A GROUNDWATER BALANCING ACT

Environmental Flows and Levels for Groundwater-Dependent Fens of the Antelope Grazing Allotment,

Fremont-Winema National Forest, Oregon

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Note
The following report is an abbreviated version of the full suite of analyses complete in making this EFL determination. To see the longer report, please contact Allison Aldous (aaldous@tnc.org) or Leslie Bach (lbach@tnc.org).

Photo credits, cover page. Clockwise from upper left. Jamison Fen, Michelle Blackburn, *Drosera anglica* Allison Aldous/TNC, Forest Service staff at Dry Fen, Allison Aldous/TNC, *Pedicularis groenlandica*, Allison Aldous/TNC.

Contents

Introduction

Groundwater-dependent ecosystems (GDEs) include wetlands, lakes, rivers, springs, estuaries and off-shore marine environments, subterranean ecosystems, and some areas of specific terrestrial vegetation such as phreatophytes, as well as the many species that rely on groundwater to meet part or all of their water requirements (Brown et al. 2010; Eamus and Froend 2006; Sinclair Knight Merz 2011). At the same time, groundwater discharging to GDEs often is tapped or altered to meet human needs, including municipal, agricultural, domestic, and industrial water supply.

A key to protecting GDEs is to determine the amount, timing, and quality of discharging groundwater that they need, and to set limits to what is available for other uses (Aldous and Bach 2014). This requires a robust methodology for determining the groundwater needs of ecosystems that is straightforward to implement, monitor, and adapt to a variety of management situations. The Nature Conservancy and the U.S. Forest Service are developing such a method to make management decisions related to groundwater development that are protective of GDEs and seek to meet societal needs. This method is termed ***Environmental Flows and Levels (EFL*)**, and is defined as follows (eFlowNet 2007):

> "*Environmental flows and levels describe the quantity, quality, timing and range of variability of water flows and levels required to sustain or restore freshwater and estuarine ecosystems and the functions and services they provide. Environmental flows and levels include instream flows, geomorphic and flood flows, groundwater levels, and lake and wetland levels established for environmental purposes*".

The steps to setting EFLs for a GDE are as follows (Figure 1):

1. **Characterize and describe the GDE study area and management context**. This step is intended to set the stage for the EFL analysis. Key parameters should include any information necessary for evaluating whether the management activity is likely to affect the GDE(s). Study area parameters include the type and boundaries of the GDE(s); climate, physiography, expected hydroperiod or hydrograph; dominant plant communities and any other landscape characteristics. The management context parameters includes information such as current or planned water developments, diversions, contamination, or other hydrologic or water quality alterations that may affect the GDE; endangered or invasive species issues; or any other relevant management issues.

2. **Quantify the hydrogeology of the GDE**. Data and information at varying levels of complexity are used to investigate the effects of the management activity on the GDE(s), and for developing the EFL recommendations. The initial step is to characterize the hydrogeologic setting. This is followed by a water budget, and finally if deemed necessary, either analytical or numerical models. These data and analyses are used in combination to evaluate the likely effects of the management activity.

3. **Quantify the groundwater ecohydrology**. Identify the species and ecosystem processes dependent on groundwater flow or chemistry and then quantify the groundwater-ecology relationships.

4. **Develop thresholds**. Combine the information above to determine the groundwater thresholds or "tipping points" beyond which groundwater-dependent species and processes will be impaired.

5. **Evaluate groundwater management in relation to thresholds**. If the impacts to the GDE as a result of the management activity are expected to exceed viability thresholds of species and ecosystem processes, management or mitigation plans can be used to meet the needs of the GDE and of society. If the impacts are not expected to exceed the viability thresholds, monitoring is used to test the accuracy of the results.

This method was developed and tested in three groundwater-dependent wetlands (fens) in a grazing allotment on the Fremont-Winema National Forest, Oregon. Each of these steps is summarized below in the context of those field sites. Much of the data collection, modeling, and analyses are conducted simultaneously so that hydrogeologic results inform ecological analyses, and vice versa. The method allows an EFL to be determined using differing levels of hydrogeologic analysis, depending on site complexity, significance of the management action, and level of uncertainty.

The following summary is based on a full report available on TNC's Conservation Gateway site (http://nature.ly/AntelopeEFL). Many of the methods used here are described in more detail in the full report as well as in other sections of the Technical Guide.

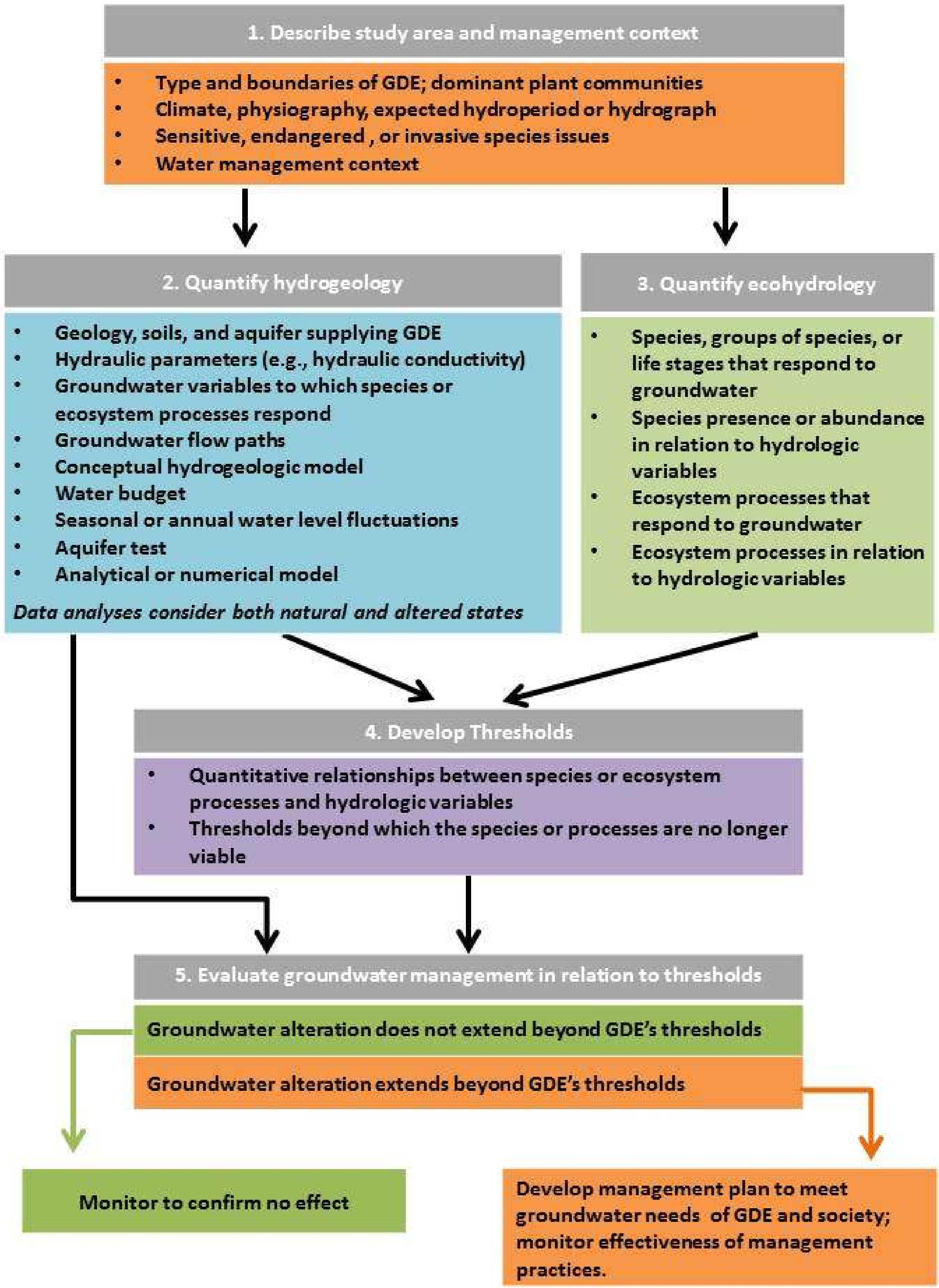

Figure 1. Method for assessing Environmental Flows and Levels of GDEs.

1 Study Area and Management Context

The study area is located east of the Cascade Range in the Basin and Range province of south-central Oregon on a broad upland bounded by Walker Rim escarpment to the west and the Fort Rock Valley to the east. The three study sites are located in the Antelope grazing allotment on the Fremont-Winema National Forest: Johnson Fen (0.35ha/0.87acres); Dry Fen (0.22ha/0.54acres); Wilshire Fen (0.19ha/0.48acres); and Round Fen (0.17ha/0.41acres)[1], all with slopes of 4-8% (Figure 2). They are small peat-accumulating wetlands fed by groundwater, termed fens, and used to water cattle during a three-month grazing period (July-September). This is done using gravity flow systems or solar pumps, whereby water is piped from a concrete spring box within each fen to watering troughs located outside the wetland boundaries. The goal of the Antelope EFL analysis was to determine how much groundwater, and at what rate and timing, could be pumped from fens in the allotment without damaging their ecological integrity. This work was conducted in part to aid in revision of the Antelope Grazing Allotment Management Plan (USDA Forest Service 2012).

The area has a semiarid climate, with the majority of precipitation falling during winter. Mean annual precipitation is 32-59 cm (13-23in) and mean annual temperature is 5.8C (42.4F), as reported by two local weather stations for the periods of record. The forest surrounding the fens is dominated by lodgepole pine with a sparse understory of grasses, forbs, and shrubs. The vegetation in the fens consists primarily of herbaceous wetland species, including forbs, sedges, a significant bryophyte understory, and wetland shrubs.

[1] Round Fen was only used in the vegetation analysis.

