CHAPTER 6

SUSTAINABILITY AND VULNERABILITY OF GROUNDWATER

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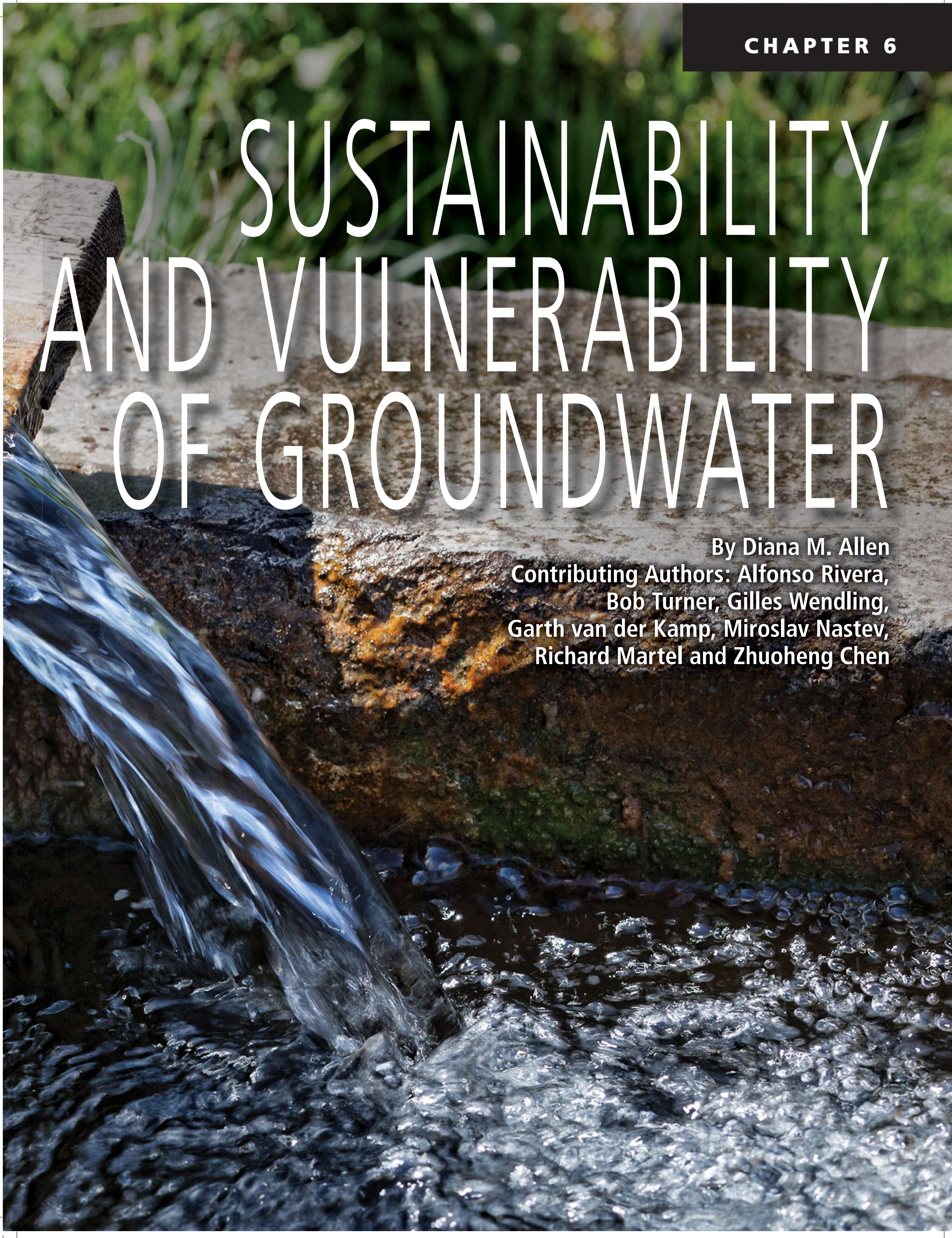


6.1 INTRODUCTION

Ecosystems along groundwater and surface water flowpaths have established a natural balance over a long period of time, because these systems are present in streams, lakes, wetlands, estuaries and along foreshores, the main conduits and discharge areas of fresh water. Any re-routing, extraction or contamination of water at locations along its natural flow path will affect all of those ecosystems which rely on water.

It is only in relatively recent geologic history that humans have created a strong footprint on this planet's natural environment. Early Man used readily accessible water sources, such as rivers, springs and lakes. The first man-made wells were dug in Mesopotamia and China to meet needs of sedentary population centres as well as those of agriculture. Extraction rates during these early times were limited because they relied largely on human and/or animal power, or pulleys.

During the industrial revolution, humans developed the capacity to lift water continuously and to drill deeper wells to meet the exponential growth in demand for fresh water. This growth was the result of population densification due to urbanization, coupled with the industrialization of agriculture and industry.

One significant result of the industrial revolution was a dramatic increase in the production and use of chemicals (for industrial processes, fertilizers, etc.). Production of both human and animal waste also increased, creating an urgent need for more efficient waste disposal methods. Waste disposal technologies, however, still are not significantly advanced to deal with the range of waste products being produced, and there was no conception of the fact that disposal of chemical and biological waste could be potentially detrimental to the environment.

Following the industrial revolution, impacts of continued population growth, mankind's often unsustainable consumption of fresh water, and our legacy of environmental pollution have forced us to seek new insights into the need for source protection for both the quality and quantity of our surface and groundwater resources. As surface water resources become more fully mapped, developed and appropriated (in the Okanagan and Southern Alberta, for example), groundwater remains the only other available source for new and future development.

Groundwater and surface water are closely related, and in many areas, the two may be said to comprise a single resource (Winter et al., 1998). Groundwater extraction, through pumping, can result in reduced river flows, low lake levels, and reduced discharges to wetlands and springs, causing concerns about drinking water supplies, riparian zones, and critical aquatic habitats (Alley et al., 1999).

Humans contribute to and experience the causes of global change processes created by unsustainable land use and climate change. There is an underlying need for better understanding of the interaction between humans and our environment, and this need becomes more urgent as changes in land use escalate in tune with human population growth. One recent development in this regard is the increased attention now being paid to the sustainable management of groundwater (and surface water) (Downing, 1998; Sophocleous, 1998).

Herewith we present our observations and conclusions concerning groundwater sustainability and vulnerability. Within the context of this chapter, sustainability relates to the quantity of groundwater available for sustainable use, while vulnerability relates to the groundwater quality.

6.2 GROUNDWATER DEVELOPMENT AND SUSTAINABILITY

6.2.1 How much water is available?

Groundwater dynamics and its hidden nature often impede the understanding and management of this vital resource. One important initial step toward sustainable groundwater resource management involves the exploration and understanding groundwater's hydrogeologic nature and mechanics.

Water moves constantly through the ground as a flux or flow. The flux occurs because water is added at certain times of the year, essentially to "top up" the aquifer within its recharge areas, causing an increase in the hydraulic potential. As a result, the water then begins to move into areas of lower hydraulic potential and, under natural conditions, eventually discharges to springs, streams, lakes and oceans. The flux mechanism can be thought of as a water slug added to the ground, which gradually makes its way to areas of discharge.

A certain amount of water is always "stored" in the ground at any given time. Some 30% of the total groundwater existent on this earth is estimated to be stored as groundwater. (Fresh water stored in rivers, lakes and as soil moisture amounts to less than 1% of the world's fresh water total, while polar ice and glaciers account for some 69%.)

Not all aquifers, however, have the same ability

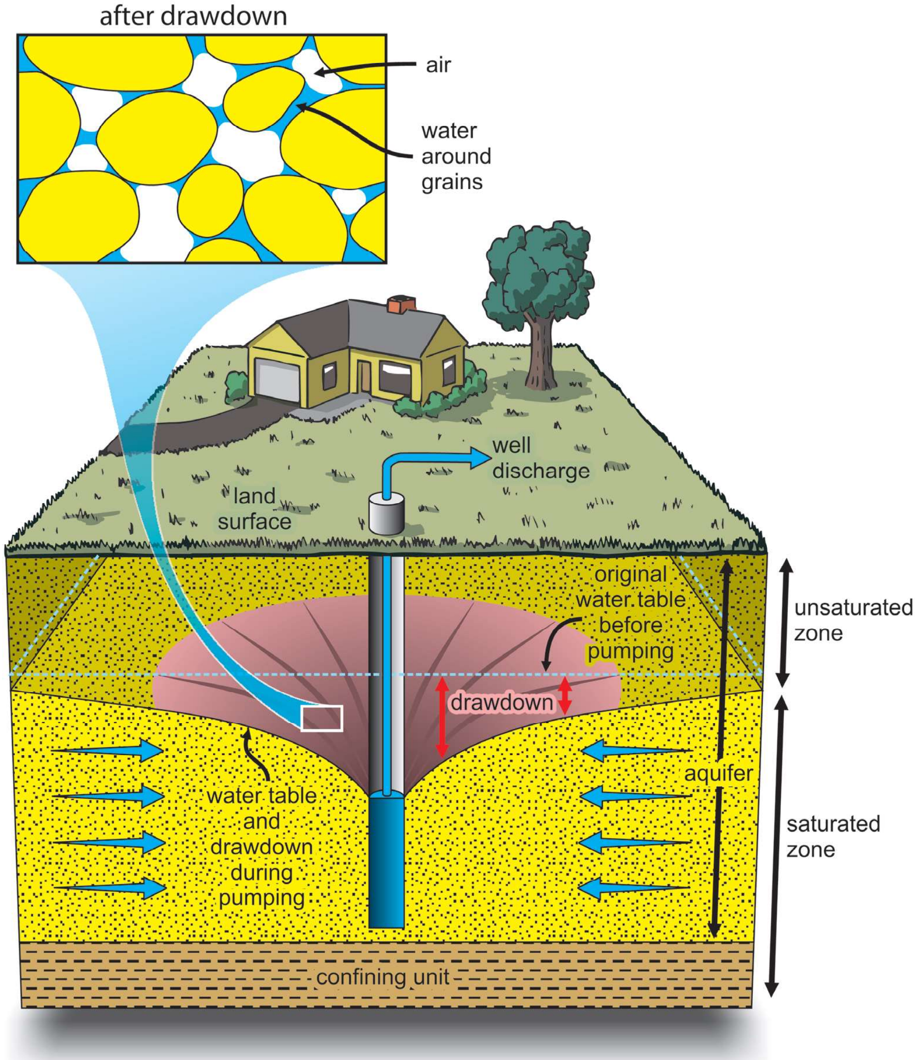


Figure 6.1 Unconfined aquifer cone of water table depression.

to store water. A general rule is the more porous the aquifer, the more void spaces exist for water storage. Sand and gravel aquifers, therefore, have a much higher storage potential than do aquifers of crystalline rock, although fractures can increase the porosity and, therefore, the overall rock aquifer storage capacity.

Whether an aquifer is confined or unconfined also plays a role in its ability to store, and relinquish, water. Figure 6.1 illustrates an unconfined aquifer and a pumping well. When an unconfined aquifer is pumped, water in storage releases from pore spaces at the water table, to be replaced with air. As a result, over time, the water table will drop, creating a cone of water table depression around the well. The pump itself acts to lower the hydraulic potential around the well screen, causing water from surrounding regions to begin travelling towards the well. As more and more water is pumped from the well, more and more outlying storage water moves in as replacement, and the cone of depression increases. In the case of confined aquifers, it is also appropriate to consider a cone of under-pressuring, which remains fully water saturated but with lower hydraulic potential closer to the well.

Pumping from a confined aquifer, one with a low permeability confining unit which overlies the aquifer, is quite different because the pore spaces never actually empty completely (i.e., no air fills the void, illustrated in Figure 6.2). Water in a confined aquifer is released from storage largely because the grains within the aquifer move closer together. In other words, the aquifer becomes more compacted as the groundwater pressure declines. The amount of movement is very small, particularly in rock aquifers, and leads to only a small release of water from storage, but, this amount can become substantial, over large areas.

An aquifer's high or low storage capacity is only one piece of the puzzle scientists use when determining how much water is available. To be capable of extracting groundwater for use, an aquifer must also be permeable enough to allow groundwater to move freely towards a well, replacing the volume of water being extracted.

Aquifers with low permeability tend to develop deeper cones of depression when compared to aquifers with high permeability. Aquifer recharge must also occur, otherwise the groundwater level continues to decline inevitably until pumping is no longer feasible. Groundwater discharge from the aquifer is also important for maintaining surface water supplies and ecosystems. Thus, it is the interplay of specific storage capabilities and the hydraulic conductivity, the nature of the aquifer, recharge and discharge, and, of course, pumping rate, that ultimately determine the groundwater amount available for use (see Chapter 2 for more details). Determining what amount can be used safely, without causing undue decline of groundwater levels and without harm to the ecosystems supported by natural groundwater discharge, offers a number of challenges to groundwater professionals and managers.

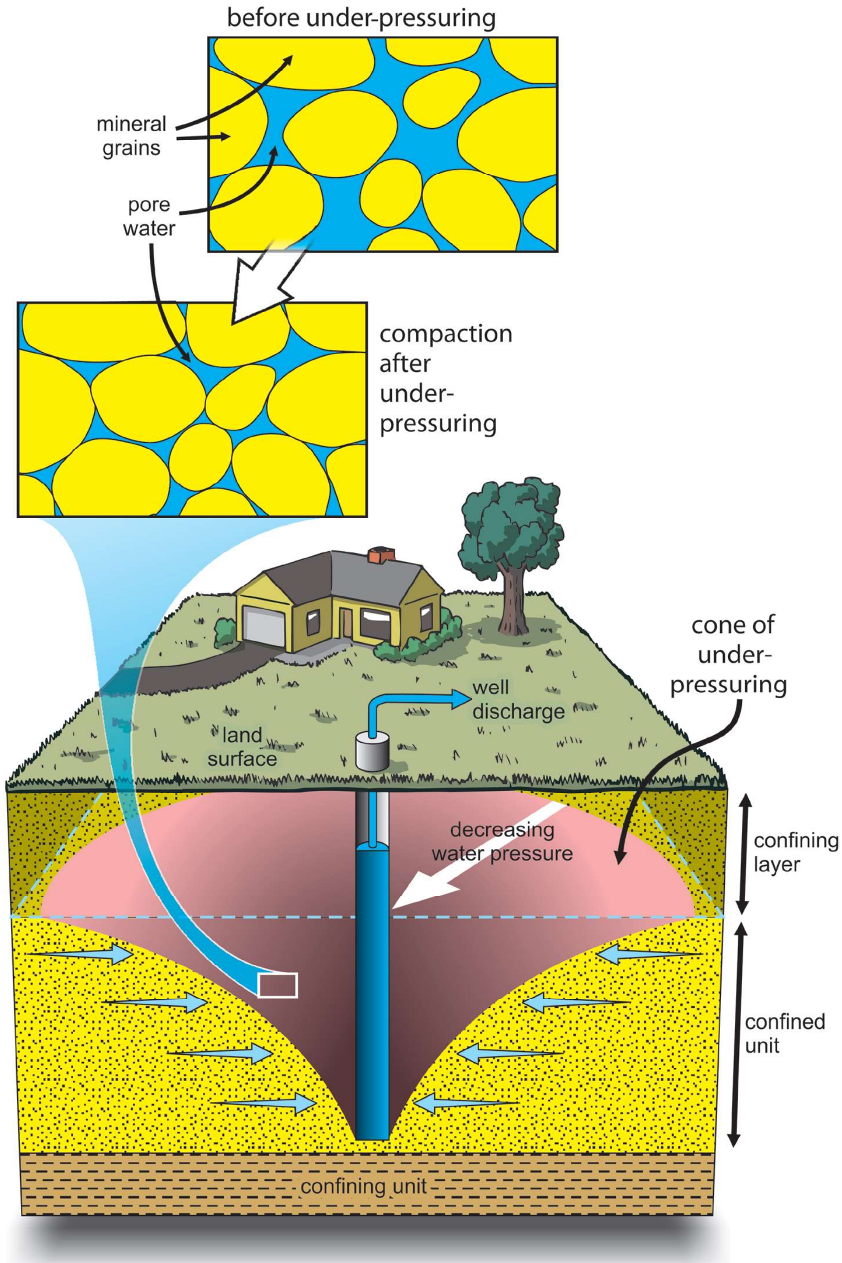


Figure 6.2 Confined aquifer: cone of under-pressuring.

6.2.2 What is sustainability?

Groundwater development can be considered sustainable when it is used in a manner that can be maintained for an indefinite time without causing unacceptable environmental, economic or social consequences (Alley et al., 1999). In practice, the process of determining "unacceptable consequences" is usually subjective, and can be narrowed down to a small number of constraints which must be satisfied: the drawdown in the pumping well should not exceed 70% of available drawdown, or baseflow in a nearby stream must be sustained during drought conditions. These constraints on groundwater development are established by regulatory agencies responsible for groundwater management, and usually in discussion with groundwater users and other stakeholders, such as wildlife agencies and watershed management groups. In the past, constraints, if any, have usually focused on the production wells themselves and, sometimes, the pumping impact on nearby groundwater users (e.g., Alberta's "Q_{20}" method; Maathuis and van der Kamp, 2006). As water resource use and land use continue to intensify, constraints on groundwater developments are beginning to be defined within the context of watershed and aquifer management plans; these adopt a much wider view, one which includes many other parameters, such as source water protection, to monitor and maintain water quality and groundwater-surface water interactions.

Groundwater sustainability must be defined within the context of the complete hydrologic system of which groundwater is a part (Alley et al., 1999). Groundwater pumping necessarily changes any pre-existing groundwater regime. As a result, most constraints are set in terms of limits on groundwater level changes and groundwater discharges to surface water. The dynamic response of the groundwater system to pumping is of over-riding importance.

Prediction of such response relies strongly on the observed behaviour of the groundwater system as a whole. The hydrogeological complexities and those of the recharge and discharge processes usually prohibit reliable prediction based solely on limited hydrogeology information. Systematic monitoring a groundwater system's response, including groundwater levels in observation wells, pumping rates, groundwater discharge to surface water (springs, baseflow of streams, lakes, etc.) is essential for informed groundwater management.

Ideally, this development should proceed in stages, guided by on-going monitoring; this process is generally referred to as an adaptive management approach. A complete assessment implies full maintenance and protection of the groundwater resource to balance ecomonic, social and environmental needs (Rivera, 2008).

One of the most important sustainability attributes is that it fosters a long-term perspective on groundwater resource management (Alley et al., 1999). A common misconception is that groundwater is renewed at the same rate each year, and that such renewals will continue to provide groundwater for use forever. Not true. Groundwater is perhaps best described as a semi-renewable resource. It is renewable in the sense that precipitation replenishes the amount of storage water annually, but non-renewable insofar as the amount replenished and time for replenishment varies yearly. As a result, groundwater stored in aquifers varies from year to year, owing to annual fluctuations in precipitation (and recharge) (Figure 6.3). Precipitation amounts and temperatures can range considerably above (e.g., large storms, snow pack)and below average[1] values.

1. Average precipitation is based on what has been measured at climate stations in a watershed over a certain period of time. In Canada, these are commonly referred to as 30-year climate normals, which describe the average climatic conditions of a particular location. At the completion of each decade, Environment Canada updates its climate normals for as many locations and as many climatic characteristics as possible. The climate normals and extremes are based on Canadian climate stations with at least 15 years of data. Data between 1971 and 2000 are used in this chapter (Environment Canada, 2006).

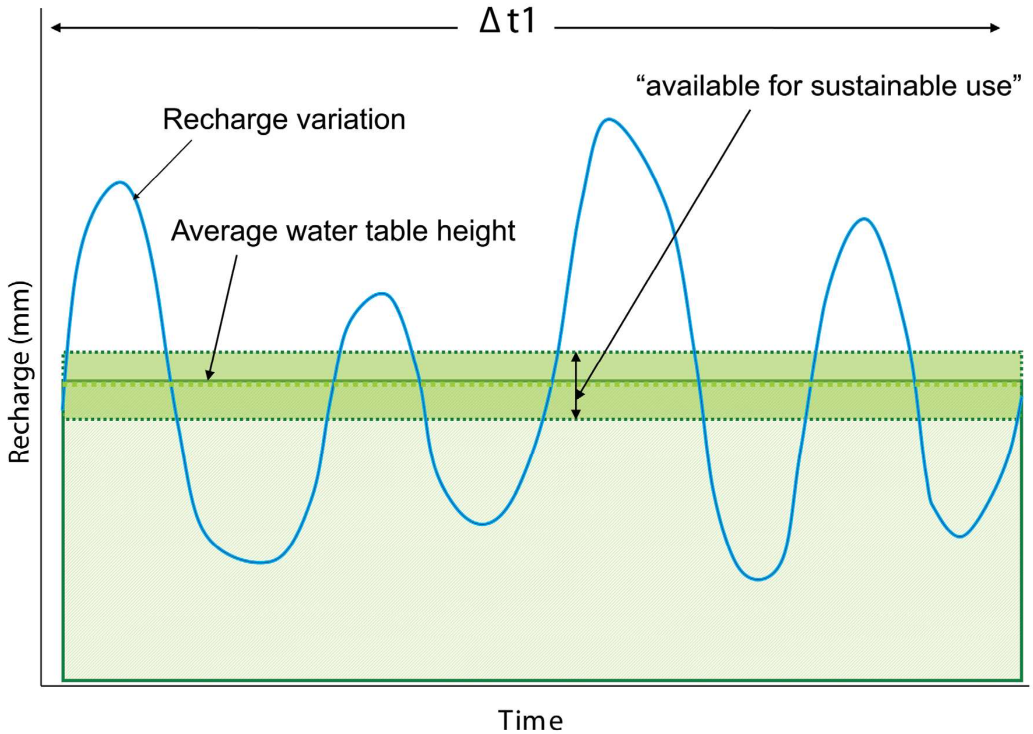


Figure 6.3 Variations in recharge over a certain time period (Δt_1), and the concept of the amount of groundwater available for sustainable use as a fraction of available recharge.

