

High resolution modeling of agricultural nitrogen to identify private wells susceptible to nitrate contamination

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ABSTRACT

Given the lack of data on private wells, public health and water quality specialists must explore alternative datasets for understanding associated exposures and health risks. Characterizing agricultural nitrogen inputs would be valuable for identifying areas where well water safety may be compromised. This study incorporated existing methods for estimating nutrient loading at the county level with datasets derived from a state permitting program for confined animal feeding operations and agricultural enterprise budget worksheets to produce a high resolution agricultural nitrogen raster map. This map was combined with data on soil leachability and new well locations. An algorithm was developed to calculate nitrogen loading and leachability within 1,000 meters of each well. Wells with a nonzero nitrogen total linked to soils with high leachability were categorized and displayed on maps communicating well susceptibility across the state of Oregon. Results suggest that 4% of recently drilled wells may be susceptible to nitrate contamination, while areas identified for mitigation are too restrictive to include all susceptible wells. Predicted increases in population density and the steady addition of approximately 3,800 new wells annually may lead to a large number of residents, especially those in rural areas, experiencing long-term exposures to nitrate in drinking water.

Key words | drinking water, fertilizer, high resolution modeling, manure, nitrate, private wells

INTRODUCTION

Nitrate is a very common groundwater contaminant (Burkart & Stoner 2007). While nitrate can occur naturally, concentrations above 1 milligram per liter (mg/L) are likely due to anthropogenic inputs, particularly agricultural activities (Warner & Arnold 2010; Dubrovsky *et al.* 2010). Nitrogen from land-applied fertilizer and manure drives accumulation of nitrate in soil through organic matter mineralization and fixation processes (He *et al.* 2011). Nitrate not taken up by plants or microorganisms can leach through permeable soil into groundwater, facilitated by rainfall and irrigation (Burkart & Stoner 2007). A report by the US Geological Survey (USGS) shows that nitrate concentrations exceeded background levels in 64% of aquifers studied within the USA (Dubrovsky *et al.* 2010). Median concentrations were highest in aquifers beneath agricultural areas.

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Nitrate contamination of groundwater diminishes supplies of safe drinking water, is costly to remediate, and poses a public health threat, especially for individuals relying on private residential wells (Levin *et al.* 2002). A 2009 USGS report on private wells states that over 4% of wells located within 30 sampled aquifers used for drinking water had nitrate above 10 mg/L, the maximum contaminant level (MCL) allowed for public water administered under the Safe Drinking Water Act (SDWA) (DeSimone *et al.* 2009). Unlike public supply wells, the safety of water from private wells is not ensured by SDWA regulations. Rather, owners are responsible for maintaining the integrity of their well, testing water regularly, and taking actions necessary to prevent exposures to contaminants (Backer & Tosta 2011). However, numerous studies

on stewardship behaviors reveal that many owners are not maintaining wells or testing the water according to health-protective guidelines (Jones *et al.* 2006; Hexemer *et al.* 2008; Kreutzwiser *et al.* 2011). This stewardship gap is a concern given that nitrate levels are increasing. A USGS study demonstrated that from 1993 to 2003 the proportion of US private wells with nitrate greater than the MCL increased from 16 to 21% (Dubrovsky *et al.* 2010). As cultivated land transitions to residential use (FIC 2006), population-level exposures to nitrate in small water systems are likely to increase given that contamination can persist for decades or longer (Showers *et al.* 2008). While the MCL for nitrate was established mainly to address infant methemoglobinemia, studies also suggest that nitrate in drinking water is associated with adverse effects on thyroid function, respiratory tract infections, and reproductive and developmental outcomes in infants (Gupta *et al.* 2000; Manassaram *et al.* 2006; Tajtáková *et al.* 2006; Gatseva & Argirova 2008).

In Oregon, over 90% of rural residents depend on groundwater. In many areas, it is the only source of potable water (DEQ 2011). An exact count of active wells is unavailable; however, current estimates are that Oregon has over 350,000 private wells (DEQ 2011), suggesting that 23% of the population are relying on private wells for water (USCB 2010). Furthermore, approximately 3,800 exempt-use wells are drilled annually (2008), comprised mainly of small group or single domestic wells not subject to Oregon's water rights permitting process.