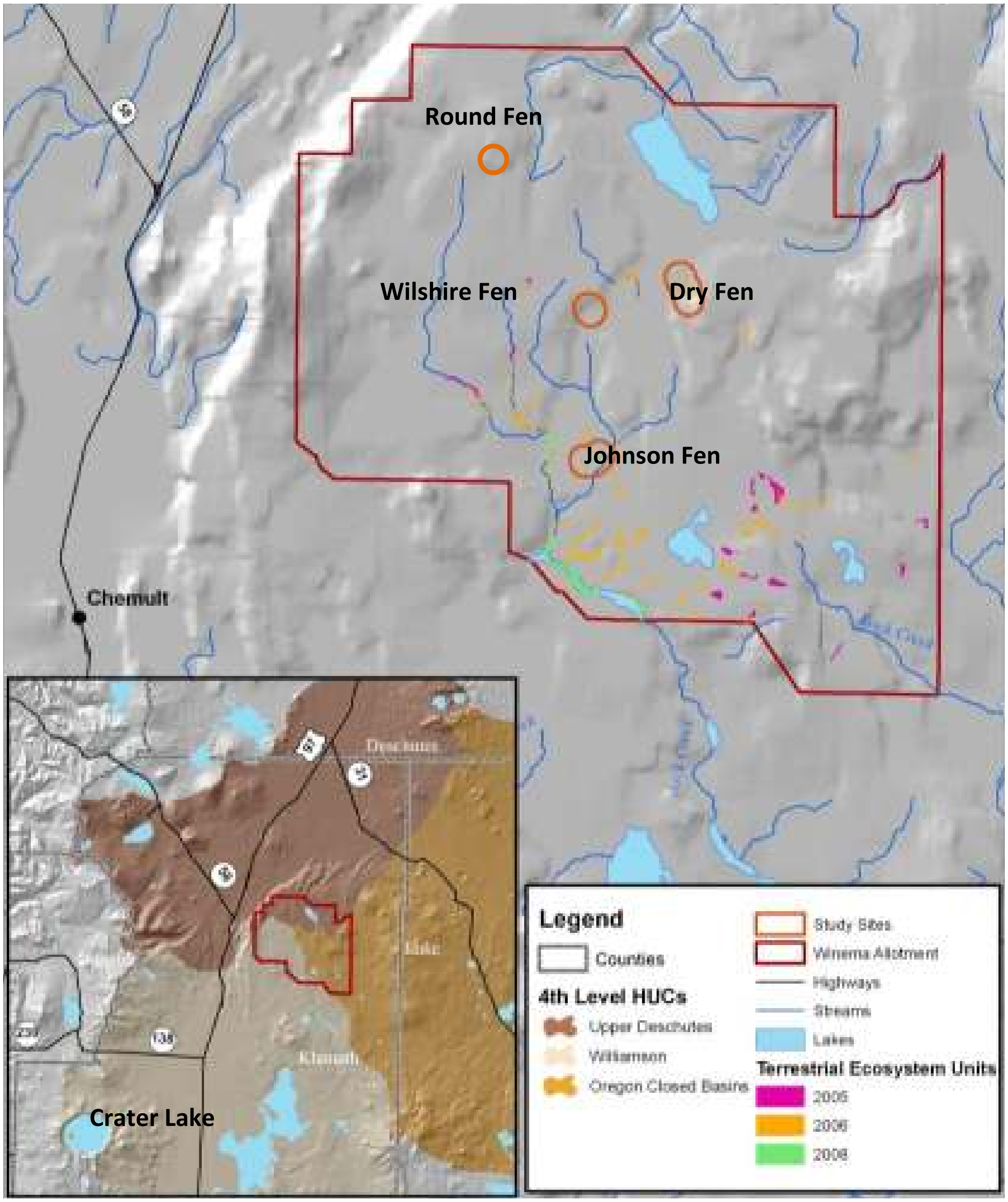

Figure 2. Map of allotment boundaries in the Chemult and Silver Lake Ranger Districts, Fremont-Winema National Forest, Oregon, as well as locations of the study sites, HUC boundaries, and Terrestrial Ecosystem Unit Inventory (TEUI) wetland land units.

2 Quantify hydrogeology

The groundwater flow system was investigated in various ways with the goal of evaluating the potential effects of the proposed groundwater withdrawals on the GDEs. This included a water budget, aquifer test, and analytical and numerical models. Because each of these analyses is associated with a certain degree of precision and accuracy, they were considered together in a “weight of evidence” approach to determine potential effects. Other EFL projects may employ fewer or different analyses to achieve the same outcome. In this section, supporting data sets are described first (soils, hydraulic properties, and shallow monitoring wells), which are used in the subsequent analyses, which are described next.

i. Supporting data

1. Soils

Methods:

Data on soils and shallow surficial geology, obtained from augered cores and boreholes, were used to construct the conceptual hydrogeologic model; for interpreting water level data; in developing the analytical and numerical models; and for developing the groundwater-ecology relationships for the process of peat accretion. Within the fen boundaries, we characterized the soils and shallow geology (i.e., pumice, ash, paleosols) down to bedrock using a hand auger (AMS 8 cm diameter mud auger). Soils and rock strata also were recorded uphill of the fens where deeper piezometers were installed in the bedrock. At each auger hole, we described horizons and recorded their depth, including the occurrence and depth of organic soils.

Results:

Auger holes extended to the paleosol and/or assumed bedrock (a depth of 2 to 3m) except in situations where soil properties (stoniness or loose consistency) prevented soil extraction with an auger. Auger holes in the fens show a consistent stratigraphy across the study area (Figure 3). The surficial layer is peat (organic soil) in varying states of decomposition, ranging from fibric peat at the surface to more hemic, and even sapric peat at depths of approximately 1m. Below that is pyroclastic ash and pumice of Mount Mazama (7627 $\pm$ 150 cal yr B.P.) that blanketed the study area up to 2 to 3 m in thickness (Borchardt et al. 1973; Zdanowicz et al. 1999; Bacon and Lanphere 2006). The pumice deposit is divided into lower and upper pumice units based on grain size, composition, and sorting. A narrow black, mucky layer at the top of the pumice deposit was noted in some of the augered holes and was assumed to create a leaky confining layer and artesian conditions in places.

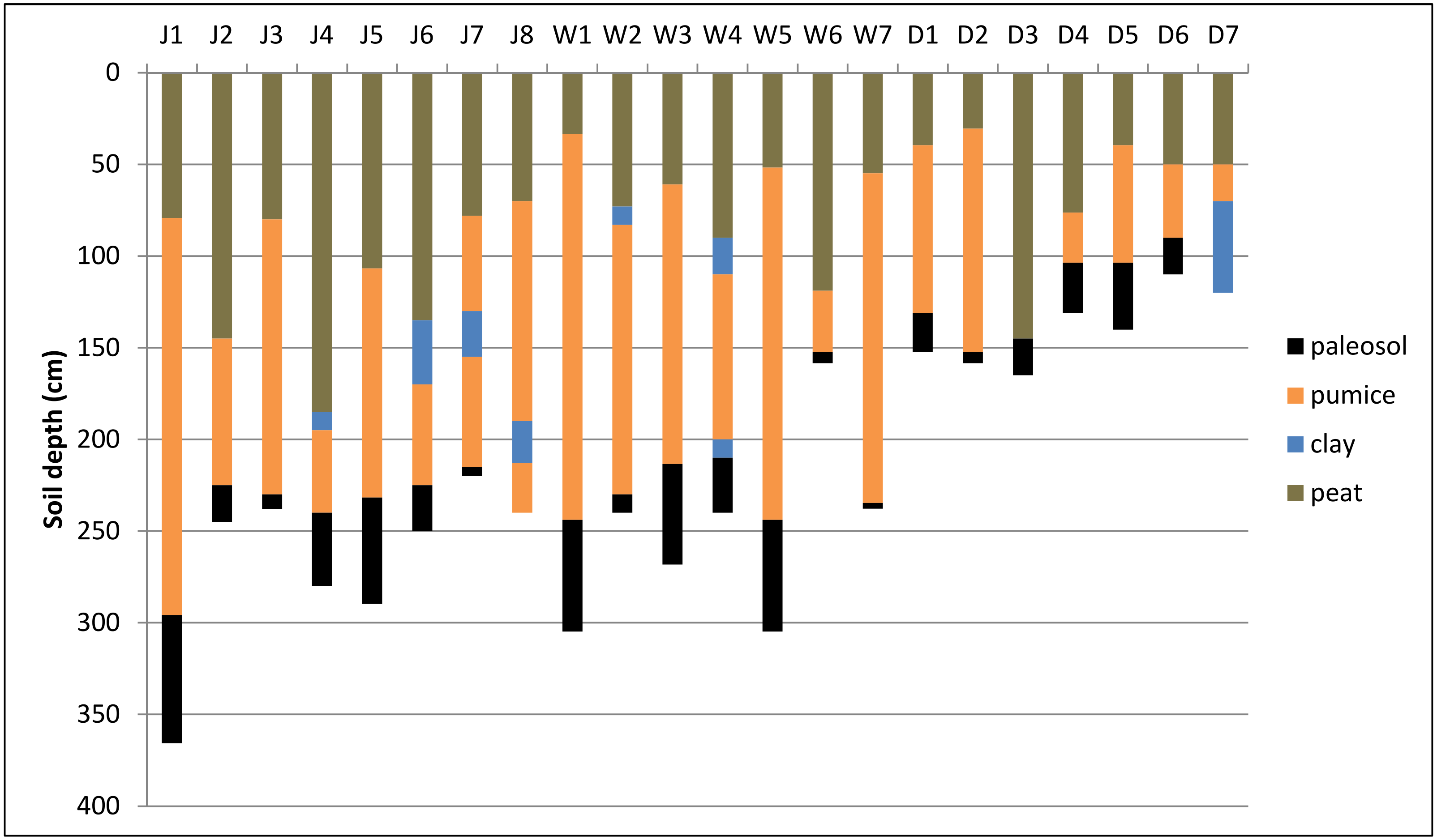

Figure 3. Soil profiles at well borings for the nested wetland piezometers, at Johnson (J1-J8), Wilshire (W1-W7), and Dry (D1-D7).

2. Hydraulic properties

Methods:

Single well slug tests were performed in Sep 2009 to estimate estimate the hydraulic conductivity of the peat and pumice layers. An aquifer test was performed to estimate various hydraulic characteristics (e.g., hydraulic conductivity) of the groundwater system. The test was done by stimulating the aquifer through constant pumping and observing its response (drawdown) in observation wells. Testing was conducted in July 2011 at the Johnson fen spring box. The Johnson fen spring box is a cement box with an open bottom approximately 60cm x 60cm (2ft x 2ft) in size and 1m (3.5ft) deep. The bottom intake zone of the spring box is located in the pumice below the peat layer and above the bedrock. Data were analyzed using the Hvorslev method (1951).

Results:

The average hydraulic conductivity of the peat and pumice layers from slug tests was measured at 0.24 m/d and 12 m/d[2] respectively. These average values were assumed to be valid for all three fens. The characteristics of the sapric confining layer at the base of the peat are unknown, so a reasonable value was assumed (0.001 m/d) (Bredehoeft et al. 1983).

