated with lead pipes used for plumbing in older houses (Swistock et al. 1993). However, the sample containing 137 µg L\(^{-1}\) was from a new modular home with a well less than 3 months old. When resampled (4 weeks later), the levels were well below the MAC.

Three percent (28 samples) were found to have levels of selenium (Se) > the MAC of 10 µg L\(^{-1}\) (Table 2). Of these 28 samples, the majority (23 samples) had concentrations <30 µg L\(^{-1}\), two samples had concentrations between 41 and 50 µg L\(^{-1}\), and three had concentrations >50 µg L\(^{-1}\). Steichen et al. (1988) found 9/103 Kansas well sites with Se levels >10 µg L\(^{-1}\), attributing some of these concentrations to naturally occurring soil and rock formations in exposed cretaceous shales. Miller et al. (1978) found that 30% of the 160 wells tested in a Montana study had Se levels > the MAC. In Alberta, Warren and Dudas (1992) found Se concentrations generally <0.5 mg kg\(^{-1}\) in till materials; however, weathering of shallow outcrops of sedimentary rocks such as shales with pyrite may induce higher Se levels in parts of Alberta (Miller et al. 1996). Other possible selenium sources include fertilizers and feed additives (Miller et al. 1996).

Less than 1% (3/816) of the samples had concentrations of uranium > the MAC of 100 µg L\(^{-1}\) (Table 2). These levels are interpreted as naturally occurring due to the background geology of the aquifers from which the water was drawn. In all three cases, concentrations of TDS, sulfate, sodium and selenium or nitrate were above the GCDWQ; as well, two of the samples had concentrations of chloride and boron or manganese above the MAC or AO guidelines. At only one of the sites was the water consumed by the residents. At that site, health-related problems were noted; however, these problems were originally attributed to high sodium. The other two sites made use of the water supply for livestock, but health problems in the stock were also noted.

While most of samples with such exceeding levels would be due to the background geological conditions of Alberta (Pupp et al. 1989; Miller et al. 1996), presence of lead and some of the other parameters (e.g., copper, zinc) would likely be due to the water distribution system (i.e., well casing, waterlines, connections and faucets).

The GCDWQ MAC for individual compounds is set at a level to avoid health risks even if the water contains that substance at that concentration and is consumed for an entire lifetime. However, some chemicals for which a MAC is defined are more toxic than others, or may affect individuals differently, depending upon the health and age of the individual. Therefore, the water at 32% of the sites that exceeded the MAC and IMAC for at least one component is considered poor and should not be used for human consumption without further investigation.
Parameters Exceeding the Aesthetic Objectives

The parameters ranked from most to least frequently exceeding the AO, in the following order: TDS>Na>Fe>Mn>pH>SO₄>Cl>Al>Zn>Cu (85, 64, 36, 34, 23, 19, 6, 2, 1 and 0.1 % of the samples, respectively). Ninety-two percent of the 816 FWQS well water samples exceeded at least one aesthetic objective (AO) defined by the GCDWQ, and in many cases two or more. While the AO many not have a direct health effect, exceeding the AO may decrease the palatability of drinking water, increase staining of fixtures and laundry, or interfere with water treatment processes (Health Canada 1996). Fifty-one samples (6%) had concentrations that exceeded the AO of 250 mg L⁻¹ for chloride (Table 1). The water supply was treated at 21 of the 51 sites; however, at only 11 sites were treatment methods used that were capable of removing chloride ions. Nine samples contained more than 1000 mg L⁻¹ of chloride and one sample contained 3,150 mg L⁻¹.

Twenty-three percent (188/816) of the samples exceeded the AO guidelines for pH defined by the GCDWQ of between 6.5 and 8.5 (Table 1). Sixty-four percent (523/816) of the FWQS samples had sodium concentrations that exceeded the GCDWQ AO of 200 mg L⁻¹ (Table 1). At this level of sodium the taste of the water can become offensive. Twenty-three samples had concentrations of sodium that exceeded 1000 mg L⁻¹ with one sample having 2100 mg L⁻¹. Thirteen percent (67/523) of those respondents whose water exceeded the GCDWQ for sodium also indicated that they used a water softener. Approximately 67% (45) of those sites with high sodium and softening equipment also had a total hardness greater than 200 mg L⁻¹, with the highest being 2324 mg L⁻¹. Treating the water to make it softer by replacing the calcium and magnesium ions (to reduce total hardness) with sodium only increases the sodium content of the water that in these cases already exceed the AO, thus increasing potential health concerns.

Nineteen percent of the samples exceeded the AO of 500 mg L⁻¹ for sulphate set in the GCDWQ. Fifty water samples had concentrations of sulphate greater than 1000 mg L⁻¹, with one sample containing 3370 mg L⁻¹. Eighty-five percent (692) of the samples taken exceeded the AO guideline of 500 mg L⁻¹ for total dissolved solids (TDS).