Nitrate contamination of Oregon groundwater has been demonstrated by the USGS (DeSimone *et al.* 2009) and State agencies (DEQ 2011; Hoppe *et al.* 2011). Oregon has three Groundwater Management Areas (GWMA), comprising approximately 3,000 square kilometers (Figure 1), which were designated due to widespread, nonpoint source nitrate contamination of groundwater (DEQ 2011). While levels in GWMA aquifers consistently exceed the MCL, sampling of wells or aquifers in areas outside of GWMA has also shown levels of nitrate above this health-based threshold (Feaga *et al.* 2004; DEQ 2011; Hoppe *et al.* 2011). Aside from Oregon's Domestic Well Testing Act, which requires testing when properties with a well change ownership (Hoppe *et al.* 2011), currently no state legislation or state-level program exists to ensure well water safety equivalent

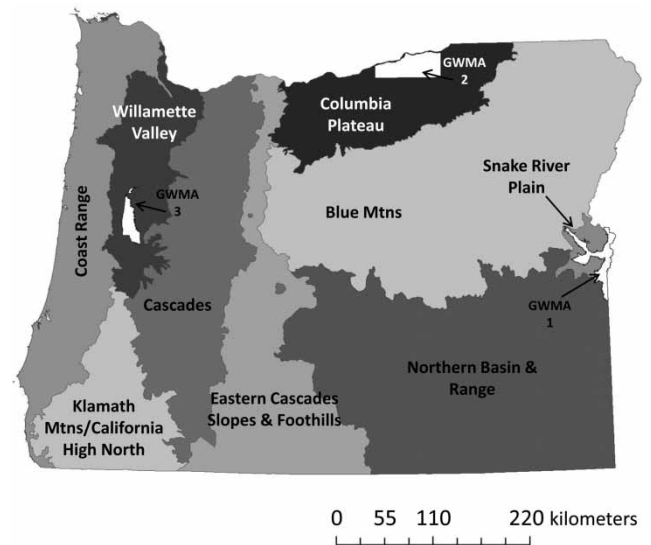


Figure 1 | Study area with nine ecoregions and three GWMA. 1 = Northern Malheur County, 2 = Lower Umatilla Basin, 3 = Southern Willamette Valley GWMA.

to the SDWA. In addition, a survey by Mitchell & Harding (1996) demonstrated that many Oregonians were not testing their well water as recommended; however, more current studies on well stewardship behaviors are needed.

Public health and water quality specialists in Oregon and across the USA need an understanding of groundwater conditions and contamination threats in order to protect source water and human health, especially for private well owners or residents moving into areas not serviced by public utilities (Rowe *et al.* 2007; Masetti *et al.* 2008). Characterizing the influence of land-applied manure and fertilizer, generally recognized as major contributors to groundwater nitrate (Fields 2004; Galloway *et al.* 2008), would be especially useful for identifying where private well water safety may be at risk.

Numerous studies investigating nutrient loading to groundwater include some measure of fertilizer or manure inputs, many relying on geographic information systems (GIS) for processing and distributing data spatially and temporally (Swartz *et al.* 2003; Greene *et al.* 2005; Nolan & Hitt 2006). However, these measures are often based on coarsely aggregated data applied to a limited study area. Given that fertilizer and manure applications can vary substantially over short distances, depending on location and type of farm and crops grown (Luo & Zhang 2009), low resolution datasets may not adequately capture the spatial fluctuations

in nutrient loading necessary for predicting groundwater contamination at well point locations (Slaton *et al.* 2004; Elliot & Savitz 2008). Furthermore, it is a common challenge to find detailed spatial information with full statewide coverage because of effort, expense, and often legislation required to collect data for such a large area.

Ruddy *et al.* (2006) describe methods for allocating data on state fertilizer sales and estimating manure generation from livestock in order to characterize nitrogen inputs at county-level. These methods were adapted in the current study and combined with data unique to Oregon, including crop-specific fertilizer expenditures, information on livestock populations, and a spatial index of soil sensitivity to nitrate leaching (SSNL). These datasets were combined in a GIS and compared with new well construction across the state in order to identify wells susceptible to nitrate contamination. The objectives of this study were to:

1. advance methods for estimating and distributing data on nitrogen from fertilizer and manure applications by incorporating state-level data to improve spatial resolution of nitrogen loadings; and
2. integrate these data with information on new well construction to identify areas of the state where new wells may be affected by nitrate contamination.

MATERIALS AND METHODS

Study area

The study area included all of Oregon, which is divided into nine ecoregions (Figure 1) (Thorson *et al.* 2003). Oregon's agricultural economy is closely tied to its climate. Counties comprising the Willamette Valley and northern Cascades account for nearly 50% of the state's gross farm and ranch sales, followed by counties of the Columbia Plateau (NASS 2013).

The population of Oregon is approximately 3,831,000 and mainly concentrated in the Willamette Valley. While 81% of the population is urban dwelling, only 1.2% of the land is designated urban; there are less than 40 persons per square mile in Oregon compared to the national average of 87 persons (USCB 2010).