Another common misconception is that if water extraction equals recharge, there will be no impact, and extraction can continue forever. In reality, extraction is only a portion of the stored groundwater and this amount, reflects on only a small portion of the recharge added each year (Figure 6.3). When groundwater recharge is lower, so too is the water table. The effects of this lowering are evidenced throughout the entire aquifer system, ultimately resulting in decreased stream baseflow levels during the low flow season when groundwater is the dominant source of stream water. Although the contribution of groundwater to total streamflow varies widely among streams, hydrologists estimate the average contribution ranges between 40 and 50% in small and medium streams (Alley et al., 1999). During droughts, when other water sources dry up, groundwater flow to streams is especially important for maintaining aquatic ecosystems, including fish populations.

Artificially lowering the water table, through pumping for example, can reduce or even reverse groundwater flow to streams with consequent negative impacts on streamflow. But effects of groundwater pumping are often slow to manifest themselves. Because groundwater development may take place over many years, both current and future development should be taken into consideration in any water management strategy, along with information on climate variability and climate change.

When trying to determine how much groundwater is available for sustainable use, we need to consider the period for which we have collected data. Figure 6.4 shows us that information provided through the time period Δt_1 will be different from that provided for the longer time period Δt_2. Sustainability estimates based on data collected during the period Δt_1 may grossly overestimate the available groundwater. We must also

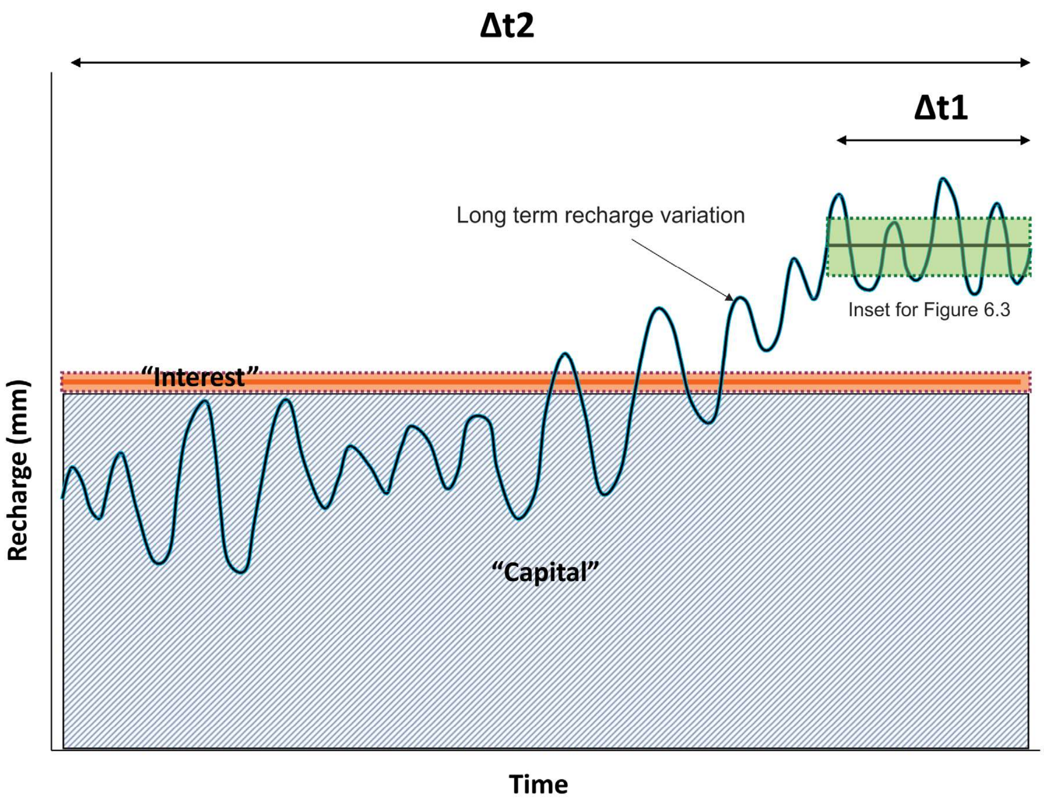


Figure 6.4 Recharge varies over short- (see Figure 6.3 corresponding to Δt_1 time period) and long-time scales (corresponding to Δt_2 time period in this figure). If the sustainability of an aquifer is determined based on a data set of limited length, this may not provide an accurate measure of available water. Removal of groundwater from an aquifer system should be viewed as using the "interest" rather that exploiting the "capital".

consider the time scale at which the groundwater moves through a specific aquifer. For a given time period (Δt_2), there will be a volume of water that corresponds to the recharge "excess", or, in other words, the volume of water for which removal will not translate into large ecosystem disturbances. Using the financial analogy of "capital" and "interest", we are then using the interest in such cases, and not the capital, although in times of temporary water scarcity (e.g., in semiarid and arid regions; Dragoni and Sukhija, 2008), it may be necessary to tap into the groundwater located in long-term storage ("capital" in Figure 6.4). This is a management decision, because the practice is not sustainable over long periods.

A number of definitions regarding groundwater sustainability have been proposed over the years. Discussions of "safe yield", "sustainable yield" and similar concepts have created considerable confusion, resulting in disagreements over definitions as well as some serious misconceptions (Devlin and Sophocleous, 2005; Alley and Leaky, 2004; Bredehoeft, 1997; Bredehoeft, 2002; Wood, 2001). Regardless of which term is used, sustainable or safe groundwater development (extraction) means there must be no unacceptable consequences. Any definition needs to define the time period involved and the specific unacceptable consequences.

A general definition of sustainable yield is often made in terms of specific constraints. These might include, for example, the fact that the withdrawal rate should not exceed some fraction of the recharge rate, or that the groundwater system must come to a new and acceptable steady-state condition. Such constraints may be useful for particular groundwater developments, but it is a mistake to accept them as generally applicable overall. Recharge rates in many cases are quite irrelevant to the setting of constraints.

Most groundwater in Canada's Prairie regions occurs in complex heterogeneous aquifer systems which have little or no relation to local surface watershed boundaries. Additionally, a large proportion of groundwater discharge occurs through evapotranspiration in diffuse areas; the recharge/discharge processes are highly variable both in time and space. Thus, groundwater monitoring relies primarily on observation well records and the reporting of withdrawal rates. With the exception of a few isolated surficial aquifers, estimates of recharge and discharge rates have high uncertainty and are of limited relevance to sustainable groundwater management for the majority of Prairie aquifers (van der Kamp and Maathuis, 2006). There is rarely a need to specify conditions wherein a new steady state must be reached, because these groundwater systems may exist in a transient state for the far foreseeable future without any observable unacceptable consequences.

Transboundary aquifers require additional constraints on groundwater development, including, principally, the need to harmonize regulatory requirements of different jurisdictions and the need to plan for an equitable usage of groundwater resources (see Chapter 16). Recently, the consequences of over-pumping have become clear in the Great Lakes region, and the U.S. Geological Survey issued a "wake-up" report. This report illustrated the fact that over-pumping on the U.S. side of the Great Lakes Basin can cause groundwater to flow away from major bodies of water rather than into them (USGS, 2005 as cited by Nowlan, 2007). No similar research results have been reported on the Canadian side.

6.2.3 Groundwater development: The risk or consequence factor

Groundwater development has potentially far-reaching consequences—beyond those of causing a decline in water level and/or pressure around a pumping well (see Figures 6.1 and 6.2).

Adjacent pumping wells have a zone of influence that represents the cumulative effects of pumping (the cones of depression coalesce (Figure 6.5a). This phenomenon not only results in a greater, and more widespread, lowering of the water table or potentiometric surface, but can also lead to issues of well interference between well users. The cumulative impacts of all nearby wells should be taken into consideration when predicting the sustainability of an aquifer or even a particular well; however, the extent to which this is done in Canada is open to question.

Maathuis and van der Kamp (2006) stated that all jurisdictions within Canada include an assessment of potential impacts on other users, and on surface water, as part of the licensing procedure. Criteria for evaluating such impacts, however, vary widely from one jurisdiction to another and are, at best, only vaguely defined. Furthermore, Maathuis and van der Kamp indicated that it is not clear how the cumulative impacts of many groundwater users are addressed, or whether they are addressed at all. In Nova Scotia, for example, withdrawals from the aquifer must be sustainable[2], new groundwater withdrawals should not cause any significant adverse effects to existing groundwater users, and groundwater allocations are based on a first-come,

2. In this case, sustainable is defined as not causing unacceptable environmental, economic or social consequences.

first-served[3] basis, with priority given to drinking water applications (Nova Scotia, *Environment Act*).

Ontario issues permits for groundwater withdrawals over 50,000 L/day excepting domestic or traditional agricultural use[4], and a determination of any potential interference with existing groundwater users is required (Ontario, *Water Resources Act*).

Alberta legislation takes a number of considerations into account when granting new water allocations; these include protecting the aquifer from over-development, protecting household water supplies and those of prior license holders, and protecting the environment (Alberta Environment, 2003).

Similarly, in Saskatchewan and Manitoba, a Groundwater Investigation Report (Saskatchewan Watershed Authority, 1999) and Groundwater Exploration Permit (Manitoba Water Rights Act, 1998), respectively, must be submitted for new applications, including an evaluation of the impact of any project on surrounding users.

British Columbia does not require licensing of groundwater, and well owners have no recourse when a new well is constructed and subsequently interferes with an original well. Recent amendments to the BC Water Act provide for licensing in specific areas designated as Water Management Areas. Otherwise, only private water utilities using groundwater as a water supply source require certificates of Public Convenience and Necessity, which, in turn, requires a groundwater evaluation for the well in question (British Columbia, *Water Utility Act*).

When a well is situated near a stream or lake, the lowering of groundwater levels around that well during pumping may impact surface water levels (Figure 6.5b). Situating a well near a surface water body can effectively draw water from that body, although the effects of pumping may not be as noticeable or as detrimental during high runoff periods (e.g., spring and early summer) as they are during periods of low flow (e.g., late summer). Wells located near streams or lakes may be also more prone to contamination by microbial pathogens originating in these surface waters.

The term GUDI, or Groundwater Under the Direct Influence of surface water, is used to refer to groundwater sources (wells, springs, infiltration galleries, etc.) that may lie in close connection to surface waters (e.g., AWWA, 1996, 2001; Ontario Ministry of Environment, 2001; Nova Scotia Department of Labour, 2001; U.S. EPA, 1991a; U.S. EPA, 2001). A number of criteria exist to identify GUDI sources, including whether the source is in a sensitive setting (i.e., a well in an unconfined aquifer, a spring, a well in a karst aquifer, etc.), and the source's proximity to surface water (the source must be located at a sufficient distance away from the surface water body to minimize risk of contamination). The well must be properly constructed, and available chemical and microbiological data must show the raw well water does not regularly or periodically contain bacteria. GUDI groundwater supplies pose a public health risk and must be identified and carefully managed.

Groundwater development over large areas can lead to significant regional lowering of groundwater levels (Figure 6.5c). Fortunately, unsustainable groundwater development has not been a major problem in Canada. Aquifer water levels, for the most part, appear to be relatively stable across

3. The "first-in-time, first-in-right" doctrine is common to other jurisdictions across Canada (except Saskatchewan) and elsewhere around the world.
4. Non-traditional crops are commonly thought of as low acreage, niche crops such as ethnic fruits and vegetables, culinary and medicinal herbs, and plants for industrial uses (e.g., fibre hemp). A non-traditional crop may be new to a region or simply new to the grower.

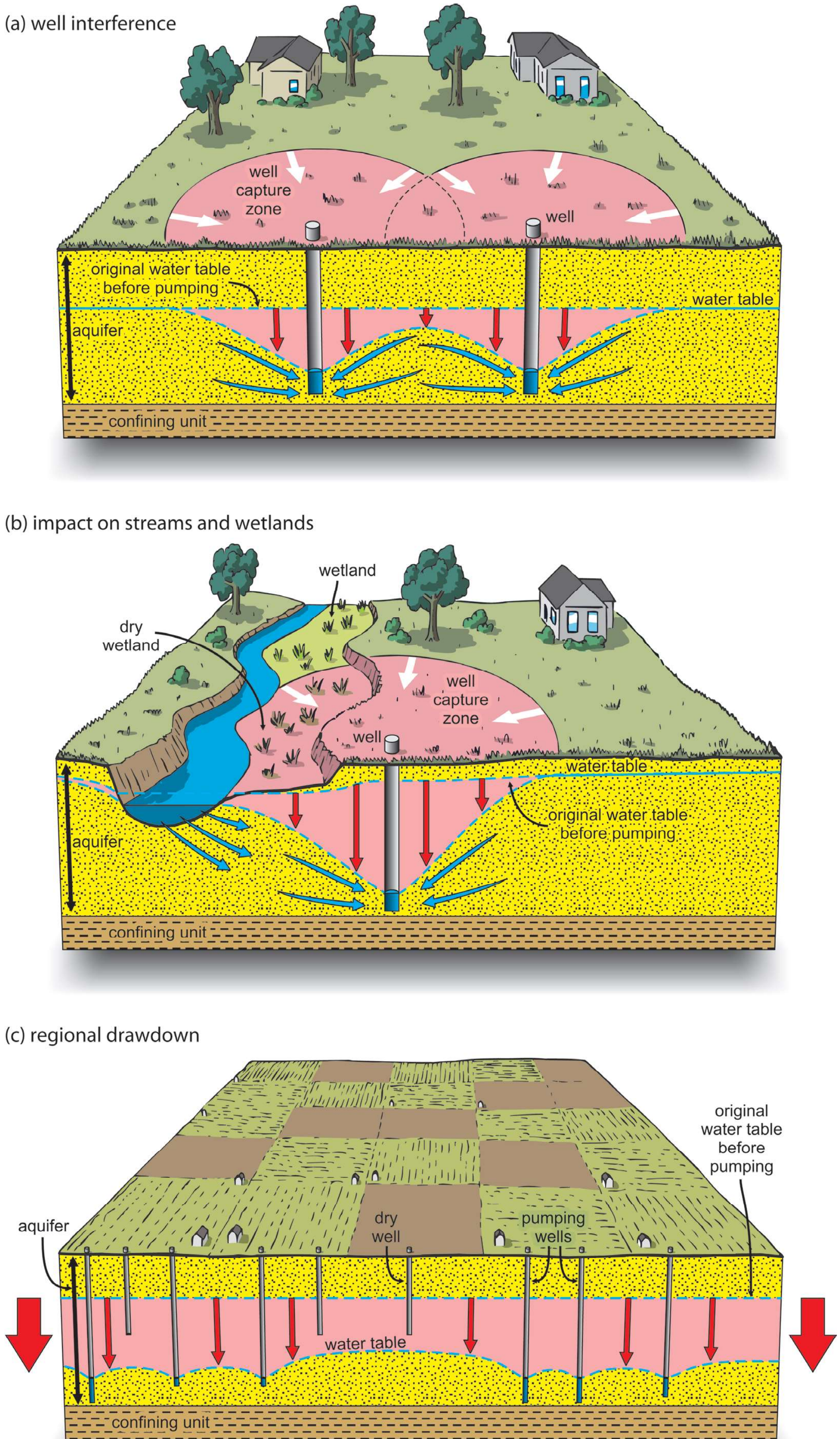


Figure 6.5 Impact of declining water levels due to: (a) well interference, (b) impact on streams and wetlands, (c) regional drawdown.

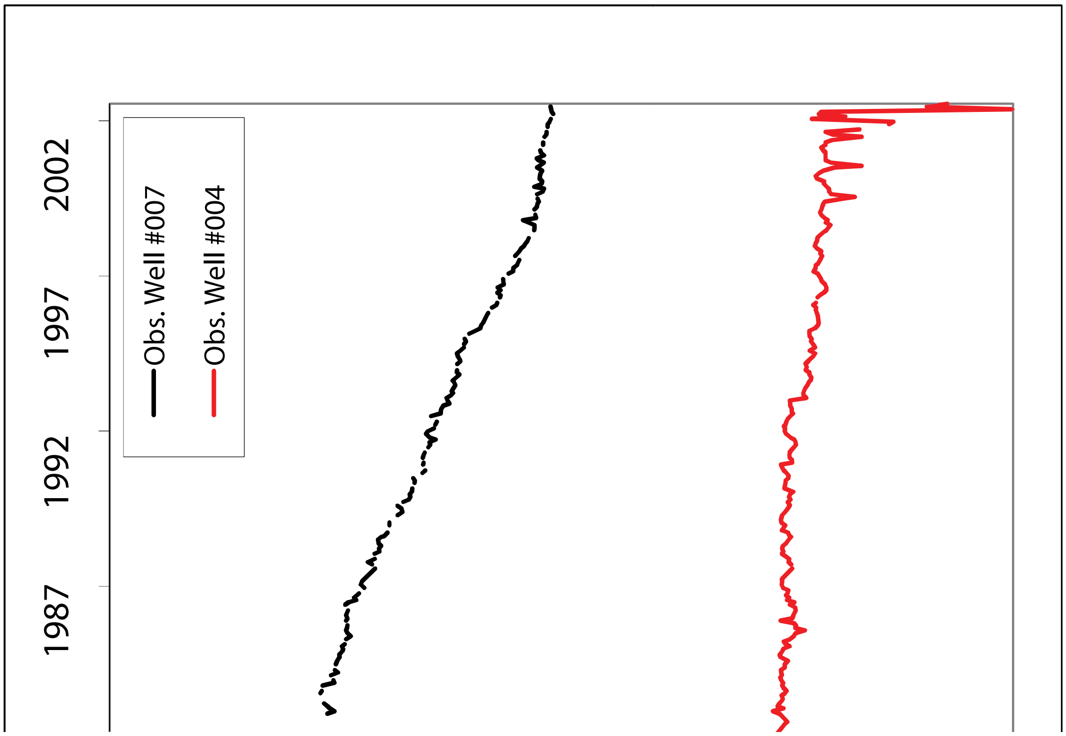


Figure 6.6 Indications from two BC provincial observation wells #004 and # 007 of a long-term declining trend in groundwater level within the Langley region of the Lower Fraser Valley.

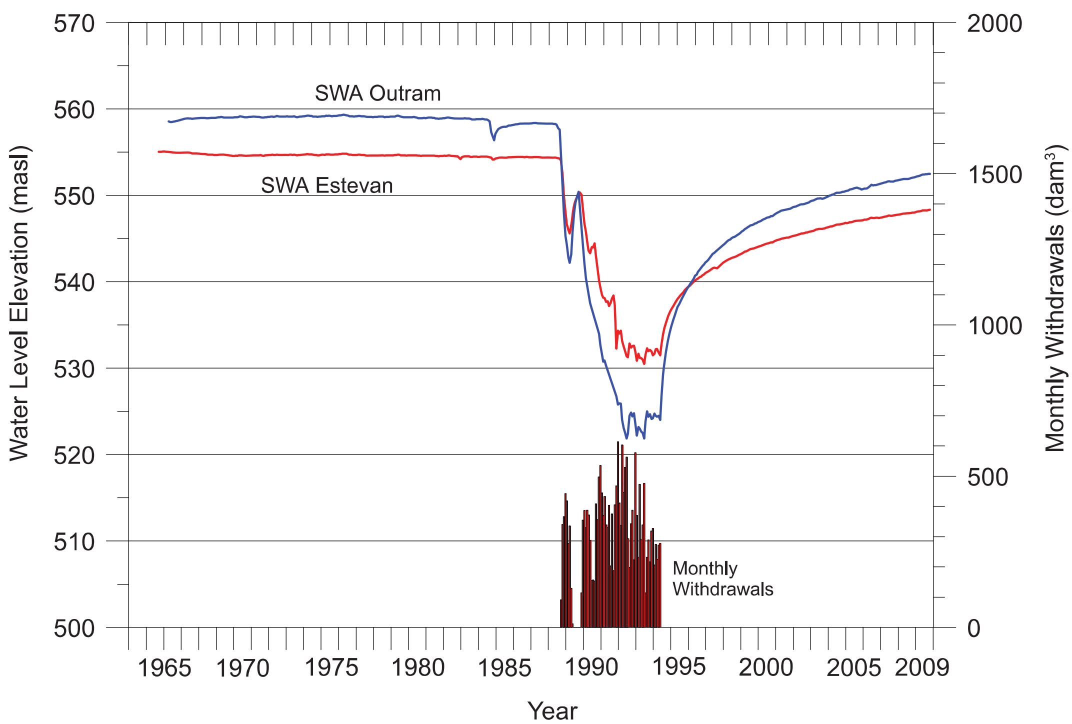


Figure 6.7 Total monthly withdrawals for the Estevan aquifer production wells and water level data for the Outram and Estevan observation wells (7.3 km and 8.6 km respectively distant from the well field). Total monthly withdrawal in cubic decametres (dam^3) (With permission from Saskatchewan Watershed Authority).

most of the country. In those areas where there has been significant development, however, scientists have discovered early signs of declining water tables. Groundwater levels in the Langley area of the Lower Fraser Valley, for example, have declined by over 1.5 metres since 1980 (Figure 6.6).

Southwestern Saskatchewan's Estevan Valley aquifer system is one of the major preglacial buried valley aquifers in that province (van der Kamp and Maathuis, 2006). The system is unique in that it has been the subject of groundwater resources evaluations since the early 1960s. Withdrawal data for production wells and water level data for two observation wells situated 7.3 and 8.6 km away from the well field (Figure 6.7), respectively, indicate that recovery of water levels due to over-pumping will take several decades. The pumping's zone of influence extends tens of kilometres, with drawdown of up to 13 m observed 35 km from the well field. At the Canada-U.S. border, 23 km from the well field, maximum drawdown was 20 m. The zone of influence extends into the U.S. portion of the aquifer, but apparently these effects did not translate into limitations on the withdrawal rate (see Box 10-2 for more details on this case).