For the aquifer test in Johnson fen a constant drawdown level of 28cm (0.92ft) in the spring box was established at about 300 minutes into the test. Drawdown in all observation wells was negligible. Fluctuations in water levels were less than 1cm and were within the range of measurement error. Because the three study fens are very similar in terms of hydrology, plant species composition, and soil properties, we assume that results obtained for the Johnson fen are applicable at the other study sites. The aquifer test shows that substantial water table elevation changes would not likely occur under an extraction rate of less than 0.0026 m^3/min (0.7 gal/min).

3. Monitoring wells

Methods:

Two kinds of wetland water table wells and piezometers were used to monitor water levels. The first type was large nested wetland piezometers, where individual wells within a nested set are located in close proximity to one another with each one measuring head in a different water-bearing zone. Each set of nested wells consisted of one water table well that is screened over the first 0.9m, and 1-3 additional open-ended piezometers, installed to 1m, 1.5m, or 3m. Each site had 3-9 sets of nested wells. The piezometers were 2" internal diameter schedule 40 PVC, driven into the peat and pumice using a steel drive rod (Sprecher 2008). The water table wells were factory screened, 2" diameter schedule 40 PVC pushed or pounded into the peat layer.

The second type was small shallow water table wells (Solinst™ schedule 40 PVC, 2 cm internal diameter and factory screened over their below-ground length), installed in a series of transects across the four fens in 2010. The transects were used to monitor the position of the water table as well as vegetation at the sites (described in section 4.1), and they were parallel to one another and spaced roughly 10m apart

[2] The k_{pumice} reported in Appendix 6, Table 1 is 21.2 m/day. However, the value 12 m/d was used for the model and flownet calculations after several high outliers were removed.

(Table 1). They also were perpendicular to the topographic gradient, assuming this followed the slope of the water table (which was later confirmed). Plots (50cm x 50cm) were installed along each transect with a random start point and then spaced approximately 5m apart. Because they were different total lengths, each transect contained a slightly different number of plots (Table 1). The water table wells were installed in a subset of the plots in one of the corners, to a depth of 30 or 60 cm. The wells were installed entirely within the peat soil, with the exception of upland plots, which had only mineral soil. Water table data were collected in approximately half of the vegetation plots, so we expanded the water level data set by extrapolating to some of the plots without a well.

Table 1. Number of transects, plots, and wells at each site

Site	# Transects	Total # Plots	Total # Plots w WT Wells
Dry	4	34	14
Johnson	5	43	25
Round	4	19	15
Wilshire	5	39	24

All water table wells and piezometers were monitored manually from their time of installation until Nov 2011. Several wetland water table wells were instrumented with pressure transducers which recorded water levels at 30 min intervals. The elevation of the water table wells and piezometers, spring boxes, and watering troughs at each site were surveyed using a transit level.

Results:

Nested piezometers in the fens show that groundwater occurs primarily within the pumice and peat layers (Table 2) and in some cases there is an upward gradient. This gradient indicates that upward groundwater flow into the plant rooting zone is occurring due to groundwater discharging into the fens. In some portions of the fens, groundwater levels are above ground surface, indicating that the peat layer provides some confinement for the groundwater system. Groundwater levels within the fens show some decline throughout the summer; however, levels remain relatively high (at or within a few cm of the ground surface) even at the end of summer when there is little or no recharge from precipitation.

Table 2. Mean, minimum, maximum water table elevations for the three fens, 2009-2010.

Site	Well #	Mean Head (cm)	Min Head (cm)	Max Head (cm)	Range in Head (cm)
Dry	WT	-12	-81	40	121
Dry	P1	-14	-49	36	85
Dry	P1.4	-10	-82	35	117
Johnson	WT	-0.6	-26	35	61
Johnson	P1	1	-28	26	53
Johnson	P1.4	-5	-91	22	113
Johnson (no J6, J8)	P1.4	4	-30	22	52

Wilshire	WT	-6	-26	32	58
Wilshire	P1	2	-61	31	92
Wilshire	P1.4	4	-17	31	48

The depth to water table along the vegetation transects show distinct differences between the middle of the fens and the edges. In the center of the fens, the water table tends to be high and stable. The transition to edge and upland water tables can be seen for entire transects along the edges of fens, as well as for some edge plots within transects (e.g., plot JV39 in Figure 4).

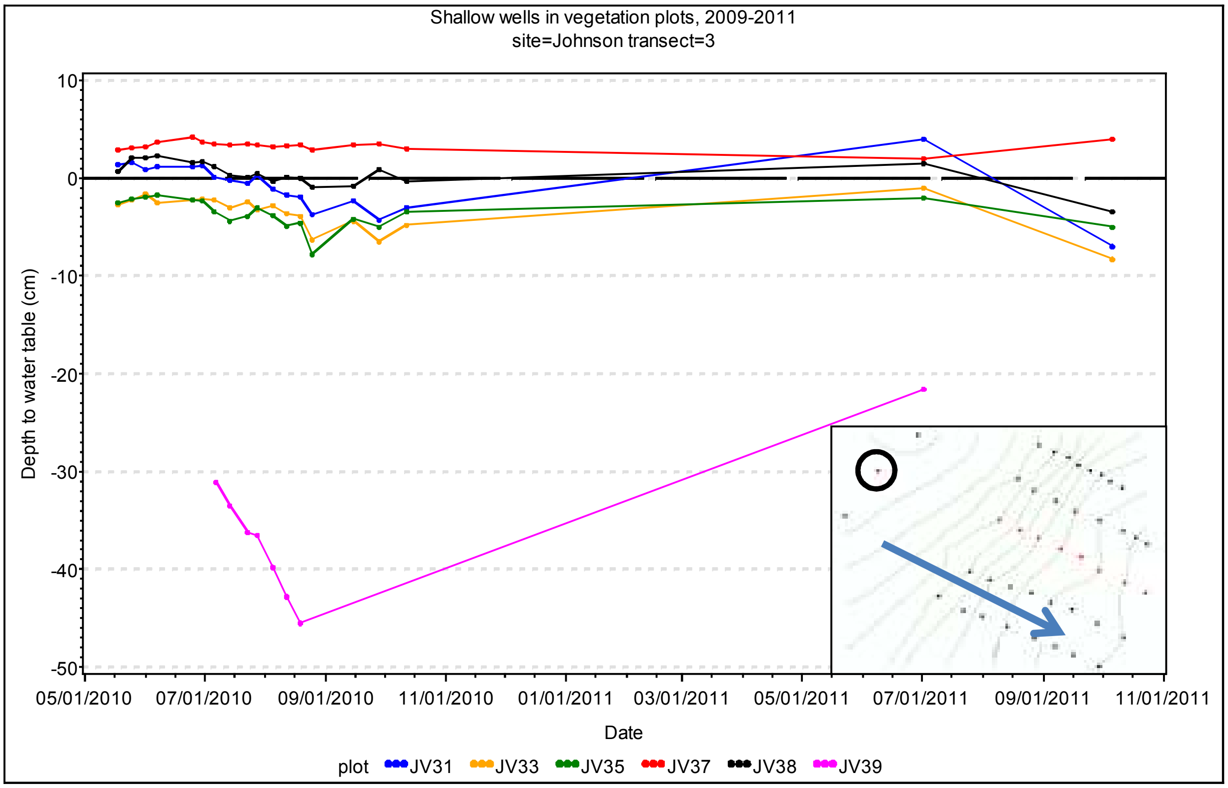

Figure 4. Depth to water table along transect 3 at Johnson fen. Inset map at lower right shows distribution of plots along the vegetation transects, where transect 3 is in red. Plot JV39 (red dot with black circle) is at the upper edge of the fen where the organic soil is very shallow (~5cm). Blue arrow indicates the direction of groundwater flow along the topographic gradient. Plot JV39 was installed after the others, so it has fewer measurements in early 2010. It also dried out by late summer, so had no measurements in fall 2010 or 2011.

ii. Conceptual hydrogeologic model

Methods:

A conceptual hydrogeologic model was developed based on best professional judgment and reconnaisance field visits. This method is used as a first approximation to determine if the study site is primarily groundwater-fed; however, the same information, collected with greater detail and accuracy, is also necessary for more complex methods. The types of information needed include the locations of groundwater recharge, flowpath lengths and contributing aquifer, and groundwater discharge into the wetland. Data to develop the conceptual model include maps of soils, geology, and topography, as well as simple field observations such as the presence of peat soils, known fen indicator species, and the position of the water table at different times of the year.

Results:

From available geologic maps, the area is shown to be underlain by late Miocene and Pliocene basalt to rhyolite lava flows, rhyolite domes, and silicic ash-flow tuff (McLeod and Sherrod 1992). Pumice deposits, from a series of eruptions of Mount Mazama (now Crater Lake) about 7600 years ago, overlie the area to a depth of two to three meters (McLeod and Sherrod 1992), and the blanketing of Mazama ash over the landscape greatly obscures the underlying geology.

The conceptual model shows the primary flowpaths from the basalt aquifer through the pumice and into the fen peat (Figure 5). Additional inputs are from direct precipitation and shallow groundwater discharge directly from the pumice. The major water outlet is ET. While pumping is not shown in this figure, that is expected to be a major outlet as well, and this could have detrimental effects to the fen species and habitats.