Model development

Concentrations of nitrogen (N) from land-applied manure (manure-N) and fertilizer (fertilizer-N) were derived with methods by Ruddy *et al.* (2006) for estimating manure-N and fertilizer-N at the county-level. We adjusted these methods for the temporal extent of our project (2000–2007) and specific farming practices within Oregon. We then incorporated information on livestock populations and crop-specific fertilizer expenditures, explained in detail below, to increase spatial resolution of distributed manure-N and fertilizer-N data (Table 1).

All geospatial modeling was conducted using ArcGIS (Version 10.0). We distributed estimates of manure-N and fertilizer-N across the state based on land use and crop type, using the Oregon Cropland Data Layer (CDL) (USDA NASS 2007). The CDL program utilizes seasonal satellite imagery to locate and identify field crops, adjusting results with a regression estimator, farmer-reported data, Farm Service Agency (FSA)/Common Land Unit (CLU) data, agribusiness data, and data from the USDA Census of Agriculture (Mueller & Ozga 2002). The strength and emphasis of the CDL is crop-specific categories. For other land use categories, the 2007 CDL samples the 2001 National Land Cover Database wherever FSA/CLU data are unavailable. Spatial resolution for the 2007 CDL raster is 56 by 56 meters (USDA NASS 2007). Finally, we incorporated datasets representing SSNL and new private well construction to identify wells susceptible to nitrate contamination.

Estimating and distributing manure-N

Manure-N estimates were based on livestock type and population data collected by the 2007 Census of Agriculture. All major animal categories included in work by Ruddy *et al.* (2006) were included in our analysis, with the addition of goats and mink. Census regulations require that data not be published that would disclose operations of an individual farm (USDA 2009). For counties with a livestock population reported as 'nondisclosed' we derived a livestock population for that county by summing the number of animals reported for all other counties and subtracting that value from total number of animals reported for Oregon as a whole. This residual value was allocated to the 'nondisclosed' county. If

Table 1 | Datasets used for analyses

Layer	Source	Data	Time	Extent ^a	Original units
Land use (base layer)	National Agriculture Statistics Service	Crop-specific data, other land-use categories	2007	State	56 meter pixels
Nitrogenous fertilizer	Association of American Plant Food Control Officials	Fertilizer sales	2007	State	County
	Census of Agriculture OSU Extension	Fertilizer expenditures Crop-specific fertilizer recommendations	2007 1988–2010	State Various	County Acre
Nitrogen from manure	Oregon Department of Agriculture	CAFO locations, livestock type, number	2000–2007	State	Point locations
	Census of Agriculture	Livestock type, number	2007	State	County
Soil characteristics	Oregon Department of Environmental Quality	SSNL categories	2007	State	Polygon
New well construction	Oregon Water Resources Department	Number, location new wells	2000–2007	State	Point locations

^aScale resolution for all layers: 56 × 56.

more than one county had a ‘nondisclosed’ quantity for a livestock category, the residual was allocated using a weighted percentage based on number of farms in each ‘nondisclosed’ county that reported producing this livestock. For example, there were 30 residual animals in the ‘Hogs and pigs’ category allocated to two counties: Gilliam had two farms (40%) that reported raising hogs and pigs, while Sherman had three farms (60%). Using this information as weights, Gilliam and Sherman were allocated 12 and 18 animals, respectively.

Using methods and assumptions of Ruddy *et al.* (2006) and Goolsby *et al.* (1999), we estimated nitrogen content of manure produced by various livestock populations. Goolsby *et al.* (1999) did not provide estimates for goat or mink, so nitrogen content for manure from these animals was obtained from NJAES (2006) and Wright *et al.* (1998).

Livestock populations are reported to the Census as year-end inventory numbers. For all livestock groups, this number was assumed to represent the population throughout the year; we did not adjust manure nitrogen estimates for livestock that may be slaughtered before year’s end (i.e., heifers, hens). The final equation for calculating annual estimates of total nitrogen per livestock group (LSTOCK-N) per county was:

$$\text{LSTOCK-N}_{ij} = \text{LSTOCK}_{ij} \times N_i \times \text{LCYCLE} \quad (1)$$

where LSTOCK-N_{ij} is estimated nitrogen from manure from i th livestock group in j th Oregon county (kg), LSTOCK_{ij}

represents number of animals reported in i th livestock group in j th Oregon county, N_i denotes N content of manure for i th livestock group (kg/day/animal), and LCYCLE is assumed annual lifespan (365 days).