Recently, the consequences of over-pumping have become clear in the Great Lakes region (USGS, 2005), when a computer modelling study calculated the differences in groundwater flow directions between the late 19th century and the year 2000. Under pre-development conditions, the model indicated that 1.9 million U.S. gallons per day (7,192 m^3/day) of groundwater flowed upward from the deep to the shallow part of the flow system, and into Lake Michigan, over the area of the Lake between southeastern Wisconsin and Michigan. Under year 2000 simulation, the model simulated that 1.3 million U.S. gallons per day, most of which originates as lake water, flows vertically in the opposite direction. While the loss of water from the lake is small relative to the total lake volume, model results indicate that development may have a significant impact on the hydrologic cycle.

Groundwater pumping can cause more than a decline in water levels or reductions in the discharge to streams, lakes and wetlands. As water is pumped from an aquifer, the aquifer's ambient pressure is reduced. Any water pressure reduction, whether accompanied by an increment of the overburden load or not, produces an increase in effective stress in the solid matrix. This increase results in solid matrix deformation, which manifests itself as compaction, leading, in turn, to observable land subsidence. We are lucky in Canada because widespread land subsidence has not been observed in any region to date. Several cases of significant regional land subsidence attributed to groundwater pumping have been documented around the world. These include the San Joaquin Valley and the Houston-Galveston area (U.S.), Bangkok (Thailand), Venice and Ravenna (Italy), and Mexico City (Mexico).

Groundwater quality may also be impacted by declining water levels. Usually occurrences of degraded water quality are related to salinity increases. Studies of Winnipeg's regional carbonate aquifer have revealed that fresh groundwater southwest of the city helps to prevent saline groundwater on the west side of the Red River from migrating eastward. This saline/fresh water boundary is strongly controlled by river systems (Charron, 1965; Grasby and Betcher, 2002). Charron (1965) demonstrated that this boundary was west of the Red River and south of the Assiniboine River in the early 1900s. He also

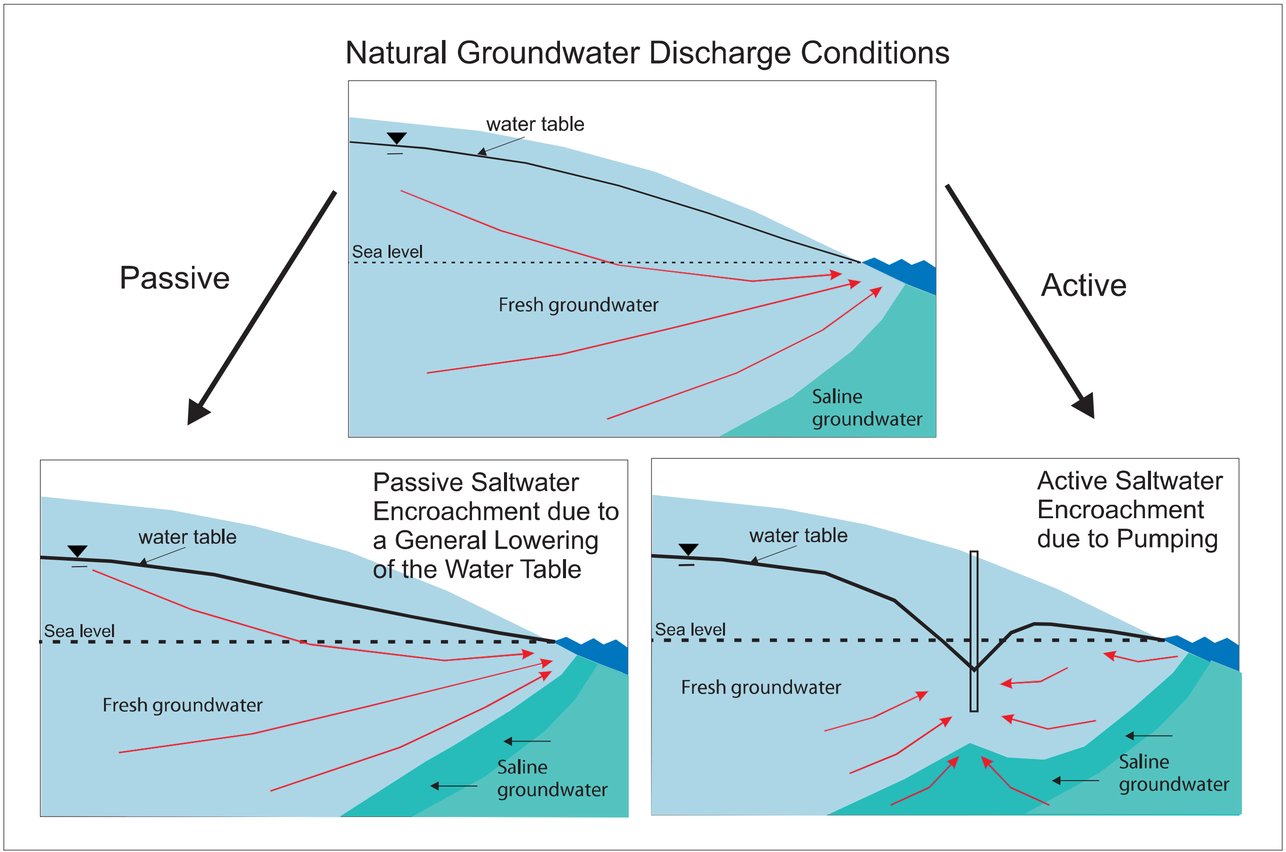


Figure 6.8 Active and passive saltwater intrusion relative to natural groundwater discharge conditions in coastal aquifers.

suggested that heavy pumping in the freshwater zone during the early 1900s caused saline water to move beneath the rivers into freshwater zones within the Winnipeg area. Eastward movement of the boundary was also observed in response to dewatering during construction of the Winnipeg floodway (Render, 1970). Pumping decreases over the last 30 years have resulted in the boundary moving back to its previous position.

In coastal regions, the incursion of seawater into freshwater aquifers (commonly referred to as saltwater intrusion) can be a negative consequence of groundwater extraction and/or development. When groundwater is pumped from aquifers that are in hydraulic connection with the sea, the induced gradients may cause a migration of salt water from the sea towards the well (active saltwater intrusion) (Figure 6.8). Long-term lowering of the water table due to declines in groundwater recharge may similarly result in saltwater encroachment (passive saltwater intrusion) (Figure 6.8).

Coastal aquifers in Canada have not yet been severely impacted by saltwater intrusion, although Prince Edward Island on the East Coast, and the Gulf Islands on the West Coast are examples of two regions where saltwater intrusion is a concern. The key to controlling any saltwater intrusion problem is to maintain a proper balance between water being pumped from the aquifer and the water amount recharging it. Constant monitoring of the saltwater interface is necessary to determine proper control measures.

One complicating factor in all cases of groundwater development is that its impact may not be noticed for many decades. Because development often occurs gradually (adding pumping wells as needed over time), its cumulative effects on water levels may be difficult to distinguish from natural groundwater level variations due to climate variability.

Identifying anthropogenic factors that cause water level declines is not straightforward.

6.2.4 Determining groundwater quantity sustainability

Historically, groundwater sustainability has been an issue for the individual well owner, who might want to have some assurance that the groundwater supply will continue indefinitely. Consequently, many of the traditional methods for groundwater sustainability assessment have focused on the well itself, and these evaluations have been carried out on a well by well basis.

Increased well development and concentration in particular regions has resulted in interference between wells, lowering of groundwater levels and streamflows. These problems have highlighted the need for new methods of determining groundwater sustainability, at a much broader scale.

Groundwater is but one component of the hydrologic cycle, and it is not possible to measure groundwater sustainability without considering how that groundwater resource is connected to surface water and land use.

Methods used to determine groundwater sustainability vary considerably because of the different assessment scales employed: these include the watershed scale, the aquifer scale, and the well scale (not necessarily listed in size order). Regional aquifers may span several watersheds, and one particular watershed may encompass multiple aquifers.

6.2.4.1 Well sustainability

Most methods used to determine groundwater sustainability at the well scale rely on pumping test, which are typically conducted following drilling to determine the appropriate pump size for the well. If the test is long enough, the data (water level decline in the well or drawdown as a function of time) can be extrapolated to estimate what the groundwater level will be at some later date. This approach has led to two methods widely used in specific regions of Canada: the Q_{20} method (in the Prairie Provinces) and the 100-day method (in BC).

Q_{20} Method: The Q_{20} concept (Farvolden, 1959) is based on a theoretical model for the flow of groundwater to a well in a confined aquifer (Theis, 1935; Jacob, 1940). It assumes that if the well is to last 20 years, the drawdown curve from a pumping test must not have a greater drawdown than what would be available where it intersects the 20 year line on a time scale. Farvolden (1959) considered two possible cases. The first is one where a stable pumping level is established in the well, indicating that recharge balances discharge. In this case, the specific capacity of the well can be calculated:

$$\text{Specific capacity } (C_s) = \text{Pumping rate / stable drawdown} \qquad (6.1)$$

The capacity of the well can be determined by taking into account a factor of safety:

$$\text{Capacity of the well} = C_s \times \text{available drawdown (H)} \times \text{safety factor (0.7)} \qquad (6.2)$$

where the available drawdown is the distance from the static level in the well to the top of the confined aquifer.

The second case considered by Farvolden is one where the water level continues to decline, indicating that the well is drawing water from storage. This case relates to the traditional Q_{20} method, in that the drawdown after 20 years of pumping is used to determine the well's sustainable yield. This long-term drawdown is determined either by extrapolating the log-log drawdown versus time curve, or by extrapolating the semi-log drawdown versus time line, following the methodologies of Theis (1935) or Jacob (1940), respectively. Farvolden (1959) defined the safe yield of a well as:

$$Q_{20} = (4 \times p \times T \times (H_A/8) \times S_F) / 2.3 \qquad (6.3a)$$

or

$$Q_{20} = 0.683\ T\ H_A\ S_f \qquad (6.3b)$$

where T is the transmissivity of the aquifer, H_A is the available drawdown (depth to the top of the aquifer minus the depth to the static level), and S_f is the factor of safety, for which Farvolden used 0.7.

The safety factor in both of these cases was chosen arbitrarily by Farvolden, and can be considered to represent other factors, such as well inefficiency, that may affect the available drawdown. Although Farvolden introduced this Q_{20} method, he did not explicitly use the notation Q_{20}: this notation was introduced by Tóth (1966).

Maathuis and van der Kamp (2006) review several other methods similar to the original Q_{20} procedure. Those take into account other conditions, such as short-duration pumping tests, to provide apparent safe yield estimates (Moell and Schnurr, 1976), consideration of well losses (Moell, 1975), and the use of local and regional transmissivity estimates (Bibby, 1979). Maathuis and van der Kamp (2006) propose a modification to the Q_{20} method wherein extrapolation of the drawdown curve is not necessarily that of the Theis or Jacob curve, but rather any appropriate drawdown curve for the aquifer (i.e., other analytical models that best represent the drawdown in the aquifer). Incorporating a factor of safety of 0.7, they suggest the following equation:

$$Q_{20} = 0.70 \times H_A \times Q_t / S_{20,\,Qt} \qquad (6.4)$$

where H_A is the available drawdown, Q_t is the rate of pumping used during the test, and $s_{20,\,Qt}$ is the estimated drawdown after 20 years of pumping at the pumping rate Q_t, calculated on the basis of the most appropriate aquifer model. This approach effectively allows for well losses affecting local drawdown (measured early in a pumping test), and establishes the long-term pumping rate based on the aquifer's response to pumping using the most appropriate aquifer model. A simplified equation can be derived from Equation 6.4:

$$Q_{20} = S_f \times H_A \times Q_t / [s_{100\,min} + (s_{20\,yrs} - s_{100\,mins})_{theoretical}] \qquad (6.5)$$

which uses the estimated drawdown after 100 minutes, $s_{100,\,Qt}$.

Maathuis and van der Kamp (2006) have also proposed the concept of R_{20}, the radius of influence of a well after 20 years of pumping. Radius of influence is the distance from the pumping well where the drawdown S_{R20} occurs after 20 years of pumping at the desired rate. Q_r equals some given limit, which might be set at about equal to the natural fluctuation of the water level in the aquifer, say 0.5m. This radius can be determined, using either the Theis or Cooper equations, by setting the drawdown to 0.5m, and calculating the

radius at which that drawdown occurs. This allows calculation of a pumping well's potential zone of influence, and determination of which other wells or surface water bodies might be affected by the pumping.

100-Day Method: The 100-day method is recommended for determining long-term capacity (sustainability) of a water supply source well in British Columbia. The 100 day time frame is used because most B.C. aquifers are considered to recharge within 100 days. Guidelines for this procedure suggest conducting a step-drawdown test to determine an optimum pumping rate, followed by a longer-term pumping test. The long-term capacity of the well is based on extrapolating the drawdown at the end of the test to 100 days, and using the 100-day's drawdown to determine the well's long-term specific capacity. A safety factor of 30 percent (0.30) is applied. Other conditions such as well interference, surface water–groundwater interactions, water quality and seawater encroachment may impact the well's long-term capacity, but are not explicitly taken into account with this method.

In fact, none of the above methods considers the influence of multiple pumping wells on the aquifer's sustainability. Usually when the pumping rates are relatively low, and the aquifer has low transmissivity, the pumping impacts from one additional well are not likely to extend more than a few kilometres from the proposed well site. Cases involving high pumping rates and large drawdowns within confined high transmissivity aquifers may require extensive mitigative actions (Maathuis and van der Kamp, 2006).

The cumulative effects of multiple pumping

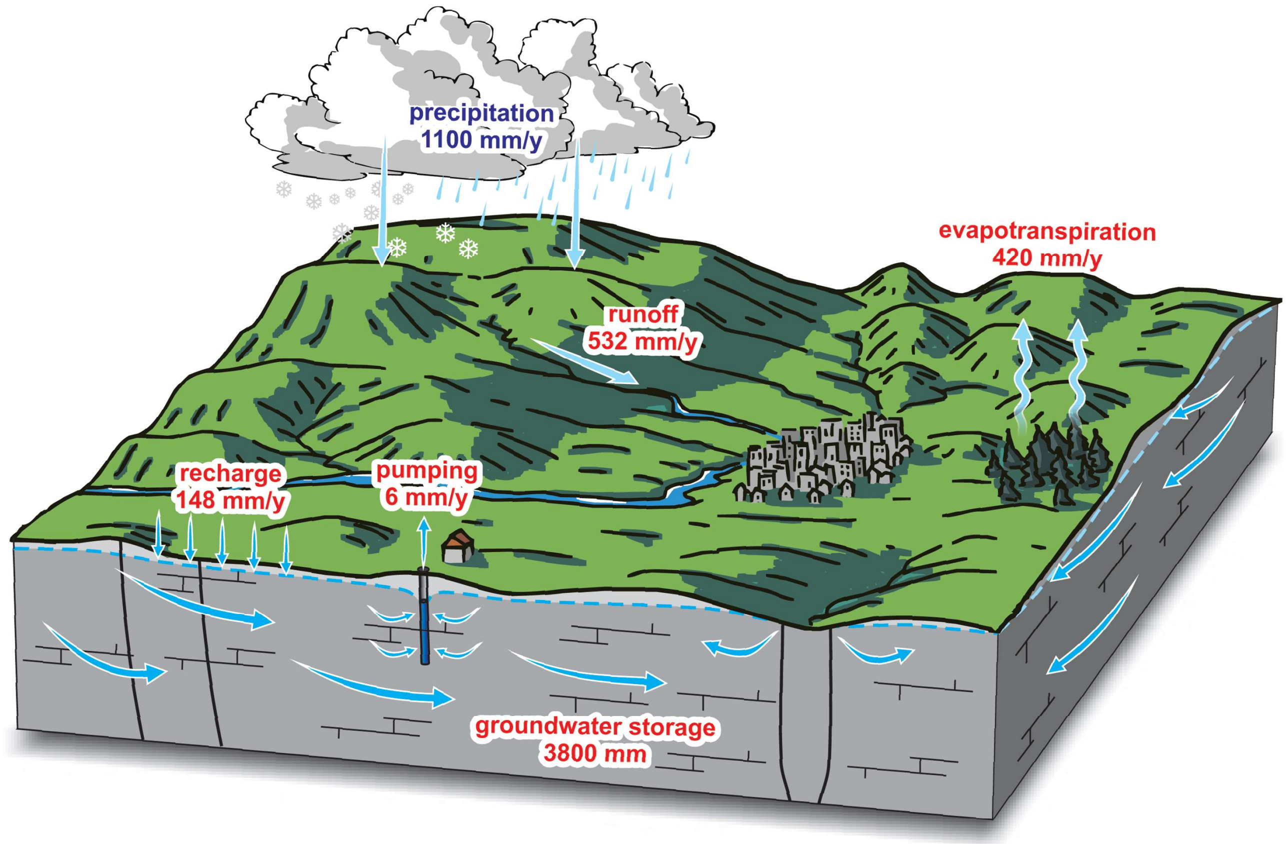


Figure 6.9 Elements of a regional water budget that can be used to assess sustainability.

wells in the same region create a composite cone of depression with a much larger radius of influence. In those areas where groundwater is being increasingly exploited, there is the risk that numerous small withdrawals from individual wells, each of which appears to have little impact, can add up to major consequences for the aquifer as a whole. Any estimation of an aquifer's (or watershed's) long-term sustainability must take into account all the groundwater uses and balance these with the aquifer's natural water budget.

6.2.4.2 Aquifer sustainability

Determining an aquifer's sustainability requires a comprehensive water budget analysis, which should result in an estimate of the net groundwater availability. Net groundwater availability is the difference between water inflow to the aquifer from precipitation, artificial recharge, surface water, etc., and the net water losses from the aquifer through evapotranspiration, pumping, discharge to surface water bodies, etc.

Elements within an annual water budget include precipitation, evapotranspiration, runoff, recharge and pumping (Figure 6.9). The budget assumes no changes in storage on an annual basis. As a result, it does not include storage in surface water bodies, storage in aquifers or storage in soil water.

Aquifer water budgets are often difficult to quantify from a physical processes perspective because of the large uncertainties in each budget components. Many aquifers extend over large areas. Precipitation is not uniformly distributed. The soils, subsurface geology, topography, land use/land cover, etc. are all spatially variable and thus, recharge is spatially variable (see Chapter 4). Evapotranspiration is a very difficult parameter to quantify both spatially and temporally. Additionally, there is limited information for most of the country on the amount of water actually pumped from aquifers on an annual basis. Another problem is that aquifer boundaries and surface watershed boundaries rarely coincide. Watersheds represent the most logical basis for managing water resources because watersheds are defined by topography. Water budget analysis on a watershed basis, however, may not include all parts of an aquifer, or there may be multiple aquifers within a single watershed.

Pressing concerns about water availability in some areas (the Prairies and B.C.'s Okanagan valley, for example) demand that comprehensive water budgets be developed for major aquifer systems, watersheds, or basins, despite the uncertainties and complexities outlined above. One of the most urgent resource management concerns is ensuring adequate in-stream flows for the ecosystems they support. This is not a trivial problem: it requires a full accounting of all water inflows and outflows at different times throughout the year.

Box 6-1 illustrates how numerical modelling is used to estimate the various water budget components for the Mirabel and Chateauguay aquifers in the St. Lawrence Lowlands. Two numerical models were developed to assess regional groundwater sustainability. The same methodology was applied in both cases: drawdown was obtained for different pumping rates applied uniformly over the study area, and the average predicted drawdown was used to estimate sustainability.

Developing regional scale models of aquifers (or watersheds) requires considerable data. The aquifers must be fully characterized with regard to area and depth, and in the geometry or architecture of the various geologic layers. Identification of natural hydraulic boundaries (rivers, streams, lakes) is important, as is quantifying aquifer recharge through direct precipitation and indirect contribution from surface water bodies, quantification of groundwater discharge to surface water bodies, and proper estimation of the pumping amount water removed from the system. In rural areas, some water is returned to the aquifer via septic system effluent, and in agricultural areas irrigation return flow (that amount of water not used by the crops) adds to the net recharge.

All water budget components are extremely difficult to quantify. Additionally, an overall water budget should also account for the amount of contaminated, and therefore unusable, groundwater, instream flow requirements needed to support ecosystems, potential changes in land use and/or land cover that might influence a water budget over time, changes in precipitation, recharge and evapotranspiration (ET) under natural conditions, climate change, virtual water imports/exports (water used to feed animals, or make wine, which are then exported), and water for manufacturing/conservation.

6.2.5 Climate change impacts on groundwater sustainability

Scientists have long been aware of the fact that both natural climate variability and climate change affect aquifer water levels. Groundwater, as important component of the hydrologic cycle, will be affected by climate change in regard to recharge, interactions between the groundwater and surface water systems, and changes in water use (e.g., irrigation) (Zektser and Loaiciga, 1993; Loaiciga et al., 1996). Future climate changes may impact the quantity and quality of regional water resources (Gleick, 1989). These changes, in turn, may lead to detrimental secondary impactson fisheries and other wildlife as a result of changes to the baseflow dynamics in streams (e.g., Gleick, 1986), disruption of the natural equilibrium in coastal aquifers (e.g., Custodio 1987; Lambrakis 1997; Vengosh and Rosenthal 1994), and a reduction in the volume of water stored in aquifers (Rosenberg et al., 1999; Loaiciga et al., 2000). Our understanding of the impact of climate variability and changes on groundwater resources, in terms of availability, vulnerability and sustainability of fresh water, remains limited.