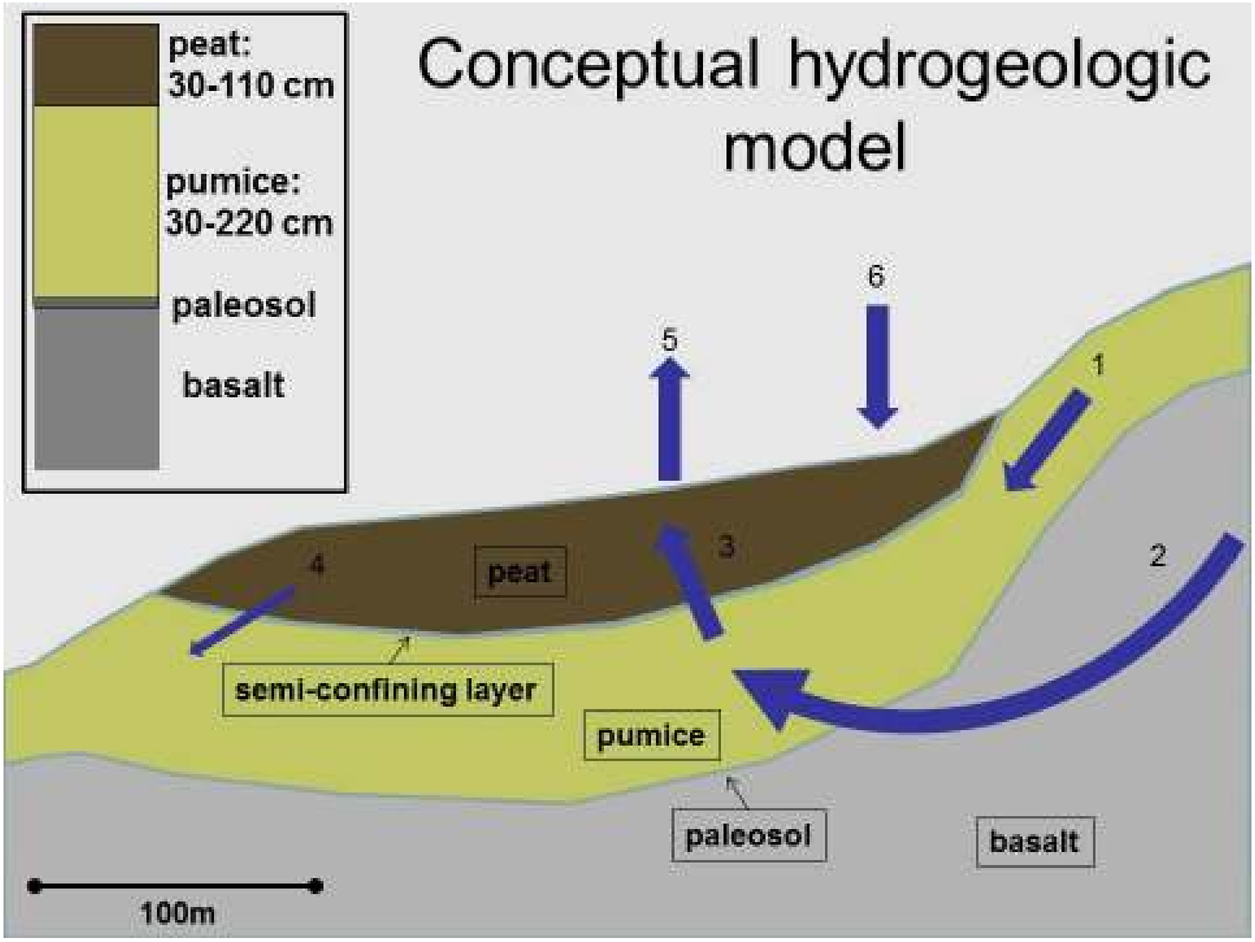

Figure 5. Conceptual hydrogeologic model cross section of the three study fens. Water balance elements are depicted as blue arrows: (1) shallow groundwater flow from local pumice deposits; (2) deeper groundwater flow through the basalt; (3) groundwater discharge from pumice to peat through leaky confining layer; (4) groundwater outflow from the fen to the pumice; (5) evapotranspiration; (6) direct precipitation.

iii. Water budget

A water budget is used to quantify the relative importance of groundwater compared to other water sources, and compare that contribution to the amount of water to be withdrawn:

Total Inflows (Direct Precip + GW Inflow) – Total Outflows (ET + GW Outflow + Pumping) = Change in Storage

We assumed little to no surface water inflow or outflow because these were rarely observed at the sites.

1. Precipitation

Methods:

Precipitation was recorded by an on-site micro-meteorological station that was installed at Johnson fen in June 2010 to measure the microclimate over the fen (precipitation, air temperature, air humidity, wind speed, solar radiation). Precipitation data also were available from nearby climate stations.

Results:
Precipitation recorded by the weather station at Johnson fen was 2.57cm over the 58-day grazing period. Mean annual precipitation measured at the Chemult and Timothy weather stations ranged from 32-59cm. Thus grazing-season precipitation is less than 10% of total annual precipitation, which confirms its seasonal distribution.

2. Groundwater Inflow

Methods:
To estimate the amount of groundwater flowing through the fen, we constructed flow nets and then used a form of the Darcy equation to calculate flow. This technique requires the following data: water level elevations from nested piezometers, hydraulic conductivity obtained from information on aquifer thickness from boring logs and slug tests.

Data from the large nested wetland piezometers were used to map the groundwater flow system within Wilshire, Johnson, and Dry fens. These maps show the configuration of the water table and groundwater flow direction. Heads from the water table wells were plotted on a plan view map and contour lines drawn. Flow lines were then drawn perpendicular to the equipotential lines. The total flux of groundwater through the fens was estimated using a flownet analysis. Once the flow net was constructed, the amount of groundwater flow through the area represented by the flow net, under steady-state conditions, was calculated using a form of the Darcy equation:

$$Q = \frac{mKHb}{n} \qquad (1)$$

where: Q = the quantity of groundwater flowing through the wetland, m = the number of streamtubes across a flow net, K = the hydraulic conductivity of the aquifer, H = total head drop across the area of interest, b = the effective thickness of the aquifer, n = number of equipotential head drops over the area of interest.

The K used to calculate the flux of groundwater through the fens is calculated as a weighted average for the pumice and peat layers that together function as a hydrogeologic unit using the following equation:

$$K_{wt} = \frac{K_{peat} b_{peat} + K_{pumice} b_{pumice}}{b_{peat} + b_{pumice}} \qquad (2)$$

Average K values from the slug tests were used to establish a K value for each layer. The thicknesses of each layer were average values from the boring logs.

Results:

Groundwater flux parameter values and final results are presented in Table 3. Groundwater flows from N to S at Wilshire fen, from NW to SE at Johnson fen, and from SW to NE at Dry fen. Flow nets also were plotted on vertical cross-sections, to show the vertical flow dynamics. An example of a flow net and vertical cross section is shown for Johnson Fen (Figure 6).

Table 3. Groundwater flux parameters used to calculate discharge rates for the flow net calculations. m = the number of streamtubes across a flow net; K = the hydraulic conductivity of the aquifer; H = total head drop across the area of interest; b = the effective thickness of the aquifer (peat, pumice, or the sum of the two); n = number of equipotential head drops over the area of interest; Q = the quantity of groundwater flowing through the wetland in metric and imperial units.

Site	**m**	**K** (m/d)	**H** (m)	$\mathbf{b_{peat}}$ (m)	$\mathbf{b_{pumice}}$(m)	$\mathbf{b_{total}}$ (m)	**n**	**Q** (m^3/d)	**Q** (gpm)
Wilshire	14	7.7	1.8	0.8	1.4	2.2	9	47.4	8.7
Johnson	9	6.1	1.4	1.2	1.2	2.4	7	26.4	4.8
Dry	13	5.5	2	0.65	0.53	1.18	10	17.2	3.1

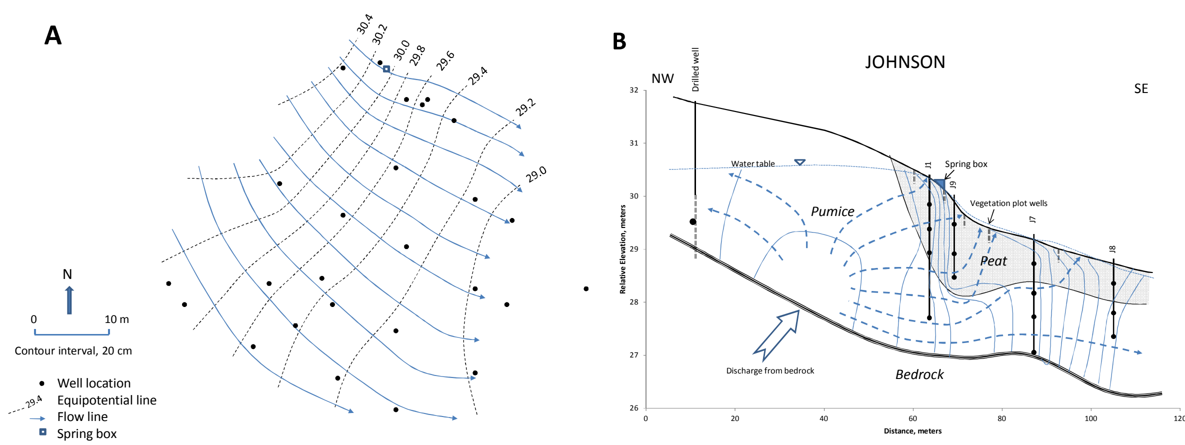

Figure 6. Flow nets generated to indicate flow of water through Johnson Fen based on water levels measured May 18, 2010. (A) plan view and (B) cross sectional view showing soils and geologic strata. Groundwater flow direction is indicated by flow lines (blue lines), and lines of equal hydraulic head (equipotential lines) are shown with dashed lines.

The spacing of the equipotential lines in Figure 6 gives an indication of variation in aquifer permeability values. Closely spaced contours, such as in the peat, are indicative of low permeability where a steep hydraulic gradient is needed to ‘push’ water through the aquifer. More widely spaced contours, such as upgradient in the pumice layer, indicate the aquifer is likely more permeable.

We calculated groundwater inflow to use in the water budget estimates, by converting the flux values estimated in the flownet calculations to depth by dividing by the area of the fen[3] and multiplying by the number of days (=58):

- Wilshire, Q=47.4m^3/day; area=1750m^2. Q/A*days=**157.1 cm**
- Johnson, Q=26.4 m^3/day; area=2500 m^2. Q/A*days=**61.2 cm**
- Dry, Q= 17.2 m^3/day; area=2250 m^2. Q/A*days=**44.3 cm**

3. Evapotranspiration

Methods:

There is a wide range of methods to measure or estimate evapotranspiration (ET), but most require expensive and technically challenging data collection and procedures. Estimates of ET from the study fens were determined using several methods ranging from simple and inexpensive to difficult and costly. Four methods were tested. (1) The METRIC model developed by the University of Idaho uses the visible, near-infrared, and thermal infrared energy spectrum bands from Landsat satellite images and weather data to calculate ET on a pixel by pixel basis. (2) Nearby Agrimet station use the 1982 Kimberly-Penman ET model to compute daily reference ET at each station. (3) The on-site micro-meteorological station at Johnson Fen uses the Penman-Montieth equations applied to energy balance equations to estimate ET. (4) Diurnal water table fluctuations (the White/Gerla method (Gerla 1992)) from the large nested wetland piezometers use the slopes of both the recession (declining) and accession (recovering) limbs of the daily hydrograph to estimate ET. All four methods are described in more detail in Appendices to the final report.

Results:

Evapotranspiration estimated using the four methods is summarized in Table 4.

Table 4. Compilation of estimated daily ET for selected dates using four methods. AFP=Air filled porosity, used for the White and Gerla methods.