Thirty-six percent (293/816) of the samples had iron concentrations that exceeded the AO of 0.3 mg L⁻¹. None of the samples had concentrations that were high enough to cause health concerns. The majority (76%) of respondents whose well water samples indicated high levels of iron claimed to have staining problems due to the water and 31% of those made use of an iron filter to remove some of the iron from their water. Approximately 44% of those with high iron concentrations indicated that they shock chlorinated their well or used chlorine for bacteria-related problems.

Thirty-four percent (281/816 samples) had measurable levels of manganese that exceeded the AO concentration of 50 µg L⁻¹ (Table 2). Twenty-two percent (178/816) of these sites with a manganese concentration greater than 50 µg L⁻¹ also had an iron concentration greater than 0.3 mg L⁻¹.
Eleven (1.3%) of the samples had levels of zinc that exceeded the aesthetic objectives of 5000 µg L\(^{-1}\) (Table 2). The greatest concentration measured was 17,606 µg L\(^{-1}\). The majority of these sites (9/11) with high zinc concentrations represent large diameter wells less than 30 m deep. These wells likely had galvanized well cribbing although this was not investigated further.

Although the GCDWQ do not currently include criteria for aluminum, a level of 200 µg L\(^{-1}\) has been proposed for water treatment plants (Health Canada 1999). Two percent (13/816 samples) had levels of aluminum higher than 200 µg L\(^{-1}\). Ten of the samples tested had concentrations between 200 and 700 µg L\(^{-1}\) and three samples had concentrations that exceeded 700 µg L\(^{-1}\).

**Household Water Testing**

Eighty-one percent of the participants indicated that they had tested their water at some time; however, 59% of all the participants had not tested their water in the last 5 years or tested it at all. A total of 40% of the respondents to the Farmstead Water Quality Survey indicated that they made use of some form of water treatment, primarily water softeners (22%), water distillers (13%), iron filters (12%), reverse osmosis systems (3%) or some form of disinfection (3%). The level of treatment and testing is of concern, especially in those cases where concentrations of any given parameter can cause health risks.

**Conclusions**

Of the 816 farm water wells surveyed, 32% exceeded the GCDWQ for MAC or IMAC of at least one parameter (including herbicide and coliform detections exceeding the MAC or IMAC). In addition, 92% of the sites exceeded the GCDWQ for at least one of the aesthetic objectives (AO). The majority of those objectives included TDS (85%), which had the highest number of samples exceeding the AO, followed by Na (64%), Fe (36%), and SO\(_4\) (19%) at levels higher than the GCDWQ. These parameters clearly reflect the aquifer geology from which the water was drawn. However, it should be noted that the guidelines provide a standard that is low enough so occasional exposure to water above the standard, or exposure to amounts that are only slightly above the standard, will not result in an adverse health reaction.

Fluoride, selenium, boron, uranium, lead, chromium and arsenic were found at levels above the GCDWQ MAC. Some of the higher parameter concentrations detected in the water samples tested (e.g., herbicides, coliform, nitrates, lead, zinc, chromium and aluminum) can likely be attributed to agricultural activities or to the household water distribution system. The majority of analyses with higher concentrations of most parameters (e.g., aluminum, arsenic, fluoride, iron, sodium, etc.) are due to natural geological conditions that are determined by the mineralogy of the source aquifers.
The evaluation of the FWQS data helps to identify some site-specific as well as some province-wide problems primarily related to natural geological conditions of the source aquifers, including F (13%), NO$_3$-N+ NO$_2$-N (6%), As (3%), Se (3%), Pb (2%), Al (2%), Ba (0.9%), Cr (0.2%) and U (0.4%). The effects of primary agriculture are likely limited to the 3% herbicide detections and to some nitrate and microbiological contaminations observed. The temporal effect in herbicide detections related to agronomic practice cannot be addressed with the FWQS data and hence needs to be to better identify cause and effect relationships.

Water at 32% of the sites that exceeded the MAC and IMAC for at least one component should not be used for human consumption without further investigation. Study results indicate a general lack of testing among rural residents of their water and that some rural residents may be “mistreating” their water, with improper systems for the quality indicated from tests. While the AO may not have a direct health effect, exceeding the AO may decrease the palatability of drinking water, increase staining of fixtures and laundry, or interfere with water treatment processes.

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References


Contamination of farmstead wells by pesticides, volatile organics, and inorganic
Swistock BR, Sharpe WE, Robillard PD. 1993. A survey of lead, nitrate and radon
contamination of private individual water systems in Pennsylvania. J.
SPSS Inc. 1998. SPSS Base 8.0. SPSS Inc. Chicago, IL. CD-ROM.
Warren CJ, Dudas MJ. 1992. Acidification adjacent to an elemental sulfur stock-
Weisenburger DD. 1990. Environmental epidemiology of non-Hodgkins lym-
Welch AH, Lico MS, Hughes JL. 1988. Arsenic in groundwater of the Western