To date, only a few studies have attempted to forecast how climate change might affect groundwater. Of these investigations, most have focused on groundwater recharge (see Chapter 4). (Appaih-Adjei and Allen, 2009; Chen and Grasby, 2001; Chen et al., 2002; McLaren and Sudicky, 1993; Moore et al., 2007; Piggott et al., 2001; Rutulis, 1989; Rivard et al., 2003, 2009; Scibek et al., 2007, 2009; Scibek and Allen, 2006a, 2006b; Toews and Allen, 2009a, 2009b).

Recharge timing and diverse aquifer characteristics complicate our ability to understand and to measure the impact of climate change on our groundwater resources (Rivera et al., 2004).

The response of an aquifer to shifts in climate, as measured by water level changes, is difficult to detect (e.g., Rutulis, 1989; Chen and Grasby, 2001, Rivard et al., 2004; Moore et al., 2007). Insufficient temporal data (short historic time series) often limits our ability to identify trends directly associated with climate change. Because information on groundwater extraction often nonexistent, it becomes difficult to distinguish between anthropogenic and natural influences on groundwater levels.

Different types of aquifers respond differently to surface stresses. Shallow aquifers, consisting of unconsolidated sediments, weathered or fractured bedrock, are generally more responsive to stresses imposed at the ground surface. Deeper aquifers tend to be more isolated from surface conditions by overlying aquitards (e.g., van der Kamp and Maathuis, 1991a). Shallow aquifers are affected by local climate changes, while water levels in deeper aquifers are affected by regional climate changes. Climate variability, which is of relatively short-term duration when compared to climate change, has a greater impact on shallow aquifer systems (Rivera et al., 2004), while deep aquifers have an increased capacity to withstand the effects of climate variability and, therefore, are able to preserve the longer-term trends associated with climate change. The deeper aquifers are also more susceptible to long-term declines in water storage.

There are many types of aquifers, and it is very difficult to place or locate a sufficient number of Canadian observation wells to be used for the detection of climate change signals. As a result, little research has been done in this country to relate well hydrographs with climatic variables. Nor has well hydrographic data been used on a systematic basis to address the question of climate variability impact on aquifers and groundwater resources: some regional assessments have been carried out (e.g., Maathuis and van der Kamp, 1986 for Saskatchewan; Moore et al., 2007 for BC), but from a national perspective, the only publication addressing these issues is Rivard et al. (2009).

6.3 GROUNDWATER VULNERABILITY AND WATER QUALITY RISK ASSESSMENTS

The term aquifer vulnerability has been defined, as "an intrinsic property of a groundwater system that depends on the sensitivity of the system to human and/or natural impacts" (Vrba and Zoporozec, 1994). According to these authors, intrinsic vulnerability is a function of hydrogeological factors, and specific vulnerability describes the potential impacts of land use and contaminants, in addition to hydrological factors. Focazio et al. (2002) define intrinsic sensitivity as "a measure of the ease with which water enters and moves through an aquifer; it is a characteristic of the aquifer and overlying material and hydrologic conditions, and is independent of the chemical characteristics of the contaminant and its sources," and aquifer vulnerability as "a function not only of the properties of the groundwater flow system, but also of the proximity of contaminant sources, characteristics of the contaminant, and other factors that could potentially increase loads of specified contaminants to the aquifer and (or) their eventual delivery to a groundwater resource."

In this chapter (and in keeping with terminology used in many jurisdictions across Canada), we will define **aquifer vulnerability** as a measure of the intrinsic susceptibility of an aquifer which represents the "tendency or likelihood for contaminants to reach a specified position in the groundwater system after introduction at some location above the uppermost aquifer" (National Research Council, 1993). This is a qualitative measure, based on type, thickness and extent of geologic sediments overlying the aquifer, depth to water (or depth to top of confined aquifers), and the type of aquifer material. Aquifer vulnerability is distinct from **water quality** (or **pollution**) **risk**, which depends not only on intrinsic vulnerability but also on the presence of significant pollutant loading owing to the existing type of land use or nature of the potential contaminants. **Vulnerability mapping** is a common method for representing the relative susceptibility of an aquifer, both spatially and semiquantitatively, to contamination from surface sources.

Groundwater quality risk assessments typically involve a multi-step approach. The first step is a vulnerability assessment, followed by a hazard inventory (often within vulnerable areas), finally with a risk assessment. Defining wellhead protection areas through well capture zone analysis and development of plans for emergency responses are related activities often carried out at the local scale.

6.3.1 Assessing aquifer vulnerability

Any determination of aquifer vulnerability requires a solid understanding of both the study area gelogy and the hydrologic conditions contained within. Aquifer depth and the types of geological materials above it are also critical points to consider. Aquifers closer to the surface and overlain by pervious surface materials are more vulnerable to contamination, as compared to deeper aquifers covered with thick layers of impervious materials. Aquifer vulnerability is assessed by using a variety of scales including

- the broad landscape, perhaps at the watershed scale, to identify highly vulnerable areas outside of the areas of wellhead protection
- the wellhead protection areas (WHPAs) to ensure that existing supply wells are protected
- specific locations within the watershed, including those areas with significant groundwater recharge, designated either as highly vulnerable aquifers, or are sites for future groundwater supply

Assessment of aquifer vulnerability always begins with a characterization of the assessed area's geological conditions.

As of this date, it is generally recognized that insufficient data is available to perform specific vulnerability mapping, although there is a higher degree of scientific soundness in "specific" vulnerability maps for specific pollutants (e.g., Foster, 1987; Canter et al., 1987). Consequently, researchers have developed generic mapping systems that are simple enough to apply the generally available data, and are capable of making best use of that data in a technically valid and useful manner. Several such vulnerability evaluation and ranking systems have been developed and applied (e.g., AVI, GOD, DRASTIC, SI, EPPNA and SINTACX), with examples provided by Albinet and Margat (1970), Haertle (1983), Aller et al. (1987), Foster (1987) and Vrba and Zaporotec (1994).

The purpose of groundwater vulnerability assessments is to characterize the contamination potential within a specific geologic setting and to define locations that may be more vulnerable to this contamination than others. These vulnerability methodologies consider that the natural environment protects itself when a contaminant is introduced. Piscopo (2001) considered three groundwater system elements in his method: (1) contaminants entering the system and constituting a threat; (2) soil and rock above the water table forming a barrier to contaminants percolating down from the surface, and (3) the groundwater resource below the water table that could be damaged should contaminants penetrate the barrier.

Vulnerability is evaluated by considering the thickness and permeability of the material situated above the aquifer. Low-permeability surficial soils, composed largely of clay and silt, are usually less likely to transmit significant quantities of contaminants when compared to high-permeability soils such as sand and gravel. Low-permeability materials create aquitards which function as barriers to contamination. Thickness of the overlying materials plays an important role, however, because contaminants applied, deposited or spilled on or near the ground surface, will be less attenuated and will reach an aquifer more quickly in those locations where the overlying material surface is thin. Fractures or other openings in an aquitard overlying the aquifer can also negate the natural protection. In those areas where bedrock is exposed at surface, vulnerability will be highly dependent upon the degree and inter-connectivity of fracturing.

6.3.2 AVI

AVI quantifies vulnerability by the hydraulic resistance (c) to vertical flow of water through geologic sediments above the aquifer. Hydraulic resistance is calculated from the thickness (d) of each sedimentary layer and the hydraulic conductivity (K) of each of the layers (Equation 6.6).

Hydraulic resistance (c) =
$\sum d_i / K_i$, for layers 1 to i (6.6)

Typically, saturated hydraulic conductivity (K_{sat}) is

TABLE 6.1 AVI CATEGORIES

HYDRAULIC RESISTANCE, C (YEARS)	LOG (C)	VULNERABILITY CATEGORY
< 10 years	< 1	extremely high vulnerability
10–100 years	1 to 2	high vulnerability
100–1,000 years	2 to 3	moderate vulnerability
1,000–10,000 years	3 to 4	low vulnerability
> 10,000 years	> 4	extremely low vulnerability

the variable most often used, although it produces conservative hydraulic resistance values (higher vulnerability) for unsaturated sediments above the water table. Thickness of individual sedimentary layers can be calculated directly from well records. Hydraulic resistance (c) has the time dimension (e.g., years) and represents the flux, or time per unit gradient of water flowing downward through the various sediment layers to the aquifer. The lower the hydraulic resistance (c), the greater the vulnerability.

A vulnerability map is constructed by calculating the logarithm of the hydraulic resistance (log c) for each well site and by delineating or contouring areas of similar log c (AVI) values. Resultant AVI ratings indicate surficial materials' potential to transmit water with contaminants to the aquifer over a period of time: these ratings can be grouped into vulnerability categories (Table 6.1). A location with a low class rating implies that water percolating through its surficial materials takes a long time (in the range of thousands of years) to reach the aquifer. On the other hand, a location with a high class rating suggests that contaminated water will reach the aquifer within "tens" of years.

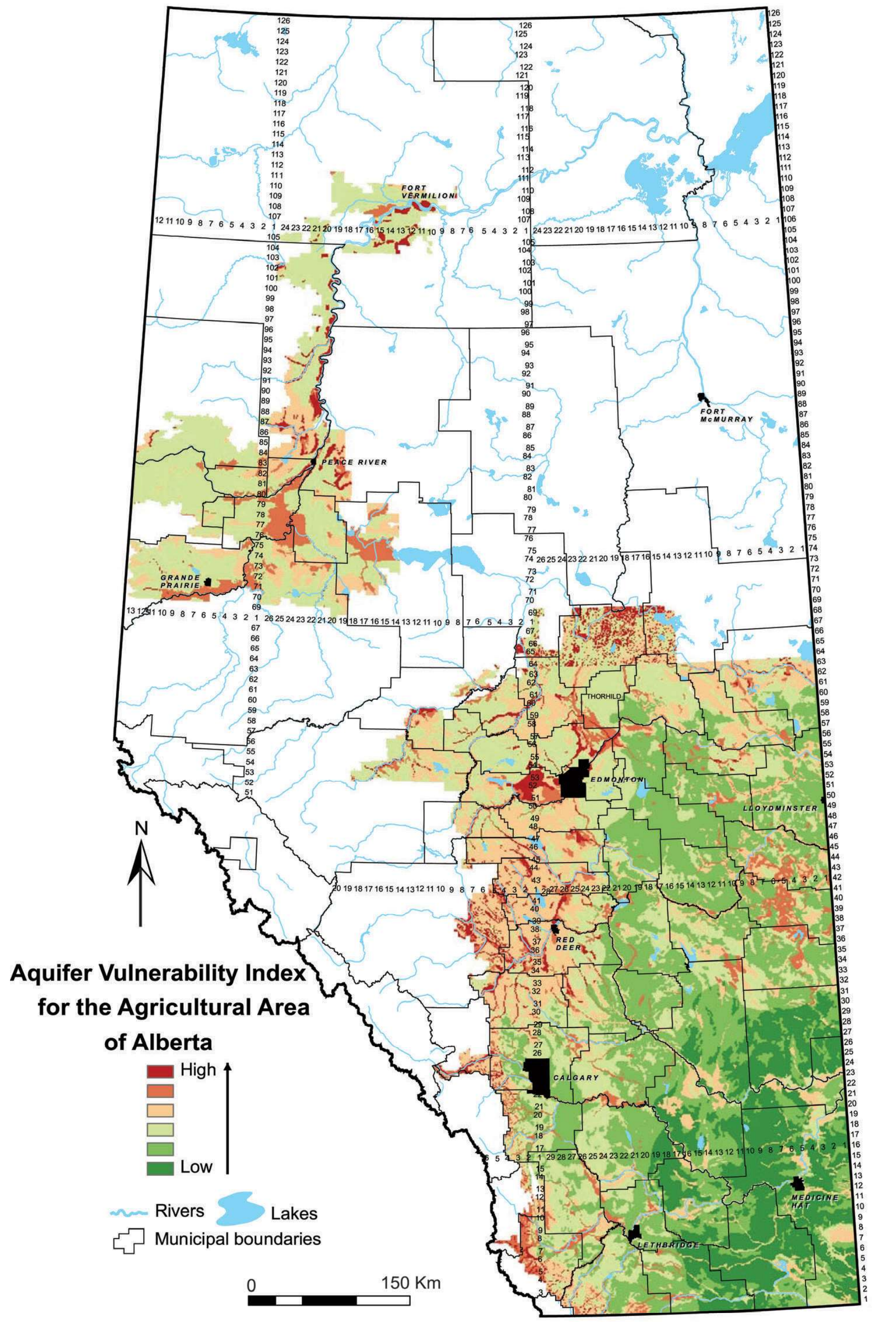


Figure 6.10 AVI map for the agricultural areas of Alberta (Alberta Government, 2005).

The AVI method has been used extensively across Canada. Here is an AVI map for Alberta (Figure 6.10).

6.3.3 DRASTIC

DRASTIC was developed by the U.S. EPA (Environmental Protection Agency) as a standardized system for evaluating groundwater vulnerability (Aller et al., 1987). DRASTIC's primary purpose is to provide assistance in resource allocation and prioritization of the many types of groundwater-related activities. Like AVI, DRASTIC can be used to establish priorities for groundwater monitoring in specific areas. DRASTIC can also be used with other information (e.g., land use, potential contamination sources, and beneficial aquifer uses) to identify those locations where special attention or protection efforts are warranted.

The DRASTIC model contains four assumptions:

1. The contaminant is introduced at ground surface
2. The contaminant is flushed into the groundwater by precipitation
3. The contaminant has the mobility of water
4. The area being evaluated by DRASTIC is 100 acres or larger

DRASTIC is a composite rating of the **D**epth to water, net **R**echarge, **A**quifer media, **S**oil media, **T**opography slope, **I**mpact of the vadose zone and the hydraulic **C**onductivity of the aquifer (Figure 6.11).

Depth to Water (D) represents the thickness of geologic material above the water table. Groundwater is more vulnerable to contamination when the water table is close to the surface and the soil, or rock barrier zone above the water table is thin, allowing little capacity for natural filtration of contaminants before they reach the water table. Net recharge (R) reflects the total amount of water per unit area percolating from the surface to the water table. The greater the water flow, the more likely contaminants will reach the groundwater resource. Key factors influencing recharge include precipitation, topography and the properties of soil and the surficial geology materials above the water table. Aquifer Media (A) represents the character of the consolidated or unconsolidated material which serves as an aquifer. Path length and travel time of groundwater within an aquifer depend on the grain size of the bedrock and the presence of fractures within the aquifer. The effectiveness of soil (S) to act as a barrier to surface contaminants depends on its physical properties. In general, soils containing clay properties and with small grain sizes are less permeable and are more effective as barriers to contamination. The slope of the land or topography (T), coupled with changes in slope can influence the proportion of rainfall that forms runoff compared to the water volume that infltrates the soil. When slopes are low, there is a greater likelihood that a pollutant will infiltrate an aquifer through the ground surface. The impact of the vadose zone (I) represents the unsaturated zone above the water table: the texture of soil and rock within this zone determines how rapidly water and, thus contaminants will infiltrate downwards toward the water table. Conductivity (C) reflects the rate of groundwater flow through an aquifer. Rapid flow allows rapid contaminant spread. Like the "I" parameter, conductivity reflects the rate at which water flows through the aquifer.

Each of the hydrogeological factors of DRASTIC is assigned a rating from 1 to 10 based on a range of standard values. These ratings are then multiplied by a relative weighting factor, ranging from 1 to 5 (Equation 6.7):

$$\text{DRASTIC Index} = 5D + 4R + 3A + 2S + 1T + 5I + 3C \quad (6.7)$$

The most significant factors have a weight of 5; the least significant a weight of 1. The final DRASTIC index represents a relative measure of groundwater vulnerability; the higher the index, the more vulnerable the aquifer to contamination.

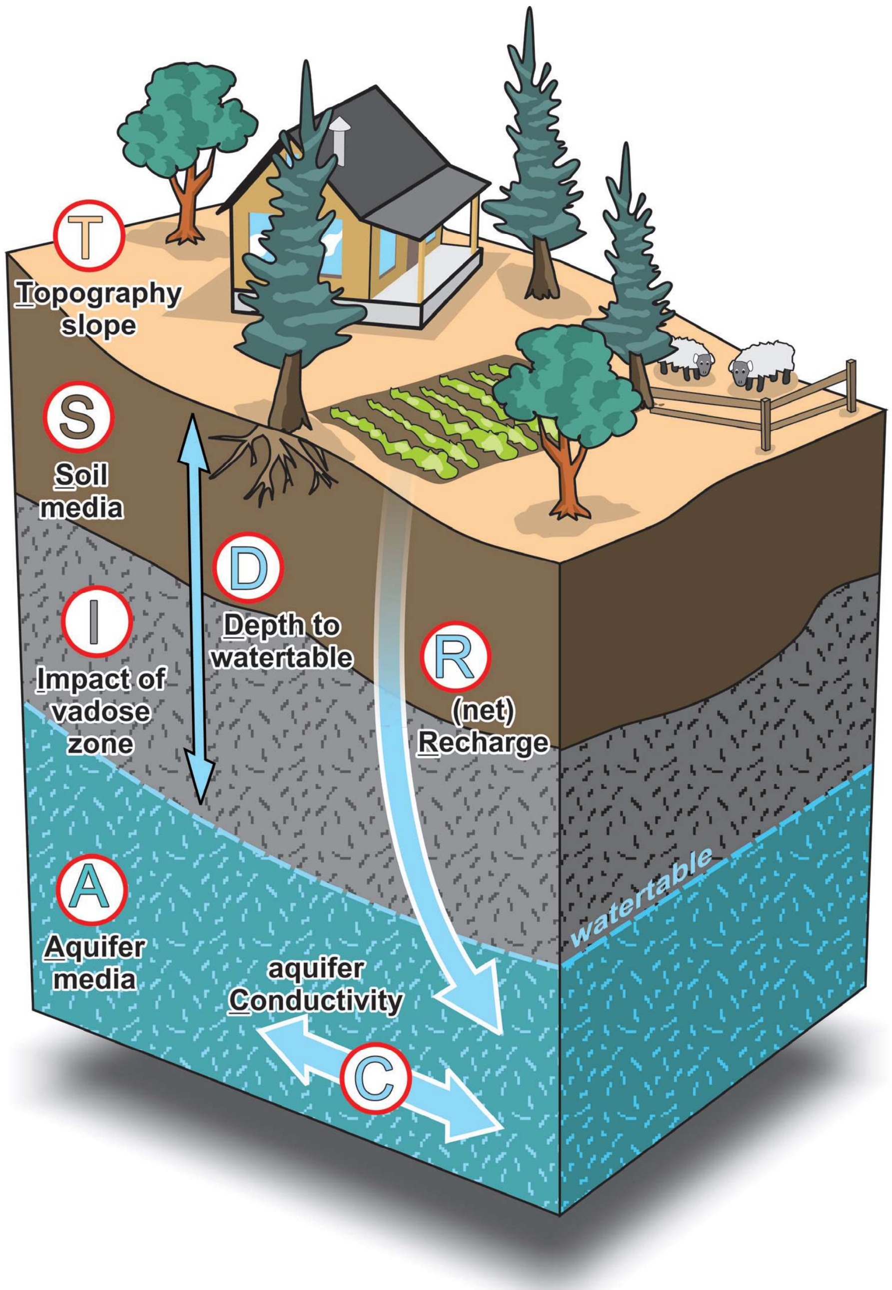


Figure 6.11 The DRASTIC factors (figure courtesy of the Geological Survey of Canada).

The smallest DRASTIC index rating is 23 and the largest in 226. Although DRASTIC is physically based, the final DRASTIC index, unlike AVI, has no physical meaning, but rather is purely an index. One advantage of DRASTIC over AVI, however, is that it includes a wider range of those parameters thought to influence the transport of contaminants through the vadose zone.

A modified DRASTIC methodology has been developed for aquifers that are strongly influenced by fractures (Denny et al, 2006). This methodology, DRASTIC-Fm, is highlighted in Box 6-2 with the Gulf Islands in British Columbia used as the case study area.

6.3.4 Comparison of vulnerability mapping methods

In recent years, capabilities of geographical information systems (GIS) have vastly improved our ability to construct vulnerability maps. Such maps were formerly done on a well-by-well basis, with indices for individual wells plotted and contoured. Today, GIS allows multiple spatial datasets—soil maps, geologic maps, digital elevation models (DEMs), etc.—to be analyzed, assigned relevant indices, and synthesized into composite maps such as those described above for AVI and DRASTIC. Consequently, aquifer vulnerability mapping is rapidly becoming a common tool of groundwater risk assessment frameworks in many jurisdictions across Canada.

Other aquifer vulnerability mapping methods, such as the BC Aquifer Classification System (see Chapter 9) or modifications of those methods described above are also used across the country. No particular vulnerability mapping method is necessarily better than another.

Wei (1998) evaluated AVI and DRASTIC against each other in the southwestern B.C's Lower Fraser Valley only to find that both indexes were generally consistent. Low AVI values corresponded with high DRASTIC indexes (high vulnerability), while high AVI values corresponded with low DRASTIC indexes (low vulnerability). Despite these general consistencies, DRASTIC indexes between 100 and 160 spanned all five AVI vulnerability categories, suggesting that DRASTIC may not be as sensitive as AVI for indicating pollution potential of aquifers with moderate vulnerability. Wei also noted that DRASTIC indexes of >160 (high vulnerability) and <80 (low vulnerability) fell under the extremely high and extremely low to low vulnerability AVI categories, respectively.

6.3.5 Wellhead protection

The purpose of wellhead protection is to prevent any contamination of groundwater used for drinking water. A **wellhead protection area** (WHPA) is the location immediately surrounding a well that must be protected from potential sources of surface contamination. The WHPA represents a surface projection of the entire three-dimensional capture area from which the water pumped from the well or well-field originates (Figure 6.12). The entire recharge area for the well (Figure 6.12) surrounds the WHPA.