Method	**ET estimates in 2010 (mm/d)**					
	Jul 2-4	**Jul 18**	**Aug 4-5**	**Aug 7**	**Aug 18**	**Sep 29**
METRIC – Dry Fen						1.75
METRIC – Johnson Fen						1.71
METRIC – Wilshire Fen						1.75
Christmas Valley AgriMet	3.56	6.60	6.10	5.59	5.08	3.05
On-site met station (Johnson)		5.58	5.09	4.42	4.32	3.26
White-Gerla – Dry Fen		7.85-8.25		5.56		
White-Gerla – Johnson Fen	0.42-2.23		7.17-7.23	7.44		3.48-4.68
White-Gerla – Wilshire Fen		5.62-6.88		6.22	7.56	

[3] The fen area estimates used here are smaller than the fen areas described in Section 2 because the the area here is only the area covered by the network of water table wells and piezometers.

Values for ET estimated using the Agrimet, on-site weather station, and White and Gerla methods were all within the same range (~3-7 mm/day), whereas the METRIC ET values were considerably lower (1.71-1.75 mm/day). For future EFL projects, using either nearby climate station estimates of ET, or an on-site meteorological station, are appropriate. The White/Gerla method is time-consuming and METRIC is costly. A mean value for all dates and methods is 5mm/day, which was multiplied by the number of days in the grazing season (58), to obtain a value of 29mm of ET for the water balance.

4. Water Usage

Methods:

Low flow water meters were installed on the water troughs at the three study fens to determine the amount of water supplied to the troughs during the grazing season. Each trough was equipped with a float valve that stopped the pump or gravity flow of water from the fen when the trough was full.

Results:

Maximum usage was 0.024 L/s (554 gal/day) at Dry, 0.012 L/s (277 gal/day) at Johnson, and 0.0066 L/s (150 gal/day) at Wilshire. Average extraction was 0.009 L/s=0.82 m^3/day (0.15 gal/min) at Johnson and Dry, and 0.003 L/s=0.22 m^3/day (0.04 gal/min) at Wilshire. To convert to water budget values, the average extraction rate for each fen was multiplied by the number of days in the grazing period, and then divided by the area of each fen covered by the network of water table wells. The latter value is smaller than the fen areas reported earlier, because the network of water table wells covered a smaller area than the total fen size.

- Johnson: 0.82 m^3/day * 58d / 2500 m^2 = 1.9 cm
- Wilshire: 0.22 m^3/day * 58d / 1750 m^2 = 0.72 cm
- Dry: 0.82 m^3/day * 58d / 2250 m^2 = 2.1 cm

5. Summary of Water Budget

The water budget for the 58-day grazing period is summarized in Table 5. Groundwater (both inflow and outflow), followed by ET, are the primary drivers of the water budgets in all three fens. By contrast, both precipitation and groundwater withdrawal for livestock are relatively minor in magnitude; the latter accounts for a minor amount of total outflow at all sites (5% for Johnson, 0.5% for Wilshire, and 7% for Dry).

Table 5. Wetland water budgets for a 58 day period (July 17-Sep 13, 2010)

Water Budget Component (cm)	Johnson	Wilshire	Dry
Precipitation	2.57	2.57	2.57
Surface inflow	0	0	0
Groundwater inflow	60.7	157.1	43.6
ET	-29	-29	-29
Surface outflow	0	0	0
GW outflow (residual)	-37.4	-135	-30
Water Usage	-1.9	-0.72	-2.1

Total inputs	63.3	159.7	46.1
Total outputs	-67.9	-164.8	-61
Change in storage[a]	-4.6	-5.1	-14.8

[a] average drop in water table over the 58-day period

iv. Modeling the effects of pumping

To evaluate the effects of pumping on the fens, we model groundwater flow using separate analytical solutions, and then a numerical model. These models allowed us to evaluate different withdrawal scenarios and to incorporate the various layers within the study system through which groundwater flows, including the peat, a semi-confining layer, and pumice. However, there were no data available to calibrate the models, which must be taken into account in interpreting the results.

1. Analytical model

Methods:

Withdrawing water or pumping a well causes a cone of depression, or drawdown, in the water table of an unconfined aquifer or in the piezometric surface for a confined aquifer. An analytical approach can be used to quantify the amount of drawdown from extraction points, which was done using the Theis equation (Theis 1935) for the confined pumice pumping, and the Neuman solution (Neuman 1972) for unconfined peat pumping. The equations require a number of simplifying assumptions including a homogeneous, horizontal aquifer of infinite extent, constant pumping rates and fully penetrating wells. Both equations require inputs of hydraulic properties (e.g. hydraulic conductivity and storativity or specific yield) to determine drawdown over distance and time.

To quantify the drawdown in the Antelope fens we used the hydraulic conductivity measurements determined by slug testing (see section 3.1.2) to calculate transmissivity and assumed values of storativity and specific yield from the literature. The analytical model approach is limited by the fact that each layer is evaluated independently, and therefore interactions among layers cannot be incorporated.

Results:

Drawdown was estimated for a pumping scenario of 75 days[4], which was the length of the grazing season at this location. The first scenario analyzed was extracting water from a shallow well screened in the 1m thick peat layer. The analytical analysis demonstrated that, using known and estimated aquifer parameters, pumping an adequate supply of water from the peat is not practicable. Even a small amount of pumpage (0.002 L/s, 0.03 gpm) causes the well to go dry.

On the other hand, pumping from the pumice could be sustained for at least 75 days at rates up to 0.115 L/s (1.8 gpm). Pumping from the pumice was evaluated by calculating drawdown curves at different

[4] This is longer than the time period used in the water budget because it includes the entire potential grazing season, rather than the actual grazing season which occurred in 2009.

pumping rates (0.09 L/s, 0.009 L/s). In general, field pumping rates averaged 0.009 L/s. The higher rate was evaluated for comparison. The cone of depression associated with pumping extends approximately 100m from the spring box, but the maximum drawdown is -5cm for the lower pumping rate and -50cm at the higher rate[5]. The analytical solution showed that pumping from the pumice could be sustained for at least 75 days at rates up to 0.115 L/s (1.8 gpm).

The analytical model presented two problems that could only adequately be resolved with a numerical model. First, the effect that pumping of the pumice layer would have on the peat layer above could not be deduced from this method. Over time, pumping from the pumice could cause gravity drainage out of the peat and water table decline in the fen. Second, one of the assumptions of the Theis analysis is an aquifer of infinite areal extent on the scale of the studied area. The fens are only about a third of a hectare in size making this result unreasonable. In reality, hydraulic boundaries and permeability variations within the aquifer will limit the extent to which the cone of depression can reach out, resulting in a smaller area of influence, less water available for extraction, and a potentially larger drawdown.

2. Numerical model

Methods:

To account for complexities in the hydrogeologic setting of the fens that violate several of the simplifying assumptions of the Theis or the Neuman solutions, numerical groundwater flow modeling was undertaken using MODFLOW (Harbaugh 2005).

The purpose of the MODFLOW model was to examine the potential effects that pumping in the pumice may have on water levels in the peat above it. The model developed for the fens is composed of three layers; a one-meter thick layer of peat on the top, a 0.1 meter confining layer below that, and a one-meter thick pumice layer at the base. All layers are horizontal and of effectively infinite extent. The hydraulic conductivity for the peat layer was held constant at 0.2 meters per day (m/d), the average value determined by slug testing. The hydraulic characteristics of the confining layer at the base of the peat are unknown, so a range of reasonable values were used for different model runs, specifically K values of 0.01, 0.001, and 0.0001 m/d. The pumice layer was modeled as having a K value of 0.82, 21, and 126 m/d, the range of K values measured by slug testing, for each value of the confining layer K values. Specific yield of the peat layer and storativity of the pumice layer were estimated from the literature as described for the analytical model. A number of simplifying assumptions were made for this model including treating the bedrock as a no flow boundary, and excluding evapotranspiration and recharge.

Results:

The numerical model (Figure 7) showed that pumping in the pumice over the 75-day season, at a rate of 0.009 L/s resulted in negligible drawdown in the peat layer above (3.1 cm at the pumping center and

[5]This document follows the convention from the ecological literature where a negative number indicates the height of the water table below the surface, and a positive number indicates the height of the water table belowground.

0.7cm 30m away) (Table 6). However, if the pumping rate is increased to 0.09 L/s, more significant drawdown occurs (32cm at the pumping center and 7.6cm 30m away). It should be noted that only a limited range of pumping is possible because of the thin layers overlying the bedrock. Overpumping would quickly dry the aquifer (peat and pumice layers) in the vicinity of the pumping well and no further pumping would be possible.

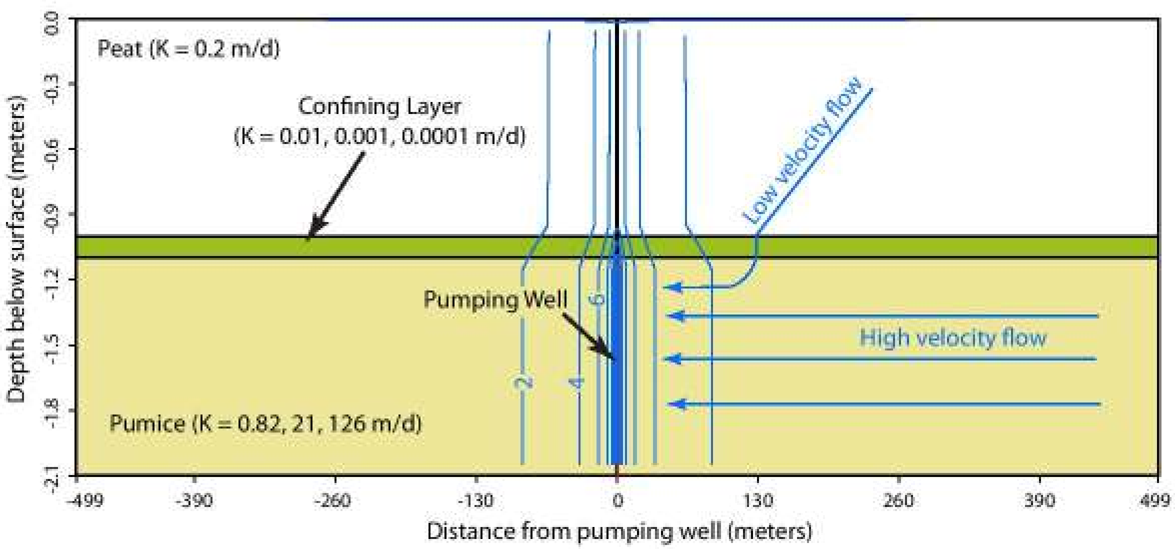

Figure 7. Cross section through the center of the three-layer MODFLOW simulation of the fen, showing the peat, pumice, and confining layers. The modeled hydraulic conductivities used for eah layer are shown (k). Pumping simulations for the three-layer model were all done from the pumice, or lower layer. The pumice and peat layers are each 1m thick, and the confining layer is 0.1m thick. The contour labels in blue are given as mm of drawdown after 75 days of pumping at a rate of 0.009L/s. Contour interval is 2 mm. The water table is shown as a blue line at the top of the figure. The results shown are for a k of 0.01m/d for the confining layer and 21 m/d for the pumice.

Table 6. Comparison of modeling results for drawdown in the pumice layer at the extraction point and at a radius of 30 m.

	Extraction Rate 0.009 L/s		**Extraction Rate 0.09 L/s**	
	Drawdown at spring box (cm)	**Drawdown at 30 m radius (cm)**	**Drawdown at spring box (cm)**	**Drawdown at 30 m radius (cm)**
Analytical	6.5	2.4	65	24
Modflow 3-layer	3.1	0.7	32	7.6

3. Model comparison

A comparison of predicted drawdown at the spring box and 30m away from the spring box by the two different approaches is presented in Table 6. Because of a lack of data for model calibration, and because the aquifer test was of short duration and a single low pumping rate, it is not possible to determine how well any of these models presents an accurate depiction of actual conditions. Both the Theis and the Neuman equations used in the analytical model treat the full thickness of the assessed portion of the aquifer system as homogeneous and isotropic and cannot take into account variations in permeability or multiple geologic layers. The three-layer numerical model partially addresses the limitations of the analytical approach by subdividing the aquifer into three layers with distinct properties, though still homogeneous and isotropic within each layer and effectively infinite in extent.

In this application, the analytical analyses predicted twice the amount of drawdown in the pumice layer when compared to results of the numerical model. This is mainly because the water in the Theis solution does not come from gravity drainage of the aquifer, but only from expansion of the water due to depressurization and collapse of the matrix. The Neuman solution also included gravity drainage of the aquifer, and therefore yields more water for a given decline in head. For this reason, a given pumping rate will show less drawdown in the Neuman solution than for the Theis solution. The advantage of the analytical analysis method using the Neuman or Theis equations is their simplicity which greatly minimizes time and cost. They are good tools for determining rough estimates of the magnitude of drawdown for various pumping rates and aquifer parameters. The MODFLOW model takes into account the hydraulic characteristics of the different hydrostratigraphic layers, and the effects of gravity drainage in the unconfined peat layer. Nevertheless, both models show a maximum drawdown less than 10cm for the current pumping rate, and at most 65cm for a pumping rate an order of magnitude higher. These results are compared to the ecohydrology relationships in section 5.

3 Quantify ecohydrology

The habitats of these fens and their species are dependent on groundwater because they are saturated to the surface year-round and groundwater and direct precipitation are the only sources of water to these wetlands. Two groundwater-ecology relationships were selected for analysis. First, plant species distributions have been shown consistently in the scientific literature to respond closely to the position of the water table. Second, the process of peat accretion is also dependent on a high and stable water table provided by groundwater discharge.

We assumed that these species and processes respond primarily to a local water table that is perennially at, or close to, the ground surface. The water table is maintained by a constant flux of groundwater from the volcanic source aquifer that underlies the area. We developed EFL for the position of the water table and did not consider other hydrologic attributes such as groundwater flux rates or water chemistry because the primary stress to these fens is water withdrawal and there is a robust literature describing the relationship between plant species' presence and the position of the water table, and very little describing other relationships such as flux rates.

i. Wetland plant distributions

Methods:

In the majority of cases where the management issue is groundwater abstraction, the key driver is the maximum depth to water table that the species in question can tolerate. Therefore, we used the hydro-ecological relationship of maximum depth to water table (hereafter maximum depth) for a suite of indicator species, to identify water table drawdown thresholds that will inform Environmental Flows and Levels. Water levels in plots without wells were estimated by extrapolating from surrounding plots with wells, assuming a smooth water level gradient following the slope of the land surface.

In July, 2010 we conducted a vegetation inventory in each of the plots described in association with the shallow water table wells (described in the section "Monitoring Wells"). Percent cover of all species, as well as litter and bare ground, were recorded for each plot. From the complete plant species data set, indicator species were identified. Appropriate indicators were defined as those that were widely distributed (i.e., occurring in at least three of the four sites); common (i.e., occurring in at least ten plots); and have federal wetland indicator status. Plants were classified by their wetland indicator status using the USDA Plants database (http://plants.usda.gov/java/). Species retained were those classified as either obligate or facultative wetland species (OBL, FACW). Bryophytes are not classified in this database; therefore, we relied on expert opinion (John Christy, Oregon Natural Heritage Information Center, personal communication). All recorded bryophytes ultimately were classified as OBL. The site-based data were supplemented with data from the published and grey literature of depth to water table for these same species in other wetlands.

Results:

Seventy-five species were identified in the plots, 11 mosses, 1 liverwort, and 63 vascular plants. Species-area curves were created for each fen to ensure sampling adequacy at each site. From the complete plant species data set, 17 indicator species were identified (based on the criteria of being widely distributed, common, and either obligate or facultative wetland species). Of the original 75 species, 33 were widely distributed. Of those, 22 were found in 10 or more plots. Of the 22, 17 were classified as either OBL or FACW (Table 7).

Table 7. Indicator species identified using the three criteria. Species codes and indicator status are from the USDA Plants database.

Species	**Species Code**	**Type**	**# Sites**	**Indicator Status**	**# Plots**
Aulacomnium palustre	AUPA70	bryophyte	4	OBL	34
Drepanocladus aduncus	DRAD2	bryophyte	3	OBL	23
Meesia triquetra	METR70	bryophyte	4	OBL	15
Philonotis fontana var. americana	PHFOA	bryophyte	4	OBL	21
Tomentypnum nitens	TONI79	bryophyte	4	OBL	36
Juncus balticus	JUBA	vascular	4	FACW	46
Packera pseudaurea	PAPS5	vascular	4	FACW	22
Sphenosciadium capitellatum	SPCA5	vascular	3	FACW	11

Muhlenbergia filiformis	MUFI2	vascular	4	FACW-	28
Dodecatheon jeffreyi	DOJE	vascular	3	FACW+	16
Mimulus primuloides	MIPR	vascular	4	FACW+	20
Saxifraga oregana	SAOR2	vascular	4	FACW+	27
Vaccinium uliginosum	VAUL	vascular	3	FACW+	36
Carex aquatilis var. aquatilis	CAAQA	vascular	4	OBL	73
Carex simulata	CASI2	vascular	3	OBL	19
Eleocharis quinqueflora	ELQU2	vascular	4	OBL	68
Hypericum anagalloides	HYAN2	vascular	4	OBL	23

From a search of the literature, 31 papers were identified with hydrology data on the identified indicator species. These data were combined with the site-based data and used to determine groundwater-ecology thresholds.

ii. Peat accretion

Methods:

In a semi-arid climate with a strong seasonal distribution of precipitation such as the one at this site, peat can only accumulate in situations where there is persistent groundwater discharge.

Models of peat development explicitly incorporate peat accretion in relation to hydrologic factors (Childs and Youngs 1961; Ingram 1982; Clymo 1984; Belyea and Baird 2006; Frolking *et al*. 2010). Central to these models is the assumption that a peatland has two main strata: an upper oxic acrotelm where the majority of biological activity takes place and a lower, anoxic catotelm where lower rates of biological activity make this the zone of peat storage (Clymo 1984). The boundary between acrotelm and catotelm is functionally defined as the maximum depth to the water table (Childs and Youngs 1961; Ingram 1982; Belyea and Baird 2006). Based on this, we assume that lowering the water table below the summer maximum depth will lead to changes in biological activity in the acrotelm that are significant enough to increase decomposition in the catotelm (Belyea and Clymo 2001).

Our initial intent in measuring peat depths associated with the position of the water table was to collect these data for every vegetation plot. However, finding a reliable non-destructive method was challenging. Here we describe the two approaches that yielded useable data. First, we quantified the relationships between the current position of the water table and the peat deposits, for nested wells where we had collected both water level and soil profile data (described previously in "Monitoring Wells"). These data were useable but the data set was relatively small compared to spatial variation among boreholes. Second, we used the summer maximum depth to water table as a surrogate for the boundary between the acrotelm and catotelm, assuming that the maintenance of the water table above this level is a threshold for sustaining the process of peat accumulation. Moreover, because the water balance for these sites during the growing season is dominated by groundwater discharge, we assumed the water table over the timescale of decades is relatively stable and therefore this relationship can be measured accurately with three seasons of field data. These two approaches are not independent because they use overlapping data sets.

Results:

Peat has developed where the maximum depth to water table is less than -40 cm throughout the growing season (Figure 3). For the upland piezometers, where there was no peat, depth to water ranged from -80 to -160cm. Peat depths showed statistically significant, positive relationships with the mean depths to water table (Figure 8a), as well as the minimum and maximum (data not shown), indicating that more peat accumulates where the water table is higher. Peat depth also showed a negative relationship with the range in depths to water table, with more stable water tables having deeper peat deposits (Figure 8b).