Several jurisdictions across Canada (as well as the United States Environmental Protection Agency—EPA) have specific guidelines for defining wellhead protection areas. For the most part, these guidelines are based on defining a **well capture zone**(s), whose size(s) is/are influenced by the well pumping rate, aquifer porosity, and hydraulic conductivity. Size and shape are influenced by hydraulic gradient and flow direction, the orientation and density of fractures/faults. They are also influenced by dissolution features like those in karst geology wherein the capture zones of rock aquifers may extend for many kilometres due to permeability of fractures, faults and dissolution channels.

Well capture zone analysis presents a number of difficulties and uncertainties of which the most important relates to the cumulative effect of pumping from a number of wells situated in close proximity. When this occurs, the capture zones for individual wells coalesce, effectively creating a larger composite capture zone than those of the individual wells. As a result, analysts look at well clusters.

Contaminants, such as pathogens or chemicals, have varying fates and degrees of persistence, and scientists must consider times of travel (TOT) when defining well capture zones. Bacteria, for example, have a limited life span and an adequate time of travel from the point of entrance to the well may

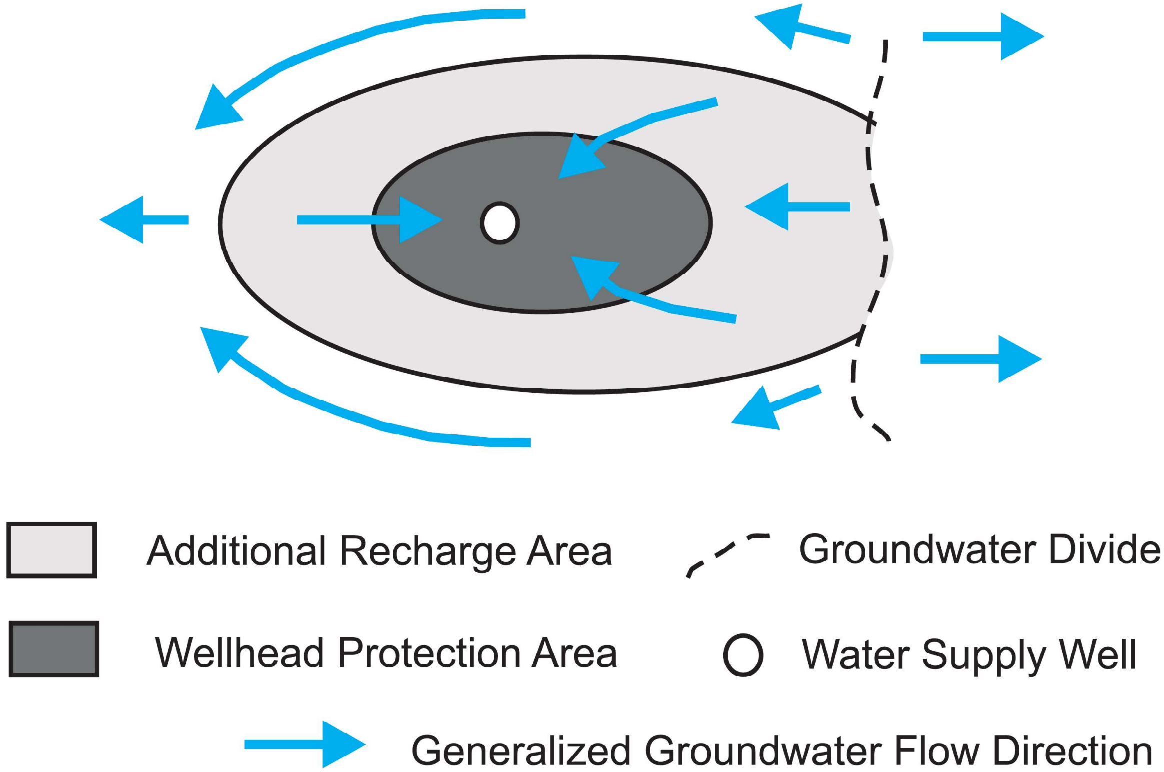


Figure 6.12 Wellhead protection area (WHPA) and additional recharge area surrounding a water supply well.

effectively inactivate these organisms. Similarly, over time, some chemical contaminants degrade into lower risk compounds or are adsorbed by the geological materials encountered along the flowpath. Other chemicals, however are stable in a groundwater setting and the risk from their presence may only be attenuated through dilution along the flowpath.

The Ontario Ministry of Environment recommends that each WHPA (for municipal groundwater wells) be sub-divided into well capture zones to differentiate between, and effectively manage, potential risks posed to well water quality from various types of microbiological and chemical contaminants that might enter the water table and/or move with the groundwater flow (Ontario Ministry of Environment, 2001). At a minimum, the Ministry recommends three well capture zones be delineated for each municipal production well/well field:

1. **Zone 1:** 0 to 2 years saturated time of travel (TOT). Land uses in this zone need to be monitored and managed to avoid all possible risks, including those from bacteria and viruses. Within Zone 1, a 50-day TOT area should be identified to recognize potential risks from day-to-day activities of the water utility itself or other contaminant sources.
2. **Zone 2:** 2 to 10 years TOT. The main focus of land use management in this zone should be to minimize risks from all chemical contaminants, although bacterial and viral risks may still be a concern.
3. **Zone 3:** 10 to 25 years TOT / Zone of Contribution. The land use management in this zone needs to address risks from

persistent and hazardous contaminants.

The methods used for defining these capture zones range from simple and inexpensive to complex and costly.

Calculated Fixed Radius method: This procedure, also known as the "cylinder method", is easy to use and is based on simple hydrogeological principles that require limited technical expertise. Calculated fixed radius capture zones are circular areas whose radius is determined using the formula:

$$r = \sqrt{[(Q \times t) / (\pi \times b \times n)]} \qquad (6.8)$$

where:
r = radius (distance from well) in metres
Q = maximum approved pumping rate of the well (m^3/s)
t = saturated travel times for each well capture zone (sec)
b = saturated thickness of screened interval (m)
n = porosity
π = 3.14159…

The method, however, tends to overprotect down-gradient and under protect up-gradient areas because it does not account for regional gradients. Unless combined with flow system mapping, this method should not be used for unconfined aquifers or for confined aquifers with a sloping potentiometric surface.

Uniform Flow Method: This procedure utilizes analytical expressions to delineate capture zones. These include

1. the distance to down-gradient null point:
$X_L = Q / (2 \times \pi \times K \times b \times i)$

2. shape of outer streamline:
$X = - Y / \tan [(2 \times \pi \times K \times b \times i) / Q \times Y]$,

where the boundary limit (asymptotic width) of the capture zone is:
$Y_L = \pm Q / (2 \times K \times b \times i)$; and

3. upgradient distance as a function of time:
$X_t = K \times i \times (t / n)$

where:
X = distance along length of capture zone (m)
Y = width of capture zone as a function of X (m)
Q = maximum approved pumping rate of the well (m^3/s)
K = hydraulic conductivity (m/s)
b = saturated thickness of screened interval (m)
i = hydraulic gradient
t = saturated travel times for each well capture zone (sec)
n = porosity
π = 3.14159…

This method is more flexible than standard analytical models because it can conform to variability in flow direction. The disadvantage of this method is that it generally does not take into account hydrologic boundaries (streams, lakes, etc.) and aquifer heterogeneities, and it assumes no recharge. It is also limited to two-dimensional analyses of flow systems and capture zone delineation.

Analytical or **Numerical Methods:** The most sophisticated methods for determining a well capture zone are based on field observations of aquifer characteristics during a detailed pumping test, coupled with calculations or numerical modelling designed to predict long-term aquifer conditions. These types of delineation methods require a qualified hydrogeologist and may be appropriate when siting new large wells or when a source protection program emphasizing extensive land use restrictions is planned. These methods require data regarding well production rate, the aquifer's lateral extent, thickness, hydraulic conductivity and flow gradients. These advanced methods are suitable for accurately delineating capture zones where there is a significant presence of: (1) discrete fractures, (2) anisotropy, (3) spatial variations in hydrogeological parameters, (4) vertical movement of water and variation in total hydraulic head with depth, and/or (5) changes in water levels seasonally or through time.

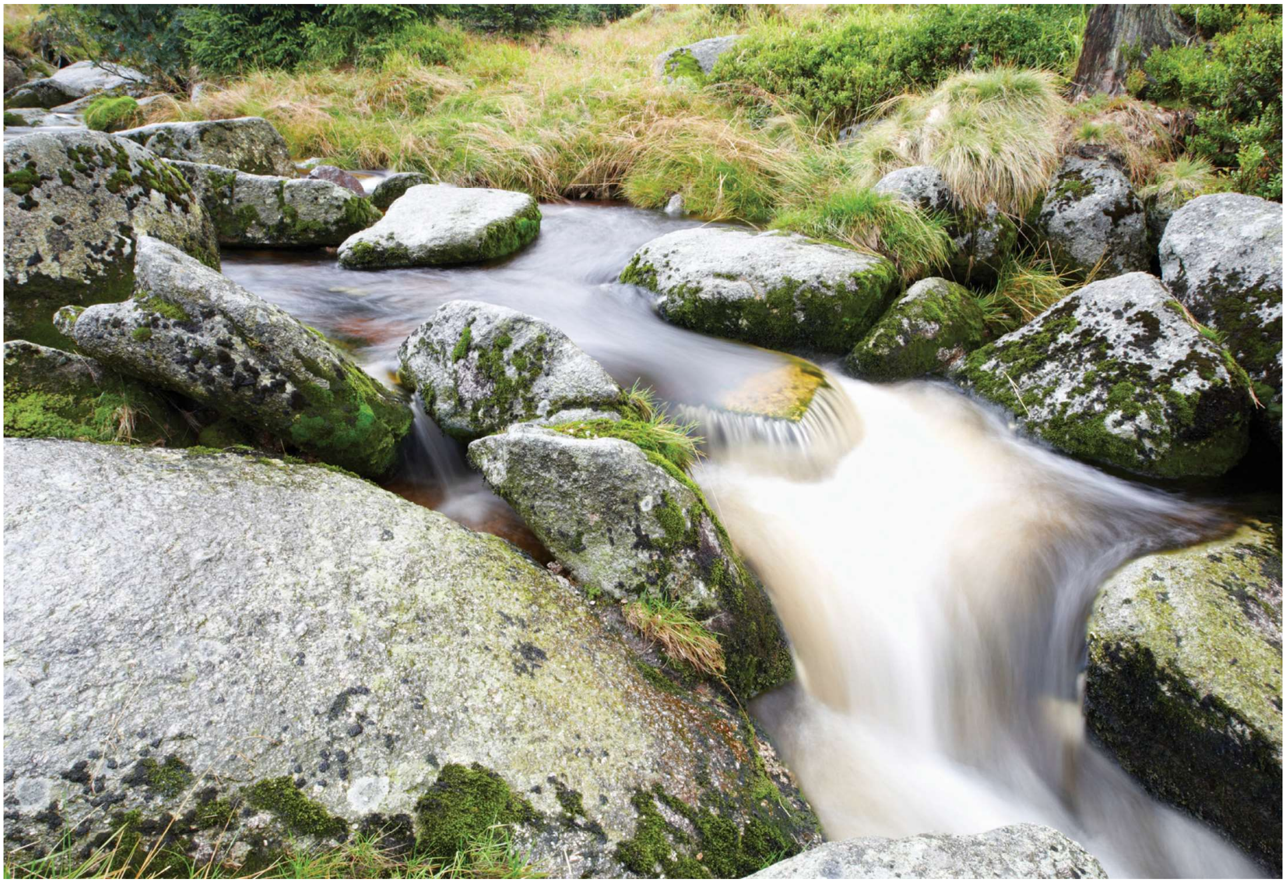
6.3.6 Integrating land use and groundwater into decision making

Land use and water resources are unequivocally linked. Land type and intensity of its use can have a substantial impact on the receiving water resource. Although, for the most part, water quality across Canada is good, an increasing population, development pressures, minimal (or the absence of) integrated land use planning, and competition for water resources, continually contribute to the water resource degradation. Both here and worldwide, the protection and provision of fresh water (in terms of quality and quantity) have become a top concern of political leaders and the general public alike. Indeed, this changing focus has created a whole new vocabulary devoted to water resources. Terms such as multi-barrier approaches to source protection, source protection plans, wellhead protection plans, and integrated water resources management (IWRM) are referred to commonly on government-hosted websites. All of these phrases stress the need for comprehensive approaches for protecting water.

Not all groundwater resources are equally vulnerable to contamination, and areas with similar land uses and contamination sources may have different degrees of vulnerability and, therefore, different response rates to protection and management strategies. Thus, as hydrogeologists seek to determine groundwater resource vulnerability, they must consider all elements of the natural landscape: land use, contamination sources, land cover, surface and subsurface materials, seasonal variations in surface and subsurface hydrology, and man-made features.

Another complication is the fact that the relationship between land use and water quality is bidirectional. Land use activities can have direct impacts on groundwater resources, while water quality (and quantity) exerts strong influences on the siting of land use activities. Land use is, in part, determined by environmental factors: soil characteristics, climate, topography, and vegetation. Thus, successful land management requires a solid understanding of the relationship between land use/land cover and water resources. The physical and chemical processes between earth's atmosphere, its land surface and its hydrosphere are dynamically linked and demand models which can represent these coupled processes accurately. Such models are extremely difficult to develop and require large amounts of data (of which there is a paucity in most areas of this country).

Regional- and local-scale aquifer vulnerability assessments provide an effective means of assembling key information assets, of identifying environmental trends, and of prioritizing the need for detailed site-specific investigations within groundwater environments (Bekesi and McConchie, 2002; Aller et al., 1987). Canada's provinces and territories have primary jurisdiction over (ground) water resources and (ground) water supply, and each province/territory may regulate its groundwater resource differently (Nowlan, 2007). The Ontario Clean Water Act (Draft Guidance Modules http://



www.ene.gov.on.ca/envision/water/cwa-guidance.htm), for example, requires that source water areas sensitive to groundwater pollution be identified through vulnerability analysis, issues evaluation and followed by a threats inventory.

An issues inventory details problems currently existing in the source water, or with problems which might be reasonably predicted to become source water issues should rising trends continue.

Threats are activities on the landscape that, if managed improperly, may cause future problems. Potential pathways, such as water wells, are also considered as possible threats. Hazard ratings (high, medium, or low) are assigned to each chemical or pathogenic contaminant of concern. The final water quality risk assessment includes a definition of whether there is significant risk, moderate risk, low risk, or negligible risk in terms of human health and/or vulnerability of the drinking water source.

Other jurisdictions, for example British Columbia, have no formal groundwater quality risk assessment frameworks, but offer tools to water purveyors and municipalities for developing wellhead protection plans (e.g., the BC Wellhead Protection Toolkit http://www.env.gov.bc.ca/wsd/plan_protect_sustain/groundwater/wells/well_protection/pdfs/intro.pdf, the BC Aquifer Classification System http://www.env.gov.bc.ca/wsd/plan_protect_sustain/groundwater/aquifers/Aq_Classification/Aq_Class.html).

Although typically regulated at the provincial or territorial level, source water protection is usually carried out at the local level and can involve multiple jurisdictions, if a watershed approach is used. There is a growing interest on the part of municipalities, regional districts, conservation authorities, and the like, to consider groundwater supply and its protection in future planning. This interest is largely a consequence of legislation, but also in response to growing concerns related to groundwater resources. One example wherein a local government has used groundwater information for planning purposes is the Town of Oliver in B.C.'s Okanagan Valley (Box 6-3). In this community, aquifer vulnerability maps and well capture zones for municipal wells were embedded in the official community plan. These maps were also included in a land use allocation model used to evaluate build-out scenarios for future growth.

Ideally, groundwater protection should be considered alongside surface water protection, because the two are inextricably linked, and because watersheds are defined by topography, they represent the most logical basis for managing water resources. Once we consider the water resource as a focal point, a more complete understanding of overall conditions in an area and the stressors that affect those conditions can be achieved. Management will then be better equipped to determine what actions are needed to protect or restore the resource.

Traditionally, water quality improvements have focused on explicit pollution sources (e.g., mining effluent) or specific water resources (e.g., a river segment or wetland). While this approach may be successful in addressing specific problems, it often fails to address the more subtle and chronic issues that can contribute to a watershed's decline. Major features of a watershed protection approach include targeting priority problems, promoting a high level of stakeholder involvement, integrating solutions which make use of the expertise and authority of multiple agencies, and measuring success through monitoring and other data gathering.

6.4 PLANNING FOR THE FUTURE

Groundwater availability and its use depend on a number of factors affecting both the natural (or raw) resource and the developed resource (that part of the natural resource that is reliably available for use). Sustainability of groundwater resources cannot be defined as an absolute concept: it is relative, with many variations. We described in section 6.2.2, how "Groundwater development can be considered sustainable if it is used in a manner that can be maintained for an indefinite time without causing unacceptable environmental, economic, or social consequences." According to this definition, we are not willing to accept negative consequences to the environment, to the economy, or to society, consequences that may be caused by unsustainable groundwater development; these consequences must be quantified before decisions are made. This is one of the main challenges for sustainable development of groundwater resources in Canada: we need to fill in the gaps in groundwater knowledge across this country.

Sustainable development of groundwater resources is not merely a scientific concept: it is a perspective that can frame scientific analysis. The evolving concept of sustainability presents a challenge to hydrogeologists as they translate complex, and sometimes unfamiliar, socio-economic and political questions into technical questions which can be quantified systematically. Groundwater scientists can and should contribute to sustainable groundwater resources management by presenting the longer-term implications of groundwater development as an integral part of their analysis (Rivera, 2008).

As seen in the previous sections, the overall security of groundwater resources is strongly linked to water sustainability (i.e., its availability and use), and its vulnerability to contamination. These elements need to be considered holistically and decisions made based on aquifer knowledge (hydrogeological maps, water budgets), as well as social (water demands), political (water laws and regulations), economic and environmental issues (ecosystem needs). Developing sustainable management strategies requires that decision-makers have a comprehensive understanding of these demands and challenges, and a detailed awareness of the economic and political instruments at their disposal (Trainer, 2010).

The physical and chemical characteristics of an aquifer may be used as indicators of the quality and quantity of groundwater, but these characteristics must be considered jointly with societal factors that determine actual groundwater availability, coupled with society's tolerance of the consequences of groundwater use for long-term security. We need to begin with some physical definition of water availability and then consider all other factors in order to make informed decisions regarding water management.

Science plays a very important role in this procedure, but science alone is not sufficient for managing groundwater resources.

6.4.1 Challenges in Canada

Sustainable use of groundwater resources demands knowledge of recharge and discharge, and information on water use/needs for domestic, agriculture and industry activities and for ecosystem services. This information is used for water balance modelling, which can be accomplished either through the use of simple accounting or by creating sophisticated numerical groundwater flow models for individual aquifers (or any other management unit, e.g., watershed, where groundwater is withdrawn by humans and/or needed by ecosystems). The Okanagan Basin Supply and Demand Study, for example, has experimented with both water balance accounting and numerical modelling to determine current and future projected water balances for the basin.

Studies such as that profiled in Box 6-1 have employed numerical models. This level of analysis requires accurate water balance data derived from assessments and long-term monitoring data (which is currently limited or nonexistent, with the exception of a few well studied and monitored aquifers across the country). Our knowledge of groundwater components in the water cycles of Canada, from local to regional scales, is not adequate or sufficiently comprehensive. When Trainer (2010) summarized the key findings of a report by the Council of Canadian Academies (2009), she noted "a lack of consolidated knowledge to define Canada's groundwater endowment and supply, coupled with a limited understanding of groundwater economics, represents a significant impediment to informed policy-making and long-term sustainable resource planning. The Council's 15-member panel—comprised of leaders in groundwater science as well as experts in the social, economic, and legal fields relevant to sustainable groundwater management—concluded that a Canada-wide sustainability framework, applied at all levels of government, is required to improve the management and understanding of Canada's groundwater.

Additionally, the Panel identified five sustainability goals (summarized below from the Council of Canadian Academies, 2009) needed to define a framework upon which a system-based approach to sustainable groundwater management could be developed: The first goal stated that to be sustainable, groundwater management must seek to prevent continuous, long-term declines in regional groundwater levels. In order to meet this goal, a comprehensive understanding of large-scale groundwater flow dynamics is required. The development of a common framework for aquifer categorization would allow integration of data from local studies into broader regional and national assessments.

The second sustainability goal required that groundwater quality must not be compromised by a significant degradation of its chemical or biological character. Sustainable groundwater management must seek to both prevent groundwater contamination in the first place, and to remediate and restore already-contaminated groundwater.

The third goal sought a sustainable management plan which balanced the human benefits of groundwater extraction against the ecosystem benefits realized by maintaining adequate stream baseflow, and wetland, river, and lake habitats.

The fourth goal spoke to the achievement of economic and social well-being. The economic benefits of sustainable management policies should be considered in the context, not just of direct economic impacts but also in contribution to Canada's environment and society. In order to promote efficient water usage, end-users should be aware of the full costs and benefits of their water consumption.

The fifth goal emphasized the need for good water governance. Water governance can be defined as the range of political, organizational and administrative processes used to articulate interests, receive input, make and implement decisions, and hold decision-makers accountable. Good governance must ultimately include the means to achieve balance among the other four sustainability goals —failure to do so means that groundwater management decisions will likely favour socio-economic interests over ecosystem and environmental interests, leading to situations that are inherently unsustainable.

6.4.2 Recommendations

To overcome the knowledge deficit on groundwater resources, Canada needs to invest in comprehensive aquifer assessments and to make long-term commitments for collecting, maintaining and analyzing groundwater data. Future studies in water stressed areas, or in areas where aquifers may be highly vulnerable to contamination, should aim to collect higher quality data, with enhanced spatial and temporal resolution and at increased precision compared to data collected in the past. Measurement techniques can be borrowed from other disciplines and adapted to provide new methods for determining the value of key variables. Data from new sensors and from existing networks must be integrated, and new observation networks established (in British Columbia, for example, recent efforts have sought to eliminate redundant data collection from observation wells and to install new observation wells in targeted areas). And lastly, measures for quality control, quality assurance, and for data sharing need to be established (among and within jurisdictions) in order to manage groundwater resources of shared aquifers.

Long-term monitoring of groundwater levels, groundwater quality and groundwater use will go a long way to support sustainability and source water protection efforts.

BOX 6-1 GROUNDWATER SUSTAINABILITY IN THE ST. LAWRENCE LOWLANDS

Groundwater sustainability was estimated for two regional aquifers located in the St. Lawrence Lowlands, southwestern Quebec (Nastev et al., 2005; Nastev et al., 2006) (Figure 6.13). The study area of the Mirabel aquifer encompasses approximately 1220 km^2 just north of Montreal. That of the Châteauguay River basin covers 2850 km^2 on the south shore of the St. Lawrence River. These two regions are relatively densely populated with more than 450,000 inhabitants;

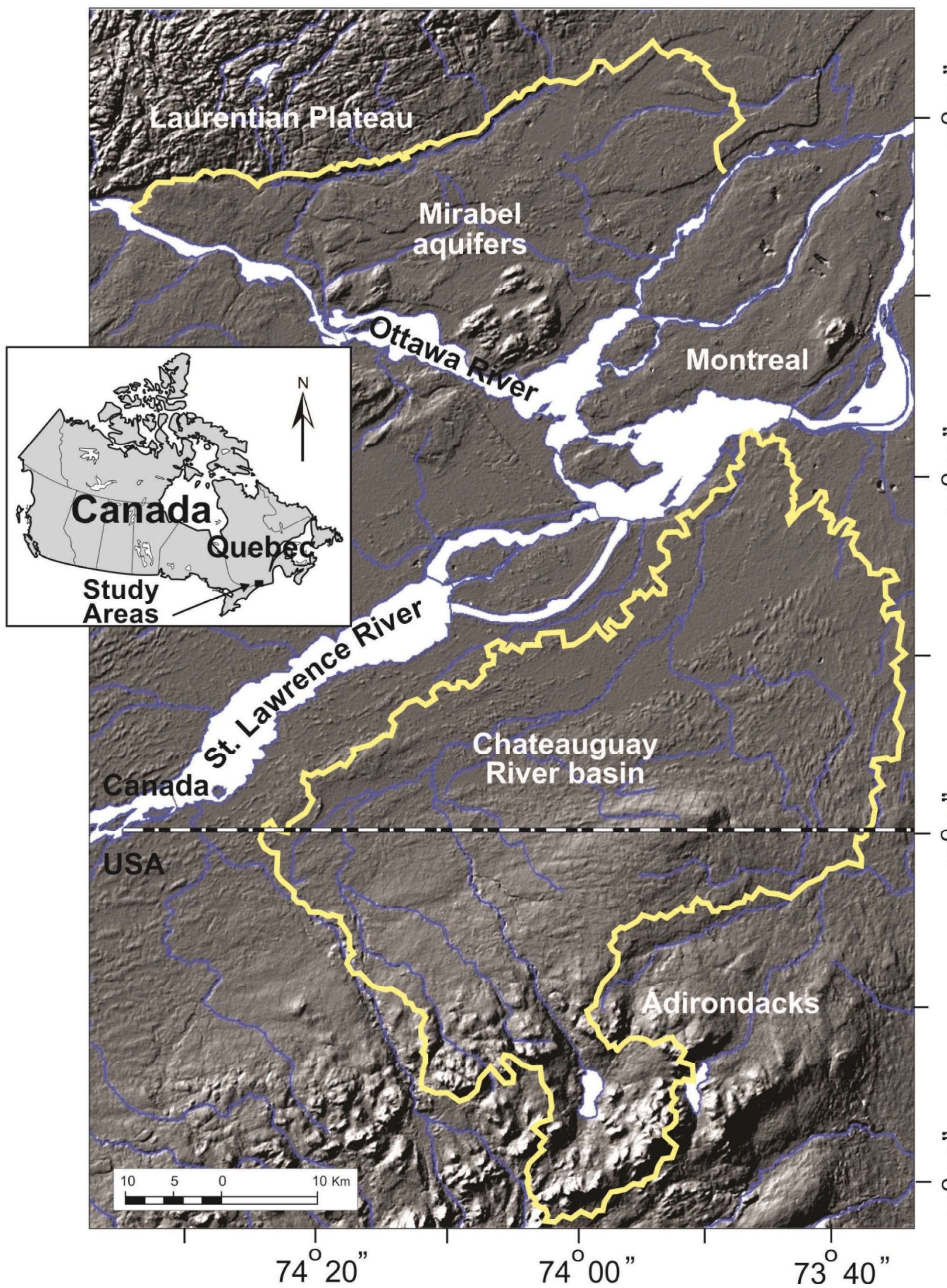


Figure 6.13 Location of the study areas.

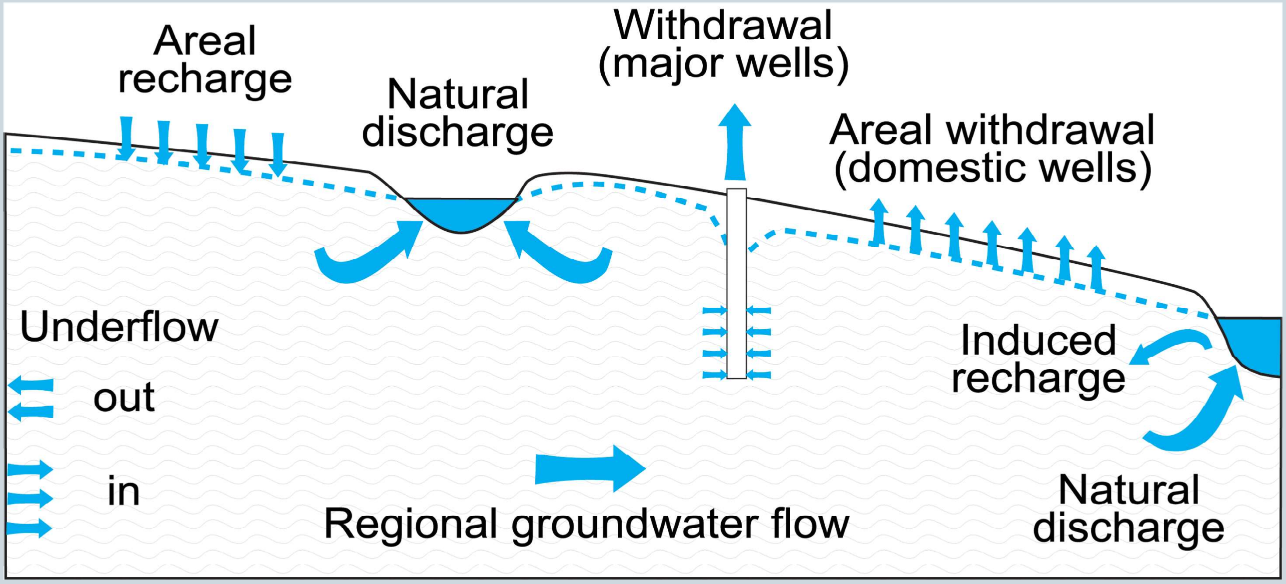


Figure 6.14 Schematic presentation of the long-term groundwater balance components for the Mirabel aquifer in Mm^3/year (Nastev et al., 2005).

both locations have intensive agricultural and industrial activities. The population, located mainly in the rural communities, is heavily dependent on groundwater for its daily needs. In addition, the Chateauguay basin forms a trans-boundary aquifer extending across northern New York State. The regional aquifers consist of sedimentary strata of the Lower Paleozoic period: sandstones, carbonate-dominated dolostone rocks, and limestones. They are underlain by crystalline Precambrian rocks which crop out to the north as the Laurentian Plateau (Canadian Shield) and to the south as Adirondack highlands. The groundwater flow in sedimentary rocks occurs primarily through sub-horizontal bedding planes and sub-vertical fractures and joints.

Lack of knowledge of the groundwater flow and of the availability of the groundwater resource in the region precludes the formulation of suitable groundwater management plans. Two numerical models of the regional groundwater flow were developed as essential tools for the assessment of the regional sustainability of the groundwater resource. The simulated steady-state groundwater balance components for the Mirabel aquifer are schematically presented in Figure 6.14. Assuming the useful depth of the aquifers from which groundwater can be practicably extracted to 200 m and an average effective porosity of 1%, the total volume of the stored groundwater was estimated at 2,400 Mm^3 and 5,700 Mm^3 for the Mirabel and the Chateauguay aquifers, respectively. Thus, the renewable quantity of the groundwater on an annual basis within the regional aquifers, obtained as the ratio between the regional groundwater flow and the volume of stored groundwater, varies in the range of 4–5%. The groundwater withdrawal for various uses amounts to only 0.6–0.7% of the groundwater present in storage.

Calibrated steady-state numerical models were used to simulate predictive scenarios. The withdrawal rates were assumed as successively increasing extraction rates applied uniformly on the top of the modelled domains. The long-term effects of the applied withdrawal rates were defined by the resulting average regional drawdown values

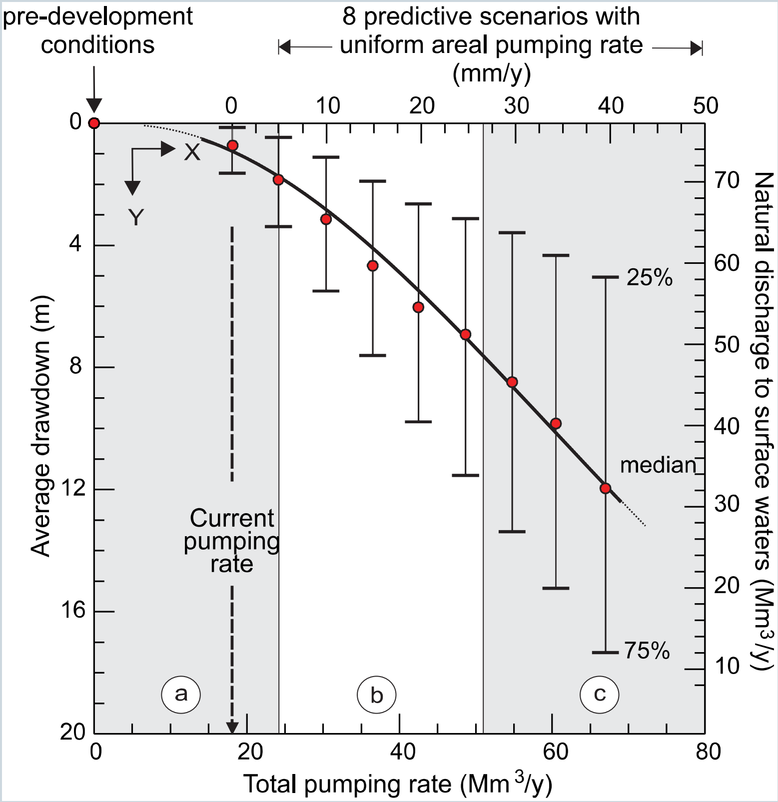


Figure 6.15 Simulated drawdowns and discharge rates for imposed uniform withdrawal scenarios for the Mirabel aquifer: (a) sustainable pumping, (b) pumping with increased drawdowns, (c) non-sustainable pumping (Nastev et al., 2006).

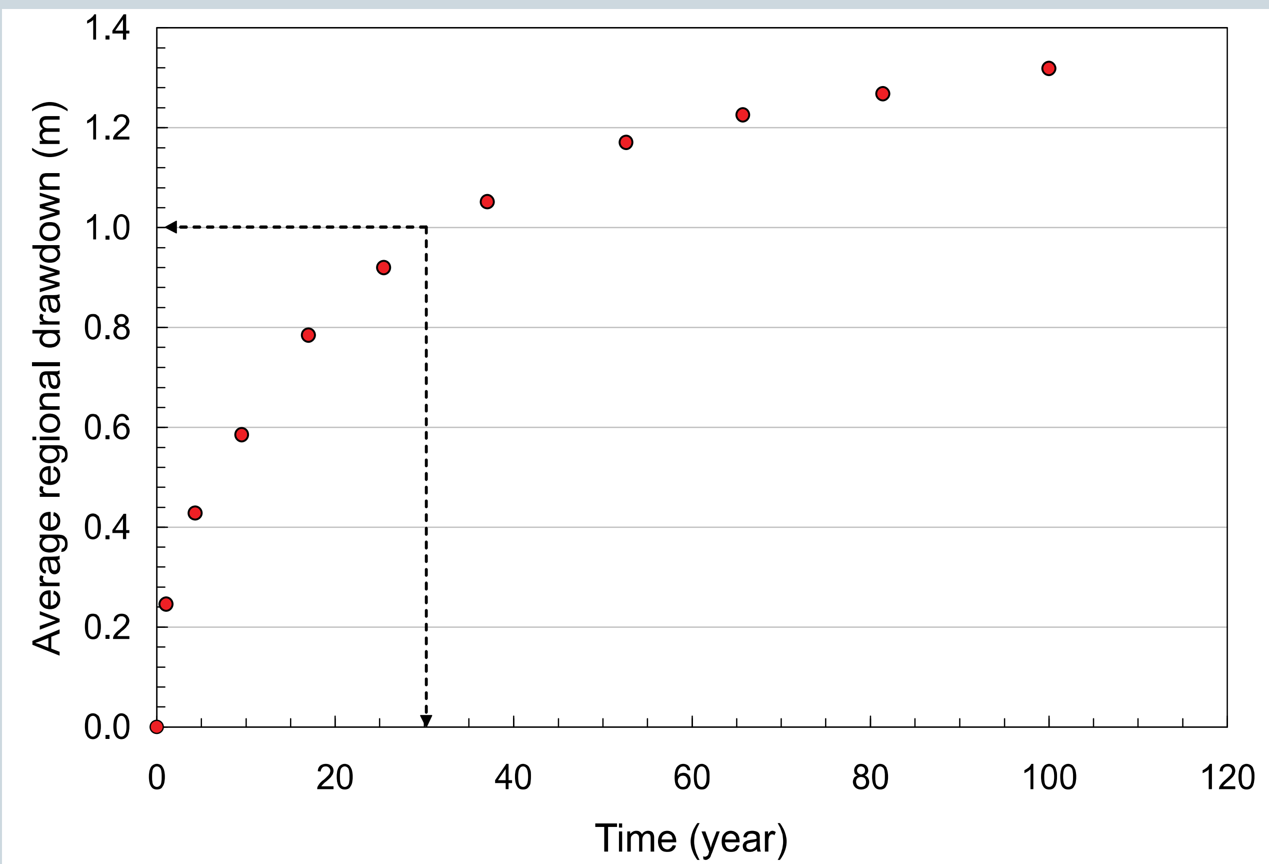


Figure 6. 16 Evolution of the simulated regional drawdown for the Chateauguay aquifer.

as the only considered negative impact. In this way, a cause-effect (pumping rate-drawdown) relationship was defined. The simulated average drawdown for the Mirabel aquifer is depicted against the imposed groundwater withdrawals in Figure 6.15. For comparative purposes, the additional uniform withdrawals are expressed in mm/year on the upper x-axis. To avoid hypothetical situations where extreme drawdowns could have been obtained, the maximal withdrawal rate was limited to 40 mm. One additional simulation was conducted with no withdrawal at all. This last scenario represents the flow in pre-development conditions.

The computed drawdown is not a random variable but rather is influenced by the simulated hydrogeologic conditions. In this case, the Gaussian distribution is no longer valid. The regional drawdown values are estimated with percentiles. The median represents the average estimate for which 50% of all drawdown values fall below. The distance between the 25th and 75th percentile indicates the range which includes 50% of the drawdown values, 25% of the drawdown values were left outside on each side of this range. Using the hand-fitted curve, it is possible to approximately estimate the average drawdown for a given regional withdrawal rate. The results show that simulated drawdown increases faster with the increase of the withdrawal rate. As the recharge component remained constant during the simulations, the applied uniform withdrawal rate is accounted for mainly by capturing of the natural groundwater discharge.

Three distinct zones are evident (Figure 6.15). The first zone (a) covers the range of sustainable pumping rates of up to 24 Mm^3/year. In this zone, the simulated drawdowns are relatively low (<2 m), and increase slowly as pumping rates increase. The second zone (b) is characterized by increased withdrawal rates. It starts from the inflection point of the fitted drawdown/pumping curve, and extends to the point where the pumping rate equals the discharge rate. Although the average drawdowns may not seem to be very high under current conditions of groundwater use in south western Quebec, they are considered high because most of the pumping wells are shallow wells that intercept only the upper portion of the regional fractured aquifers. The third zone (c) displays non-sustainable pumping rates larger than 51 Mm^3/year, wherein the withdrawal rate would exceed the natural discharge rate to streams and rivers, a function of the natural recharge rate under given climate conditions. A pumping rate that exceeded 51 Mm^3/year would reduce the discharge to surface water by 20% to 40% and could eventually lead to the drying of some streams. This could result in critical water levels and in groundwater shortages. Furthermore many wells would need to be re-drilled to increase aquifer penetration.

Finally, transient simulations were conducted in order to obtain the long term piezometric trends. As rare information exists over the historic evolution of the groundwater withdrawal, these scenarios were run assuming that the entire extraction rate was applied at time zero. Figure 6.16 depicts the evolution of the simulated regional drawdown for the Chateauguay aquifer.Even after hundred years, the model does not reach steady state conditions resulting in the maximal drawdown of 1.48 m.

BOX 6-2 AQUIFER VULNERABILITY IN FRACTURED ROCK AQUIFERS

Like many communities situated in close proximity to urban centres, the southern Gulf Islands, located in the Georgia Strait between Vancouver and Victoria (Figure 6.17), are experiencing significant development pressures. Groundwater quality issues in the Gulf Islands have been amplified by improper disposal of agricultural waste, failed septic systems, pesticides and saltwater intrusion due to both natural conditions and over-pumping. The subdued topography of the Gulf Islands lends itself to the presence of few lakes able to support domestic and agricultural uses; therefore, the majority of residents rely on fractured bedrock aquifers as a primary source of fresh water.

The Gulf Islands are a group of 40+ islands that range in area from ~1-75 km^2 and are characterized by a generally northwest–southeast trend and elongation defined by linear ridges and valleys. Elevations range from 100 to 200 m, reaching a maximum of about 350 m on Saltspring Island. Coastlines are typically rocky, with either long expanses of low relief bedrock sloping shallowly into the ocean or, alternatively, steep cliffs and narrow rocky beaches.

The geology and hydrogeology of this region have been researched extensively (e.g., Allen et al., 2002; Allen et al., 2003; Mackie, 2002; Mackie et al., 2001; Journeay and Morrison, 1999;

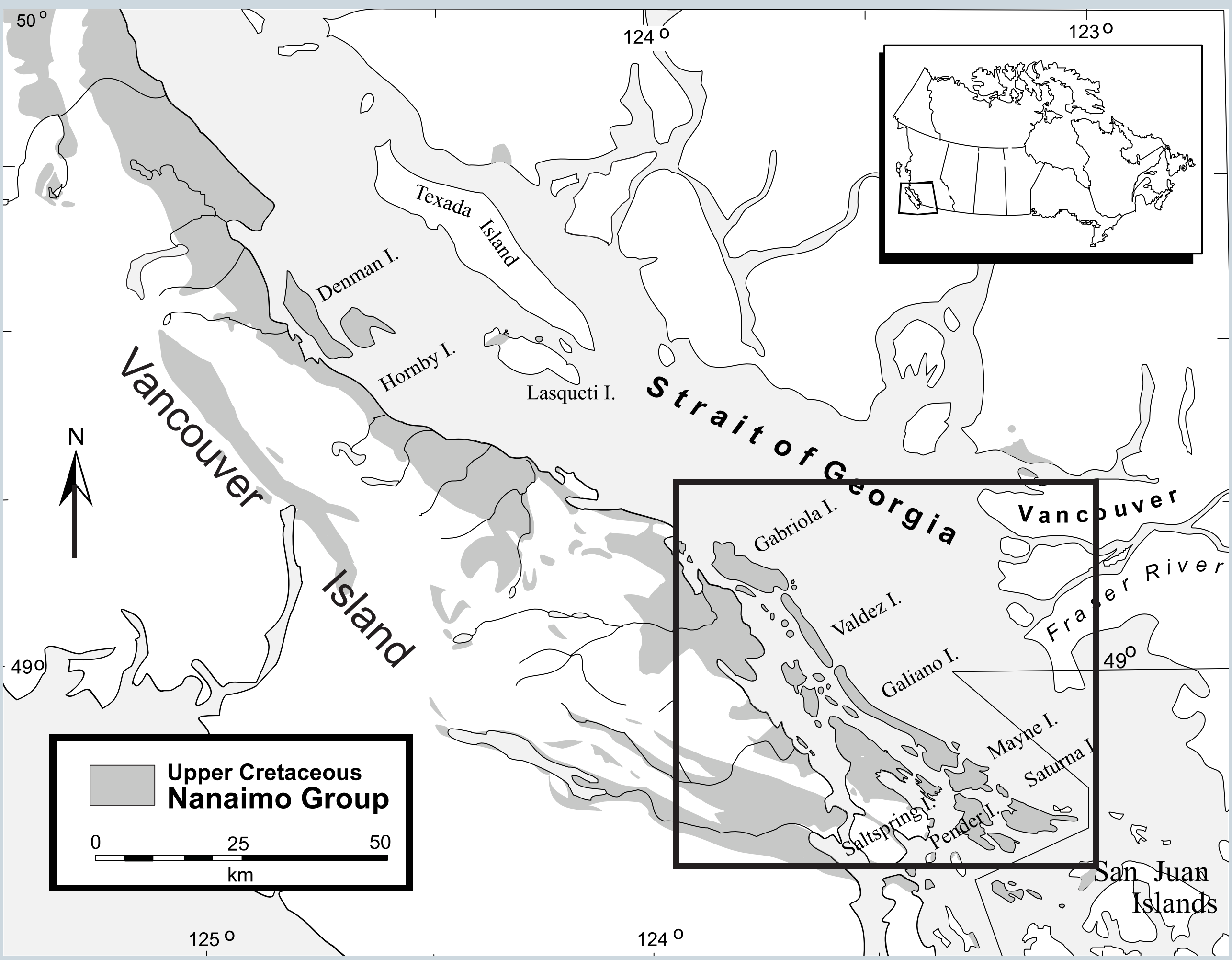


Figure 6.17 Location of the southern Gulf Islands in southwestern British Columbia.