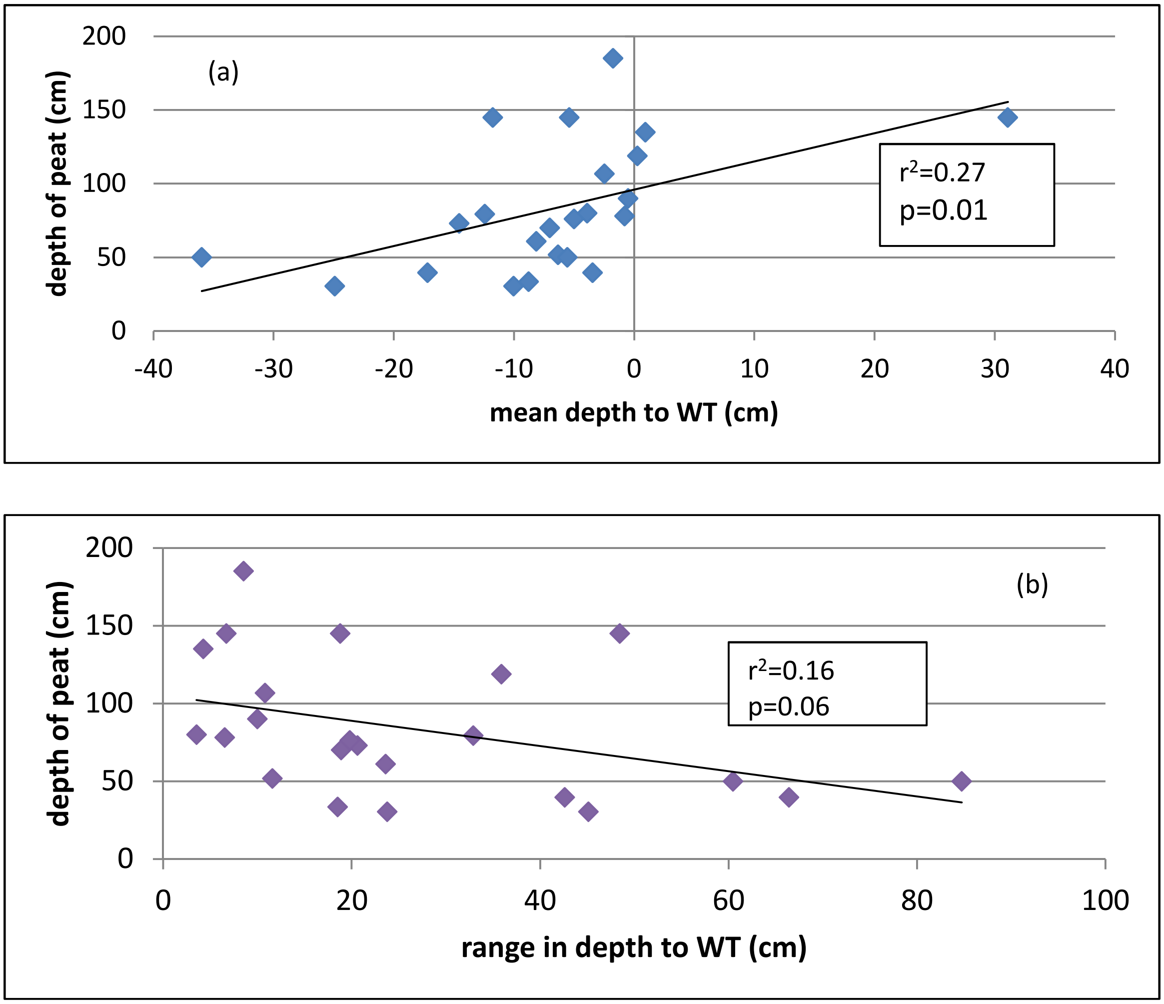

Figure 8. Relationships between peat depth and the (a) mean and (b) range in depths to the water table. N=23 for all regressions.

4 Develop thresholds

Once the groundwater-ecology relationships have been measured, we seek to answer the question, *what are the safe limits to changes in the important attributes of groundwater?* Specifically, what is the size and shape of the drawdown cone produced by withdrawal and how much drawdown of the water table can the plants and ecosystem processes tolerate?

i. Vegetation-hydrology relationships

Methods:

This step requires determining the groundwater drawdown thresholds beyond which irreversible habitat loss will occur for each groundwater-ecology relationship. The vegetation data were organized by presence/absence and by abundance (percent cover), to determine which parameter(s) demonstrated a stronger relationship to the position of the water table.

Indicator species were assigned water depth metrics according to the plots in which they occurred. For each of the indicator species, its "mean depth metric" value was an average of the mean depth to water table for the plots in which it occurred. Its "min depth metric" and "max depth metric" values were the highest of the minimum values (most shallow) and lowest of the maximum (deepest) values for the plots in which it occurred, respectively. The result is a mean water table depth and range for each species that reflects the full range of 'micro-hydrologies' where it grows in these sites.

To identify thresholds of depth to water table, we adapted an approach currently under development in the European Union in the implementation of the Water Framework Directive (Schutten *et al*. 2011; UK TAG 2012). We define the threshold value as the 75th percentile of the maximum depth to the water table, implying a management goal of maintaining the water table above that value. The European approach involves setting thresholds using logistic regression of poor quality vs. high quality sites; however, since we sampled only high quality sites we were not able to use the complete approach. This method takes a precautionary approach by setting the threshold higher than the maximum water table depth at which the species are found.

Results:

The majority of indicator plants are found in plots where the max depth to water table is approximately -20cm. However, there is a skewed (non-normal) distribution toward lower water tables, as well as a small number of outlier occurrences, which indicate that some of these species can tolerate water tables up to -70cm below the land surface (e.g., *Carex aquatilis*).

Literature values showed a greater range in lower water tables than the field data indicated. This is because some of these species also are found in seasonal meadows and other wetlands with a greater hydroperiod range, including *Carex aquatilis*, *Philonotis fontana* var *americana*, *Sphenosciadium capitellatum*, and *Vaccinium uliginosum*. These species were eliminated from the final list of indicator species because they are not adequately sensitive to changes in the position of the water table. None of the data from the literature showed species having greater tolerance to flooding (higher water tables) than the field data. The final range in 75th percentile maximum depth to water table (excluding the three species listed above) was -0.9cm to -34.8cm.

Cumulative species loss at increasing maximum depth to water table is shown in Figure 9. This assumes that if a particular species was not found at those lower water tables, it cannot tolerate those conditions. From our data, species losses began at maximum depths greater than -20 cm, and all indicator species are lost at depths greater than -70 cm. That threshold was somewhat lower (-115 cm) once data from the published literature were incorporated. We do not have an approach for developing time-based thresholds; in other words, determining how long the water table must be drawn down for indicator species to be lost. It is probably possible to draw the water table down for brief periods without losing the indicator plants.

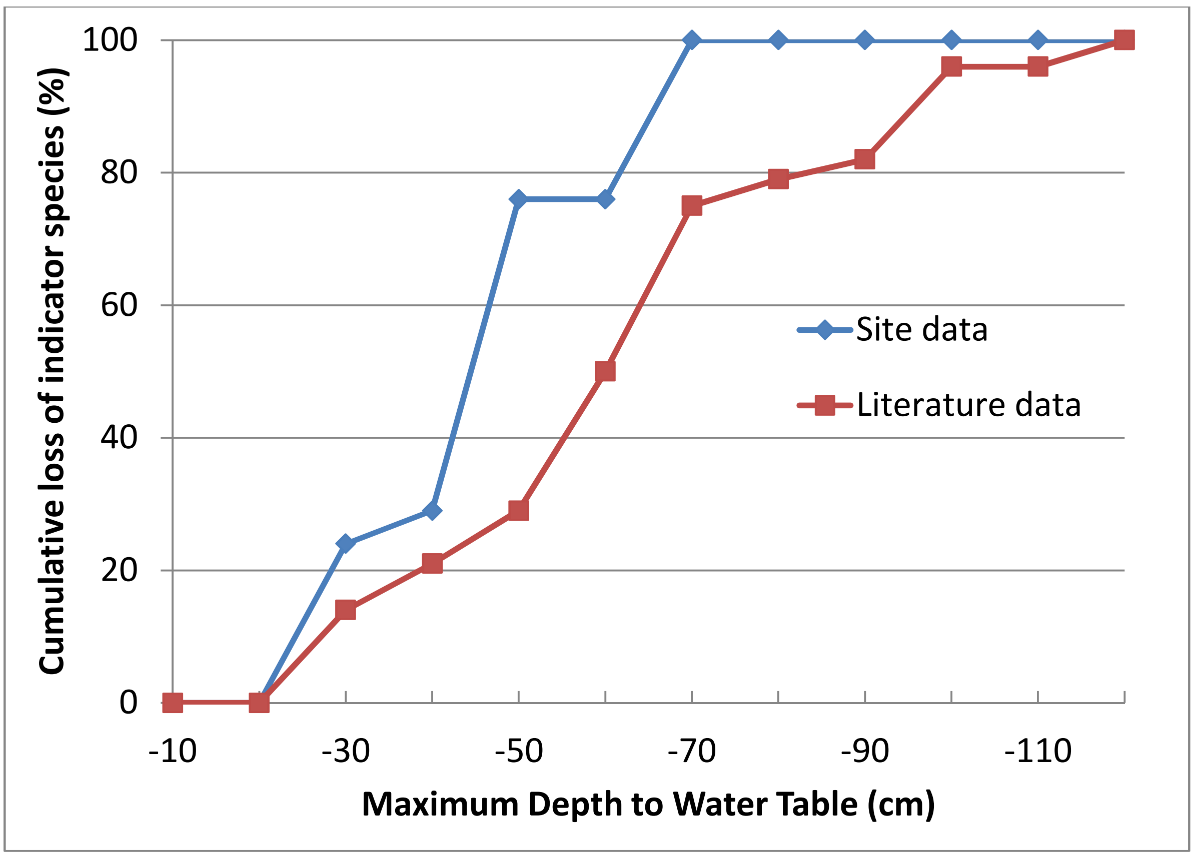

Figure 9. Cumulative percent of indicator species lost with maximum depth to water table, at decreasing 10cm increments, from 0cm to -120cm.

The relationships between the hydrology metrics and percent cover of indicator species were not as strong as for presence/absence data. This is not unexpected. The presence (or absence) of a particular species is probably largely due to local conditions. However, that species' ability to increase in abundance may be more closely related to modes of reproduction, competition, water or soil chemistry, or some other factors. Therefore we retained only the original analysis that considered only species presence or absence within a plot.

ii. Peat-hydrology relationships

Methods:

Peat-hydrology thresholds were developed in two ways. First, for each of the larger water table wells, we calculated the mean, minimum (shallowest), and maximum (deepest) depth to the water table for the three-year period of data. We performed regression analysis of peat depth on mean, minimum, maximum, and range in depth to water table after log transforming all variables to meet regression assumptions (see Figures 8a and b from section 4.2). Second, we estimated the boundary between acrotelm and catotelm using the summer maximum depth to the water table. The years for which we have field data (2009-2011) represent conditions similar to the last 50 years recorded at two nearby weather. Therefore, we combined the water table datasets from the small and large water table wells for all sites and all years to identify the summer maximum depth, which forms the basis for a threshold for sustaining peat accumulation processes. Similar to the approach for the vegetation data, we calculated the 75th percentile of the maximum depth for all piezometers with peat, with the goal of maintaining the water table above that value.

Results:

Annual maximum depth to water table data for the three fens as a surrogate for a peat accretion or maintenance threshold. Figure 10 shows the cumulative area with peat loss, indicating that the fen dries out as the water table drops, as represented by the percent of piezometers with maximum depth to water table at increasing drawdown. These data show the summer maximum water table ranges from -5 to -85 cm, but the majority fall within the range of -10 to -40 cm. The 75th percentile values for maximum depth to water were -16.6 cm (Johnson fen), -22.0 cm (Wilshire fen), -32.2 cm (Dry fen), and -29.6 cm (Round fen).

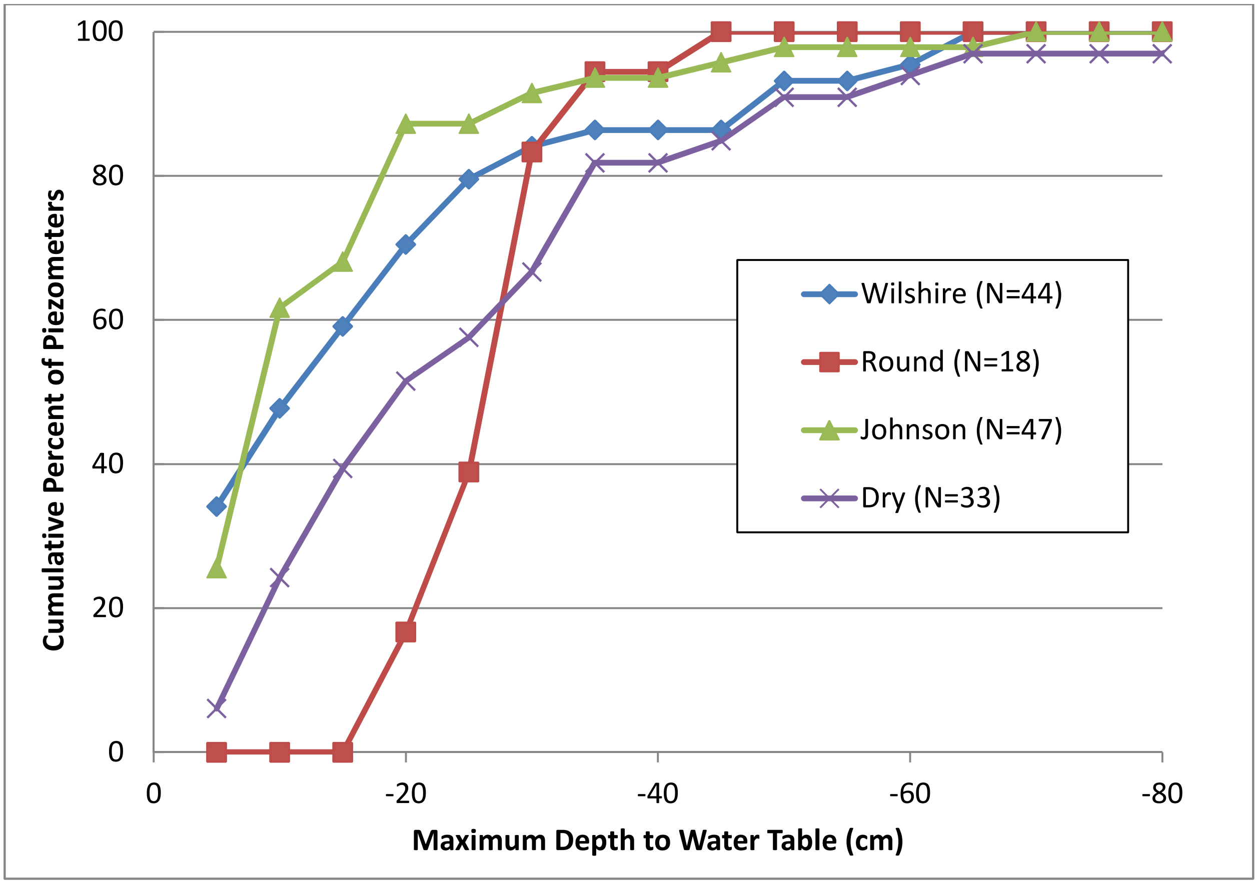

Figure 10. Cumulative percent of piezometers where the water table drops below a maximum depth to water table, at decreasing 5cm increments, from 0cm to -85cm.

The somewhat weak relationships between peat thicknesses and depth to water table metrics (Figure 8) may have been due to the low sample size (N=23) or the skewed distribution of wells with mean water tables near the surface. Another confounding factor was a relatively high level of spatial variation in peat depths (Figure 3; coefficient of variation within each site ranged from 36-66%). This could be due to the undulating nature of the pumice surface upon which the peat accumulates; thinner peat on high points and thicker peat in depressions where the water pools. In the future it may prove to be fruitful to examine the peat-water table relationship in fens with a larger sample size and/or more detailed investigative approach.

5 Evaluate groundwater management in relation to thresholds

Results from the aquifer test, water balance, and analytical and numerical models were compared to the water table thresholds identified for indicator species and peat accretion to develop the Environmental Flows and Levels recommendations. From the comparisons it was possible to derive a maximum pumping threshold that should be protective of the biotic community in the fens. We then compared this to pumping rates from 2009 and 2010 and to future proposed pumping rates. EFL recommendations were developed based on these comparisons and management objectives. Figure 11 illustrates the thresholds associated with groundwater withdrawal found in this study.

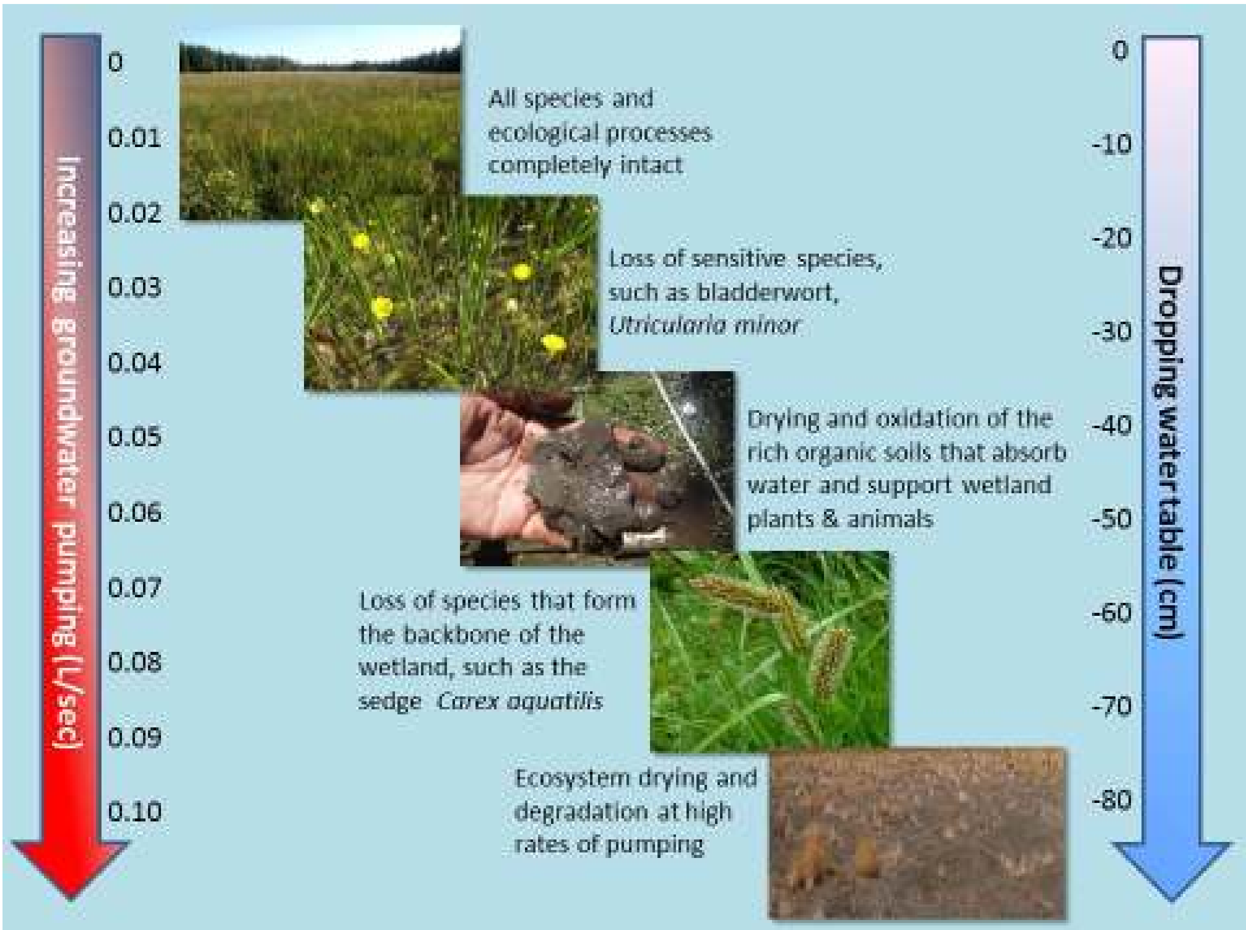

Figure 11. Likely Ecosystem changes associated with increasing groundwater pumping.

The most tolerant plant species appear to be able to withstand maximum water table depths of -70cm to -100cm, but species losses are expected to begin at maximum depths of -20cm. The peat data indicate that deposits are only present where the maximum depth to water table is shallower than -40cm but most of the piezometers in peat deposits have water tables that remain above -20cm. This corresponds to the summer maximum water table depth as a reasonable proxy for the acrotelm-catotelm boundary. Using the 75th percentile approach, thresholds are -0.9cm to -34.8cm for fen indicator plants, depending on the species, with bryophytes being somewhat more responsive than vascular plants. For the process of peat accretion, thresholds were -16.6cm to -32.2cm, depending on the site. This range in ecological thresholds is similar to what has been reported for peatlands in many parts of the world (Verhoeven *et al*. 1993; Weixelman and Cooper 2009; Verry *et al*. 2011; Sabiham *et al.* 2012).

The weight of evidence from the hydrogeologic analyses indicates the proposed withdrawal scenarios most likely won't exceed a low water table ecological threshold value of -32cm. The current withdrawal rate varies from 0-0.024 L/s (0-554 gal/day), with a daily average of 0.009L/s at Johnson and Dry and 0.003L/s at Wilshire. The short duration (32 hours) aquifer test did not result in any drawdown in adjacent wells. The analytical model of pumping from the pumice layer at 0.09L/s indicated that the threshold would be exceeded within a 10m radius of the spring box (Figure 7 and Table 6), but this was a higher pumping rate over an extended period of time compared to the current situation. In contrast, the cone of depression predicted for pumping at 0.009L/s (comparable to 2010 rates) did not exceed the ecological thresholds, yielding only a -5cm drawdown at the spring box.

The final step is to develop a monitoring plan that tests the assumption that seasonal pumping won't measurably lower the growing season water table in the fens. That can be done in a subset of fens with simple shallow monitoring wells and periodic vegetation monitoring for sensitive species such as those listed in table 7.

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