England, 1990; Dakin et al, 1983; Hodge 1995). The Nanaimo Group (Mustard, 1994) forms the majority of bedrock of the Gulf Islands, and consequently, the majority of the water-bearing units on the islands. Unconsolidated deposits, of dominantly glacial and/or marine origin do not constitute a volumetrically significant percentage of the exposed geology on any of the islands, yet are anticipated to have a significant control on recharge.

The Nanaimo Group formations do not represent a true "layer cake" stratigraphy, but are composed of laterally thickening and thinning units with both conformable and sharp, erosive contacts. Lithology varies in grain size both between and within formations. Sandstone-dominated formations contain little structure, and can attain thicknesses of 100s of metres, with only minor fine grained interbeds. Silts and muds dominate mudstone formations, with significantly lower bed thickness (mm to cm). Structurally, the Gulf Islands are characterized by gentle folds with bedding that dips in the range of 5-15 degrees, with numerous small- and large-scale discrete fractures and faults. The present distribution of Nanaimo Group formations is the result of multiple regional deformational events (e.g., Journeay and Morrison, 1999). As well, the Gulf Islands have undergone glacio-isostatic deformation in response to multiple Quaternary glaciations (Clague, 1983), which have resulted in upwards of 50 m of vertical isostatic rebound.

Due to the low primary porosity and permeability of the solid bedrock, groundwater is derived primarily from fractures as secondary permeability. Higher joint densities characterize the more thinly-bedded mudstone-dominated units, notably within the transition zones between formations where mudstone bedding thickness is generally small (Mackie, 2002). This suggests that thinly-bedded mudstone-dominant units may have a higher permeability where they are in contact with sandstone. In contrast, the sandstone-dominated formations, with much lower fracture densities, may act more as impermeable blocks with significantly more widely spaced, discrete flow zones or pathways. In this respect, intra-formation heterogeneity, in the form of fine grain interbeds within coarse grain formations, may create pockets of more highly fractured rock, which, if connected to a recharge zone, may form an "intra-formation" aquifer. Similarly, at the contacts between formations, there may be enhanced permeability in those areas where there is transitional bedding.

Structures visible at the 1:75,000 (regional scale) can be characterized on the ground as fault or fracture zones up to 10's of metres in width. These structures are often identified by lineaments that are zones of high weathering or ridges. On the Gulf Islands it was found that fracture density tends to increase by at least a factor of ten in the presence of a regional-scale fault (Mackie, 2002). Many mesoscale fractures, which cross-cut all formations, were identified on the islands; these may represent discrete flow paths or narrow (metre-scale) flow zones. These fractures tend to be older than lineament-scale fault and fracture zones and may not have as significant an effect on groundwater flow at the island scale, but they may be important at the local scale.

From a hydrogeologic perspective, the larger fracture zone structures are interpreted to have a significant effect on groundwater flow, particularly at the regional scale. Discrete fracture modelling (Surrette and Allen, 2008; Surrette et al., 2008) indicates that the permeability of fracture zones is higher than that of the sandstone- and mudstone-dominant lithologies. Furthermore, Allen et al. (2003) found that flow in most wells located near mapped lineaments was highly influenced by linear flow and that the hydraulic properties calculated for wells situated near such features were consistently higher than those for wells away from lineaments. These observations support the interpretation that large-scale fault and fracture zones exercise a dominant control on the hydrogeology, and probably act as conduits for groundwater flow at the regional scale. Thus, vulnerability maps for this structurally-controlled region would best be represented with a conceptual model that captures permeability variations derived from fracturing at a range of scales.

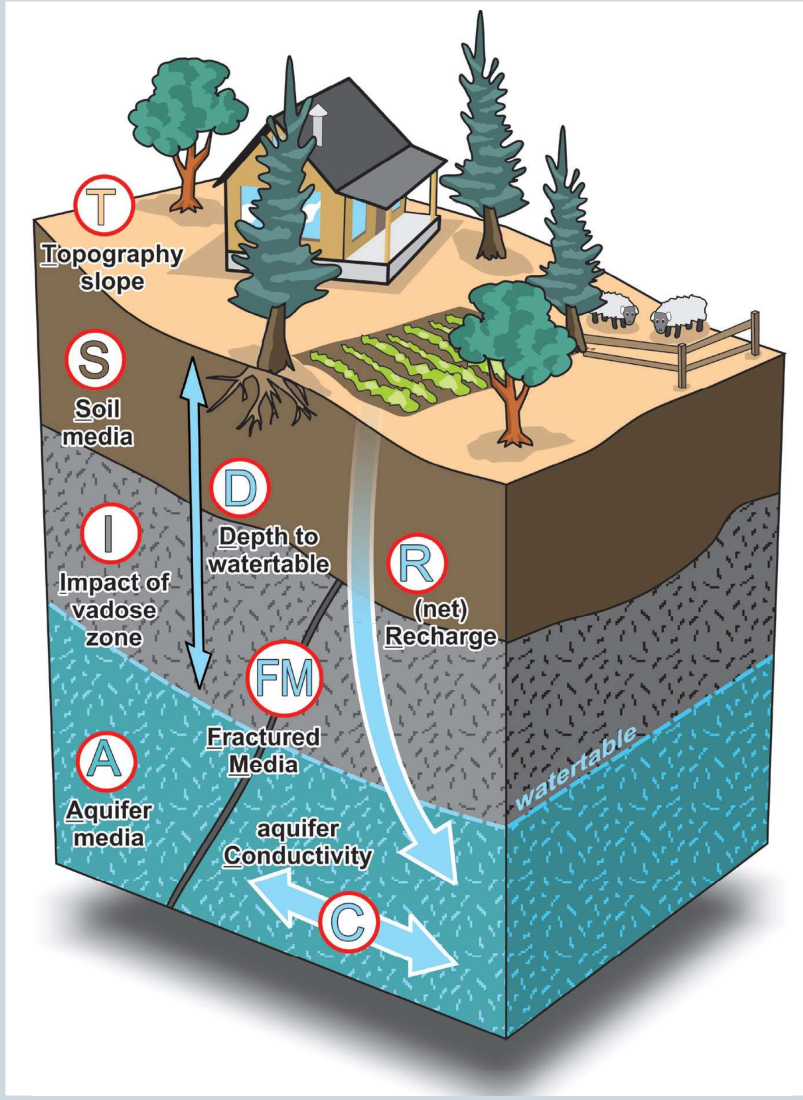


Figure 6.18 DRASTIC Methodology parameters. This figure differs from Figure 6.11 in that it incorporates the Fm parameter, which represented discrete fracture zones (figure courtesy of the Geological Survey of Canada).

A commonly-used aquifer vulnerability mapping method is DRASTIC, which parameterizes the physical characteristics that impact groundwater pollution potential (Aller et al., 1987). The term "DRASTIC" is an acronym for seven model parameters (Table 6.2, Figure 6.18). While the "A" (Aquifer media) parameter can incorporate the bulk effect of fractures on permeability, the spatial extent and characteristics of fault and fracture systems are not represented. To this end, a modified vulnerability mapping method, namely DRASTICFm, was developed, which identifies the impacts of structurally-controlled aquifers on the quality of groundwater resources (Denny et al., 2006). The Fm parameter takes into account three primary characteristics that dictate the impact of a discrete fracture network: orientation, length and fracture density (Singhal and Gupta, 1999). These three characteristics are combined into an eighth DRASTIC parameter and assigned the same weight as Aquifer Media (Table 6.2). DRASTICFm was determined to be the most representative model due to its capacity to synthesize the information sets available within the region, while identifying hydrogeological and hydrostructural trends between islands.

As an index-based model, DRASTIC assigns relative weights to each of its parameters. These weights are allocated based on a parameter's contribution to the overall susceptibility of an environment. Within each parameter, ratings are assigned to define the significance of one characteristic over another. Ratings for individual parameters were determined from direct consultation with the DRASTIC EPA manual (Aller et al., 1987) and from the application of DRASTIC to other study areas within similar environments in British Columbia (e.g., Wei et al 2004; Wei et al., 1998).

In order to properly represent the parameters within the DRASTIC methodology from a spatial context, a comprehensive collection of Geographic Information System (GIS) datasets was compiled. Key input datasets into this model include soil, bedrock geology, a water well database and a DEM. In order to bring consistency to the varying

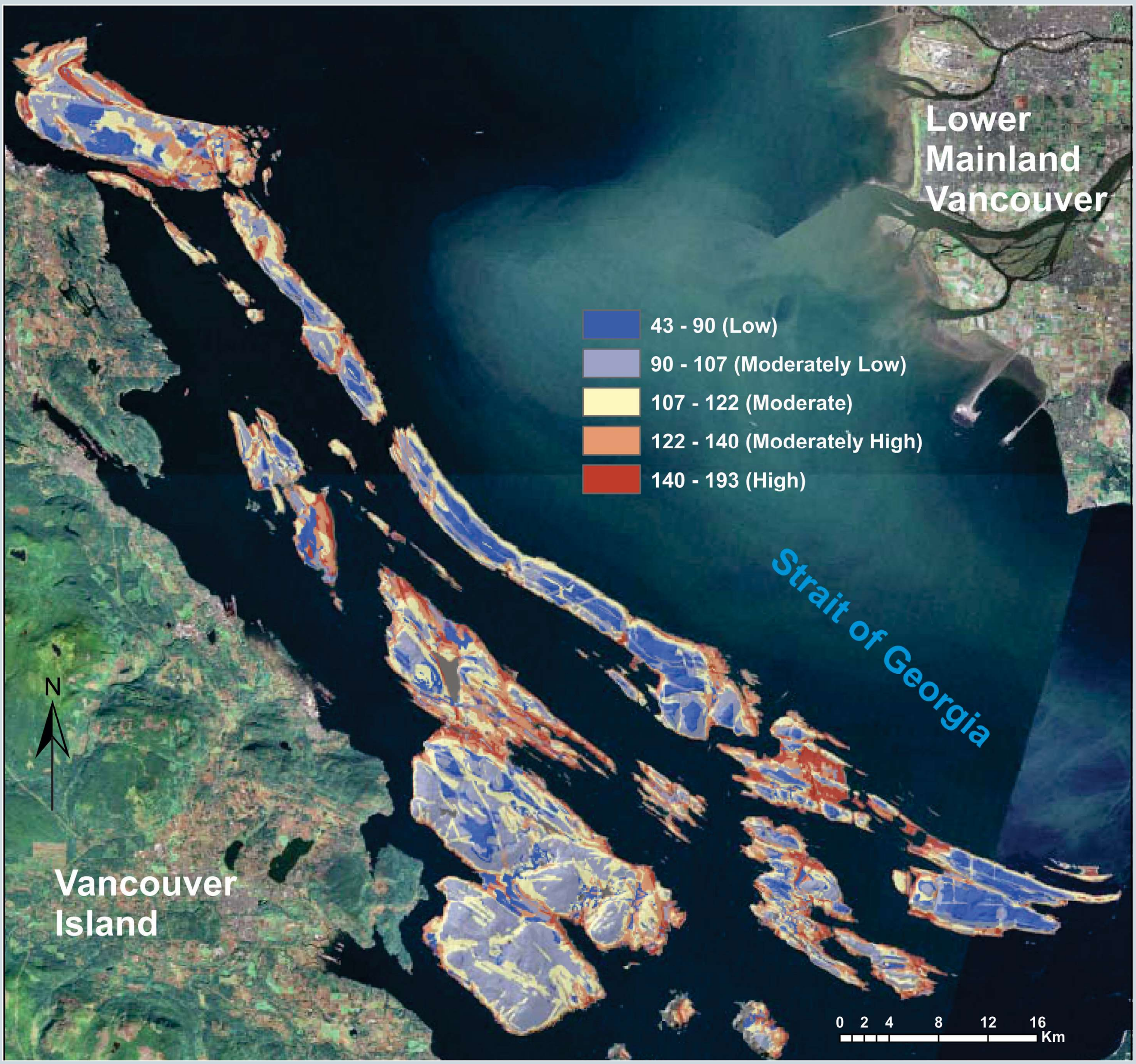


Figure 6.19 DRASTIC-Fm model output for the Gulf Islands.

TABLE 6.2 DRASTIC-FM PARAMETER DEFINITIONS AND WEIGHTS

Hydrogeologic Factor	Weight
D – Depth to Water	5
R – Net Recharge	4
A – Aquifer Media	3
S - Soil Media	2
T - Topography	1
I – Impact of Vadose Zone Media	5
C – Aquifer Hydraulic Conductivity	3
Fm – Fractured Media	3

scales of the input datasets, a constant scale was determined by the DEM (25m) and each of the layers was converted to a raster dataset. Each cell in the model output dataset is represented by a vulnerability value which corresponds to the cumulative rating of all parameters.

Model outputs were classified into 5 categories of vulnerability ranging from high to low with vulnerability rates ranging from 43 (low) to 193 (high) (Figure 6.19). General trends in the model outputs include regions of high vulnerability around island perimeters where instances of saltwater intrusion are prevalent, and in valley regions where the topography changes, recharge rates are high and structures are present. The model is quite sensitive to changes in the D (Depth to aquifer) and the presence of faults and fractures (Fm). Regions of moderate to low vulnerability (43-107) exist primarily in poorly drained soil layers with significant clay deposits. These regions occur primarily in the central portions of the islands where the thickness of material above the aquifer is greater than 10 m deep.

Regions of moderately-high to high vulnerability (107-193) exist primarily at the periphery of the islands and in areas of exposed rock where there is little or no soil material to provide a potential obstruction for a contaminant to move vertically into the vadose zone.

The overall impact of the presence of fault and fracture systems tends to augment the vulnerability of the regions within proximity to a structure. For example, the presence of faults and fractures within regions of low vulnerability increases the vulnerability range to moderately-low. This is particularly evident in the southern portion of Salt Spring Island and the central portion of Saturna Island where the presence of faults and fractures have augmented the vulnerability from moderate (107–122) to moderately high (122–140) and moderately low (90–107) to moderate (107–122), respectively. the impact of the fracture density on individual faults and fractures appears to have the most significant impact on vulnerability.

BOX 6-3 INTEGRATING GROUNDWATER SCIENCE INTO DECISION MAKING

(FROM LIGGETT ET AL., 2006)

The Okanagan Valley in south central British Columbia (Figure 6.20) has seen unprecedented population growth over the past few decades, and development continues to put pressure on water supplies in this semi-arid region. In addition, the Okanagan is one of Canada's primary agricultural areas, and a popular tourist destination. Most water supplies in the Okanagan are from surface water sources, but these are close to being fully allocated. Thus, there has been a growing interest in exploiting new groundwater resources. With an increase in demand, there will be a greater emphasis placed on strategies for groundwater protection. Likewise, sustainable community development is an important topic of discussion for communities, governments, and researchers. It is within the context of sustainable development and aquifer protection that this study took form.

In 2005–2006, Greater Oliver, situated in the south Okanagan, participated in the Smart Growth on the Ground (SGOG) initiative in support of sustainable community development. SGOG is a collaborative project between SmartGrowth BC, the Real Estate Institute of BC, and the Design Centre for Sustainability at the University of British Columbia (UBC). This team of researchers and facilitators works with communities to adopt necessary by-laws, programs and regulatory changes toward the development of a sustainable community design. Greater Oliver was chosen as one of three pilot study communities in BC. Other SGOG participants included Maple Ridge and Squamish. The primary outcome of this process was a concept plan that was developed as part of a participatory planning process culminating in a design charette[5]. The concept plan outlines recommendations on growth patterns and environmental, social and economic priorities highlighted by representatives of community-identified stakeholder groups. Water scarcity and water quality were identified as key priorities during the initial phases of the SGOG process in Greater Oliver due to the projected population growth coupled with agricultural (fruit crops, vineyards) and recreational activities that rely on water. Population growth and subsequent development in the valley are inevitable. To ensure that development occurs in a sustainable manner, a community must identify where development will occur and what sustainable measures (proximity to public transit or rain catchment systems, for example) will be adopted into new development projects.

Various tools are used to facilitate the planning process. The LUAM (land use allocation model) is a land use planning tool that prioritizes locations of new growth based on constraints and indicators identified by the community (e.g., Cromley and Hanink, 1999). The LUAM criteria for Oliver were grouped into several classes, including land use policy, land cost, amenities, infrastructure, market proximity, physical characteristics, hazard, health, and ecological setting according to specific criteria. The LUAM for Oliver was created in a commercial suite of "what-if" scenario modelling and landscape visualization tools called CommunityViz™ (Placeways, LLC, 2010). This software integrates available information, knowledge and community values in real-time to identify trade-offs and consequences between different prospective land use planning scenarios. LUAM outputs identify regions of desirable future development. Each grid cell represents the cumulative ranking of all criteria multiplied by weightings that reflect community values.

Vulnerability maps represent an effective means of synthesizing complex geologic and hydrogeologic information so it they can be used by planners and policy-makers toward the development of sustainable resource management plans and future growth strategies (Aller et al. 1987). An aquifer vulnerability map was created for Electoral Area C in Greater Oliver using the DRASTIC method (Aller et al., 1987) (Figure 6.20). Details concerning the methodology can be found in Liggett et al., (2006). Ultimately, all seven DRASTIC parameters

5. A charette is a planning technique that takes place over several days and allows all community decision-makers (municipal officials, developers and local residents) to be together at the same time to find solutions to known issues and develop a sustainable plan for their community. The Oliver charette took place in May 2006.

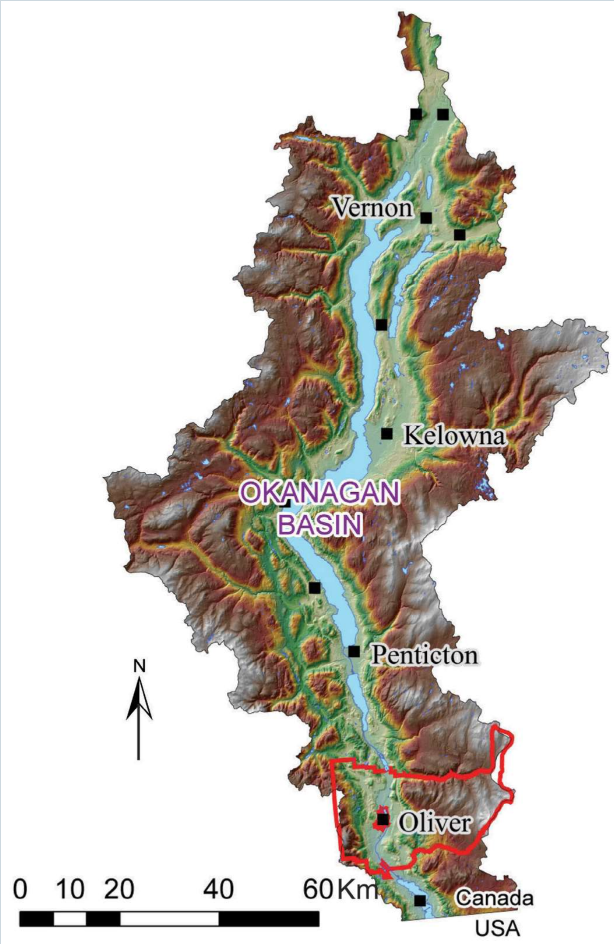


Figure 6.20 Okanagan Valley, B. C. Outlined area is Greater Oliver.

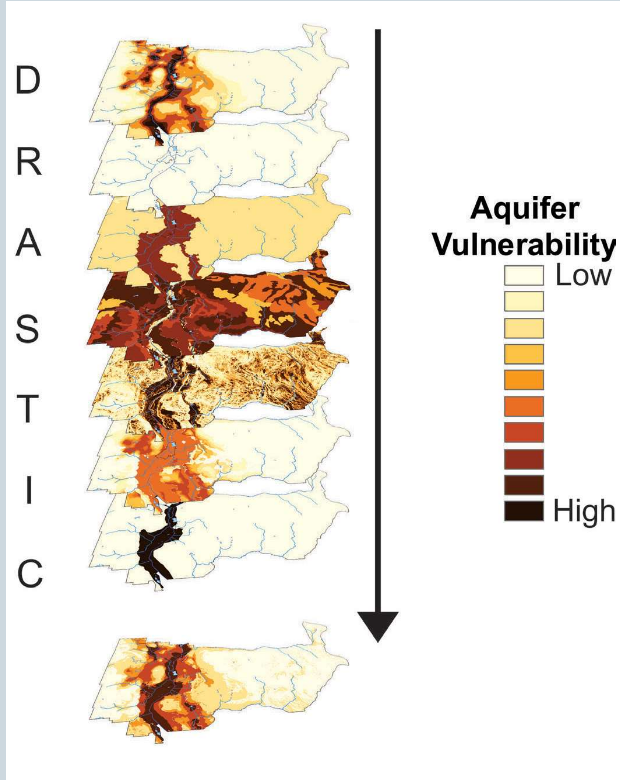


Figure 6.21 Seven DRASTIC input characteristics for the Greater Oliver aquifer vulnerability maps. The lowest map in the sequence shows the final DRASTIC map constructed by weighting the seven input maps—also reproduced in Figure 6.22.

were mapped in ArcGIS 9.1 (ESRI, 2005) and converted into raster format. The seven raster maps were multiplied by their respective weights and added together (Figure 6.21) to produce the final, spatially distributed, map of intrinsic aquifer vulnerability (Figure 6.21). Vulnerability ranges from 35 to 171 of a possible 230. Generally, the sand and gravel aquifers in the valley bottom are more susceptible to contamination than the igneous and metamorphic aquifers in the valley uplands. The shallow depth to water in the valley bottom, along with high ratings assigned to aquifer media and aquifer conductivity result in a highly vulnerable valley bottom aquifer.

Aquifer vulnerability provided an important

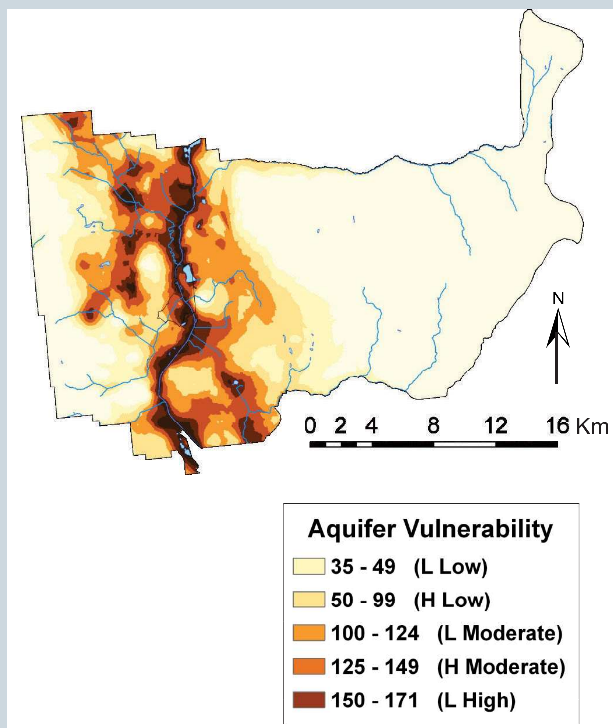


Figure 6.22 Relative intrinsic aquifer vulnerability in the Greater Oliver area.

constraint for the LUAM. It was used to identify the likelihood of groundwater quality issues in future residential or commercial settlements. It is anticipated that the positive results of this process will highlight the importance of using LUAMs, and datasets such as aquifer vulnerability maps, to represent all facets of land use planning in support of sustainable community development.

BOX 6-4 DELINEATING WELLHEAD PROTECTION AREAS—COMPARATIVE STUDY OF METHODS

Human activities, whether agricultural, industrial, commercial, or domestic, can contribute to groundwater quality deterioration. In order to protect the groundwater exploited by a production well, it is essential to develop a good knowledge of the groundwater flow system and to delineate the area surrounding the well within which potential contamination sources should be managed. Such an area is referred to as the wellhead protection area (WHPA). Regionally, the protection of groundwater resources involves land management and restrictions on potentially polluting activities in more vulnerable areas, especially recharge zones. This assessment of aquifer vulnerability is made on the basis of regional hydrogeological mapping.

The U.S. EPA (1991b) defines a WHPA as "the surface and subsurface area surrounding a water well or well field, supplying a public water system, through which contaminants are reasonably likely to move toward and reach such water well or well field." This zone can also be referred to as the zone of contribution (ZOC), i.e., the two-dimensional (2D) projection to the land surface of the aquifer volume containing all the groundwater that may flow toward a pumping well over an infinite time period. The zone of influence (ZOI) is the cone of depression caused by pumping. The zone of travel (ZOT) is defined within the ZOC and can be described as an isochrone indicating the time necessary for water or a conservative contaminant to reach the well from that location (Figure 6.23)

Several methods exist for delineating WHPAs; these differ in their degree of complexity and their precision. Naturally, the integration of more geologic and hydrogeologic characteristics of the study area increases the precision of any given method (Barlow, 1994; Livingstone et al., 1996; Bair and Roadcap, 1992; Ramanarayanan et al., 1992). From a practical perspective, the most appropriate method for WHPA delineation should be the one that simplifies the flow system as much as possible while still preserving its geological and hydrologic characteristics.

Example of WHPA delineation

Several studies have been conducted to compare WHPA methods (Springer and Bair, 1992; Forster *et al*. 1997; Bates and Evans, 1996; Paradis et al., 2007). The recent study from Paradis et al. (2007) has provided a comparison of methods for WHPA delineation in order to identify an efficient method that will be easy to use and cost-effective as well as providing a realistic delineation of the WHPA in the alluvium context. Methods selected range from simple approaches to complex computer models and include: calculated fixed radius (infiltration method), uniform flow equation (Todd, 1980), time of travel (TOT) equations (Bear and Jacob, 1965), HYBRID method (Paradis, 2000), flow system mapping, semianalytic method WhAEM (Haitjema et al., 1994), and the numerical model MODFLOW-

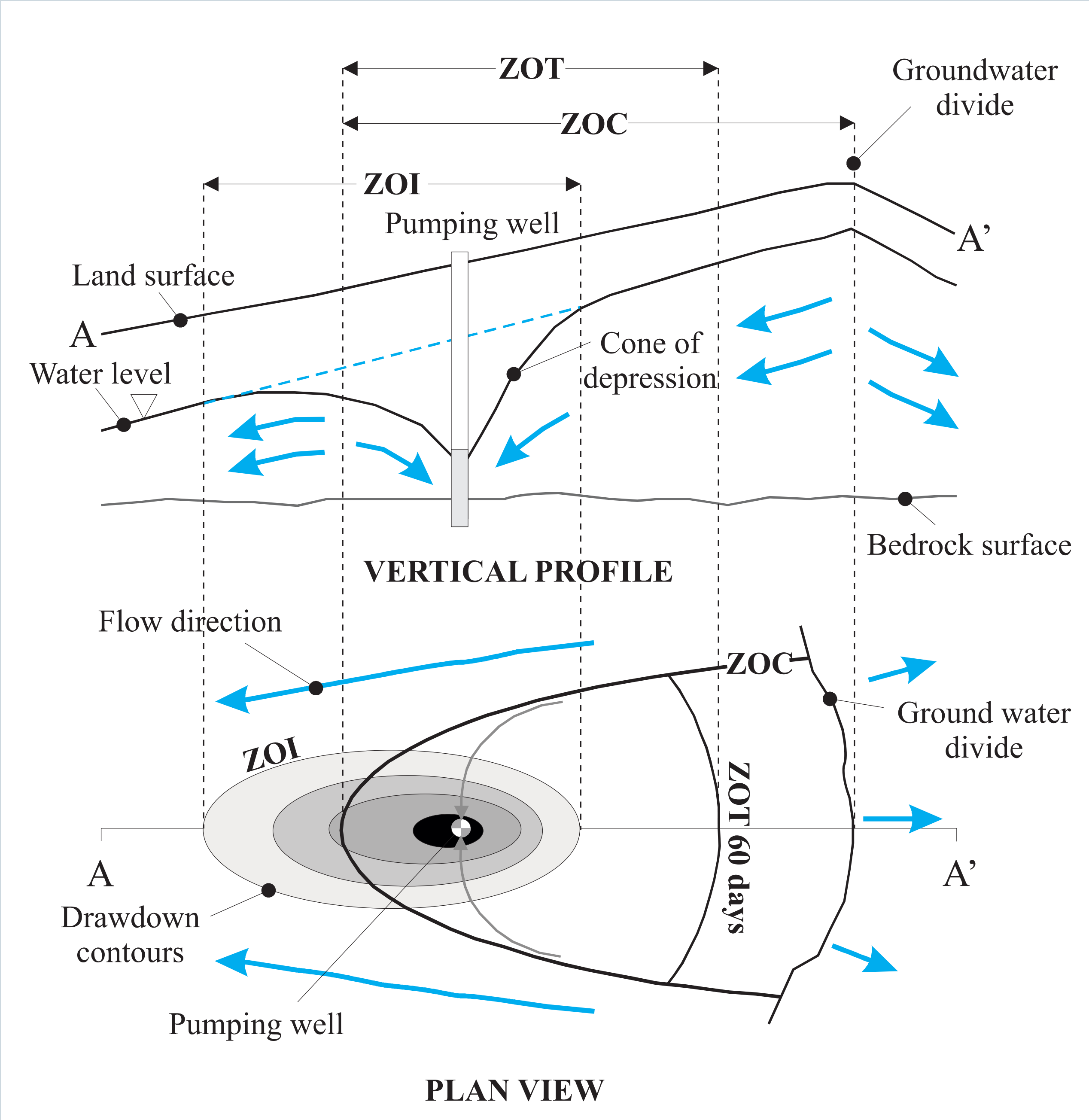


Figure 6.23 Relationship between zone of influence (ZOI), zone of transport (ZOT), and zone of contribution (ZOC) in an unconfined porous-media aquifer with a sloping regional water table (modified from USEPA, 1987).

MODPATH (McDonald and Harbaugh, 1988; Pollock, 1989) (Table 6.3).

For comparison purposes, all of these methods were used to calculate the WHPA of a test-bed site composed of an unconfined granular aquifer located in a thick sequence of deltaic and littoral sand deposits ranging from 10 to 30 m depth underlain by the Precambrian rocks of the Canadian Shield (Fagnan et al., 1999). This sandy aquifer features a series of small terraces which slope both sides of water divide, to the Aux-pommes River to the south and to the Jacques-Cartier River to the north

TABLE 6.3 WELLHEAD PROTECTION AREA (WHPA) METHODS CHARACTERISTICS
(FROM PARADIS ET AL., 2007)

METHOD OF WHPA DELINEATION	PARAMETERS	ADVANTAGES	DISADVANTAGES
Mass balance: Infiltration equation (USEPA, 1987) Cylinder equation (USEPA, 1987)	* Recharge * Time of travel * Pumping yield * Porosity (specific yield) * Saturated aquifer height/thickness	* Low cost * Easy and fast to use with less data * Low technical knowledge required	* Independent of specific flow condition * Over simplification
Analytical : Uniform Flow (Todd, 1980) TOT[1] (Bear and Jacob, 1965) TOT[1] (Darcy's Law)	* Hydraulic gradient * Hydraulic conductivity * As for Mass balance except * no recharge	* *idem* Mass balance	* *idem* Mass balance
Semi-analytical: WhAEM (Haitjeima et al., 1994) CAPZONE-GWPATH (Bair et al, 1991; Shafer, 1990)	* *idem* Analytical * Simple flow limits * Limited parameter uncertainties	* Based on idealized setting * Useful for simple flow system and recharge (WhAEM) * Non-uniform regional flow field may be superimposed (CAPZONE)	* Isotropic and homogeneous conditions * Infinite extent aquifer assumption * Complex recharge not directly taken into account
Hydrogeologic Mapping: Potentiometric (flow system) map	* Physical and hydraulic limits	* Economic and precise for shallow granular aquifer * Often the only method useful in karst and fractured media	* Must be combined with other methods * Not quantitative * Does not represent TOT * Expensive for complex settings
Combined methods: HYBRID (Paradis, 2000)	* *idem* Mass balance and Analytical * Physical and hydraulic limits	* Low cost * Easy and fast to use for simple aquifer * Good precision for homogeneous and isotropic aquifer	* Imitative form of WHPA in homogeneous and isotropic setting
Numerical Modelling: MODFLOW-MODPATH (McDonald and Harbaugh, 1988; Pollock, 1989)	* Every hydrogeological parameter * Physical and hydraulic complex limits represented * Parameter uncertainties	* May represent most hydro-geological settings * Quantitative and predictive tool	* Required good conceptual model * High data and expertise needed * May require high computational effort

[1] TOT = Time of travel

(Figure 6.24). The result is a narrow aquifer that extends several tens of kilometres long and only 1 or 2 km wide (Fagnan et al., 1999). The underlying bedrock constitutes the impermeable base of the aquifer. This aquifer serves as the water supply for the town of Pont-Rouge which consists of 20 drive-point piezometers.

The reference ZOC for the site was delineated from a potentiometric map based on the summer water levels. This map was drawn using spatial interpolation of more than 300 water level measurements taken within the granular aquifer. The map considered the interaction between ground and surface water and the physical

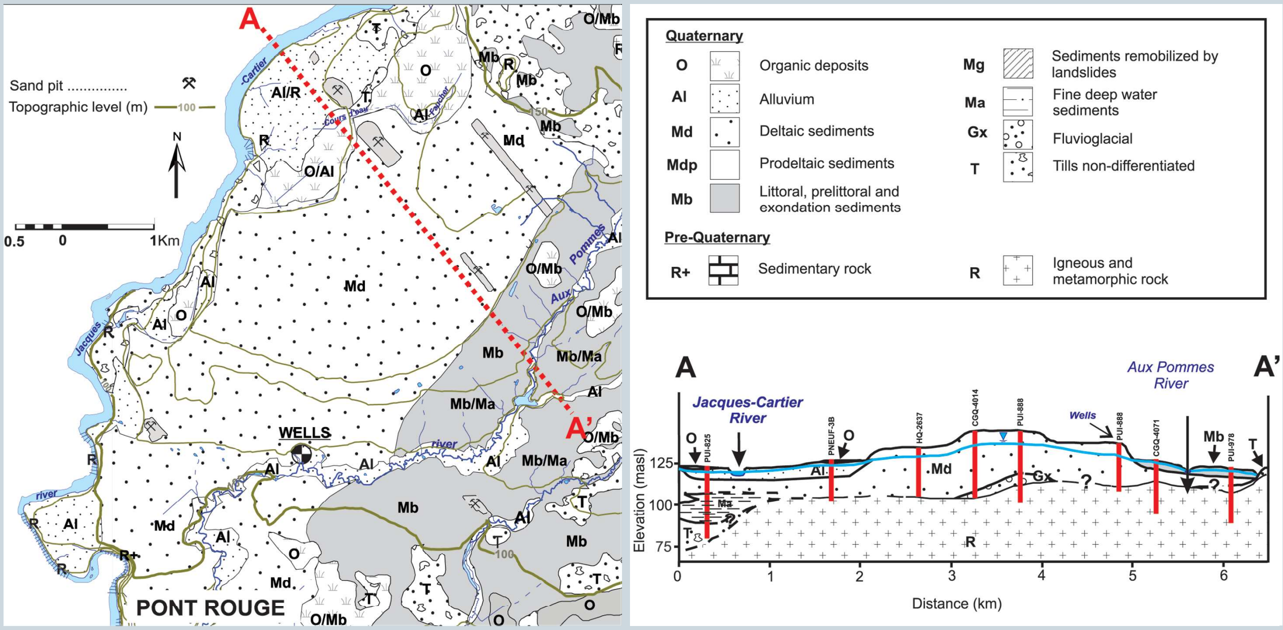


Figure 6.24 Geological map and cross-section of the aquifer.

boundaries such as rock outcrops and seepage faces. Moreover, the total area of the ZOC was constrained by a water balance between the water supply extraction rate and aquifer recharge. Table 6.4 shows hydrogeological parameters used for ZOC delineation.

Comparison of results

Figure 6.25 shows that the majority of the methods depicted a ZOC up-gradient from the pumping wells, and extending all the way to the groundwater divide. Only the cylinder infiltration method presented a ZOC equally distributed around the pumping wells. It also appears that the ZOC width as defined by the uniform flow and WhAEM methods was too narrow in comparison with the reference method provided by the potentiometric mapping technique. Consequently, only the potentiometric mapping, MODFLOW/ MODPATH and the HYBRID methods provided realistic delineation of the ZOC. One of these three methods (or all of them) may be applied,

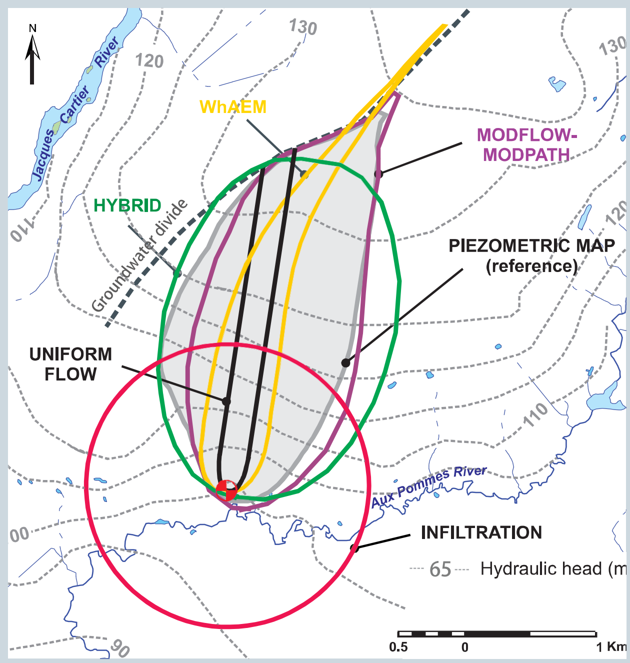


Figure 6.25 Comparison of the ZOCs at the Pont-Rouge site (modified from Paradis et al., 2007).

depending on the amount of data and the effort available. Generally, however, the most simple and cheapest method is applied first and should more precision be needed, or a counter-verification required, the more expensive methods are then utilized.

TABLE 6.4 HYDROGEOLOGICAL PARAMETERS USED BY WHPA METHODS FOR THE ZONE OF CONTRIBUTION (ZOC) OF PONT-ROUGE'S WATER SUPPLY. METHOD OF REFERENCE IS IN ITALICS AND ADJUSTED DATA ARE IN BOLD. (MODIFIED FROM PARADIS ET AL., 2007)

METHOD FOR ZOC AT PONT-ROUGE	YIELD (M^3/D)	SATURATED THICKNESS (M)	HYDRAULIC GRADIENT (%)	HYDRAULIC CONDUCTIVITY (M/D)	RECHARGE (MM/Y)	EFFECTIVE POROSITY (%)	NOTE
Hydrogeologic mapping							Up and down gradient limits obtained with piezometric map
Uniform Flow	2603	12.5	1.1	75			
Infiltration	2603				254		
HYBRID	2603				254		Up and down gradient limits obtained with piezometric map and total WHPA area using infiltration equation
WhAEM		*1		**17**	**3750**[*2]	20	*1 Base aquifer elevation fixed at 80 m *2 recharge artificially increased on one side of the model to overcome the limitation that the model cannot accommodate sloping aquifers
MODFLOW-MODPATH	2603	*3		**5-35**	**254**	20	*3 Variable base aquifer elevation obtained with boreholes interpolation

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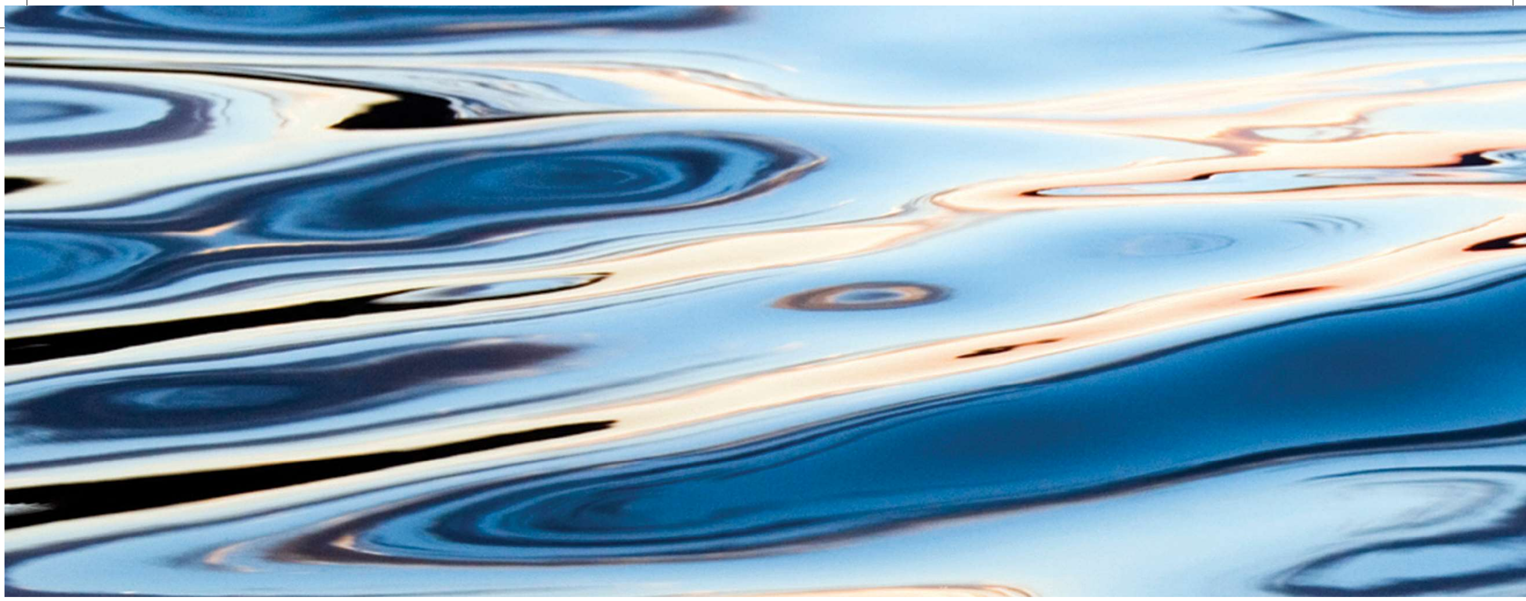

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