Estimates for manure-N per county were derived by summing all LSTOCK-N for the county. To distribute county-level manure-N estimates to a finer spatial resolution, we used information submitted to the Oregon Department of Agriculture (ODA) under State statute (ORS468B 2009) that requires a person who operates a Confined Animal Feeding Operation (CAFO) to obtain a permit from the ODA. The definition of a CAFO under Oregon statute is broader than the federal definition, such that many farms that would not be classified as a CAFO under federal law do meet state criteria. The dataset of CAFO permits obtained from ODA included location, livestock type, and number of animals for any farm permitted between 2000 and 2007. These years were selected to be relevant to the temporal extent of other datasets used in this analysis. If a CAFO lacked information on number of animals, a value was assigned based on average number of animals within the animal category for all CAFOs in the dataset. CAFOs permitted for the category ‘Other’ were removed from the dataset since most of these facilities were boarders or wildlife parks, and waste from these is unlikely to be widely distributed. CAFOs permitted as wholesalers were also eliminated as these are generally auction houses and

would not maintain animals for an entire year, a key assumption of our approach.

Following methods by Ruddy *et al.* (2006) to estimate nitrogen release per animal group, we calculated the annual nitrogen attributable to each CAFO based on the animal category for which it was permitted. Data on CAFO locations were combined with the CDL, and nitrogen totals for each CAFO were distributed to cells identified as cropland, pasture, or herbaceous grassland within a 16 kilometer (km) buffer around the CAFO (Figure 2). This buffer distance was selected based on information from Bradford *et al.* (2008) that manure from a CAFO is usually applied within 16 km of the facility. Herbaceous grassland was included as a manure ‘receiving’ category as it may be used for grazing (USGS 2010). Nitrogen totals for each CAFO were distributed within each buffer to receiving cells using a weighted-distance technique, given that manure application is likely heaviest on lands proximal to the farm and attenuates with distance to the buffer border.

The weighted-distance technique involved creating centroids from grid cells of receiving areas. Nitrogen was distributed from the CAFO to each centroid. Nitrogen quantities were inversely weighted by distance of the cell centroid from the CAFO (Equation (2)). Nitrogen quantities at centroids were aggregated for cells where the centroid was in the buffer of more than one CAFO, according to the following equation:

$$X = [N(1 - D_x/B)] / [\sum_{i=1,n} (1 - D_i/B)], \quad (2)$$

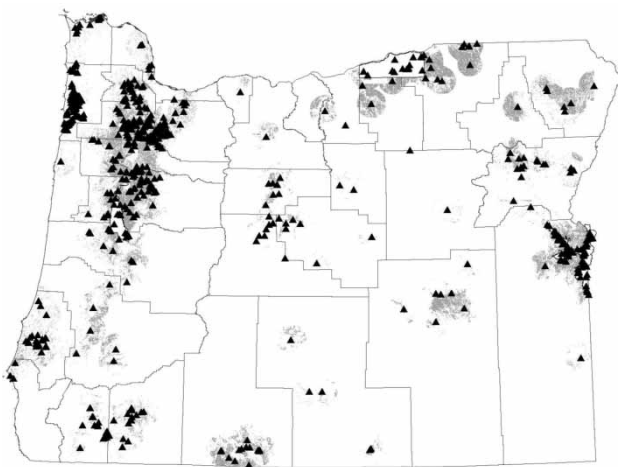


Figure 2 | State-permitted CAFOs, 2000–2007, and manure-N distribution.

where X is nitrogen received from one CAFO in cell x (kg), x denotes the specific receiving cell, i is any given receiving cell, n refers to number of receiving cells in buffer area, N is estimated quantity of nitrogen distributed from the CAFO, D_x is distance of cell x (centroid) from the CAFO (centroid) in meters, D_i is distance of point i from CAFO in meters, and B represents the 16 kilometer buffer distance around the CAFO.

We then summed all nitrogen distributed from individual CAFOs to receiving cells within individual counties (CAFO-N). These county CAFO-N totals were subtracted from the LSTOCK-N county estimates derived from Equation (1). For counties where the residual was positive, the residual value was uniformly distributed to all receiving cells located within all CAFO buffers for the relevant county. For counties where the residual was negative, it was assumed that LSTOCK-N estimates had been completely allocated using CAFO permit information, and we made no further adjustments.

We did not include loss via volatilization in our determination of LSTOCK-N due to the wide variability of nitrogen volatilization rates. Manure nitrogen loss to the atmosphere occurs mainly through volatilization of ammonia. Atmospheric ammonia stemming from agriculture can significantly impact the environment and has been implicated in widespread ecosystem damage (Loubet *et al.* 2009). However, ammonia volatilization losses vary greatly depending on environmental conditions, manure type, storage, handling, and application methods (Meisinger & Jokela 2000; Ruddy *et al.* 2006). Losses range from nearly 100% for surface application with optimal conditions for volatilization, to a few percent when manure is incorporated immediately into soils (Meisinger & Jokela 2000). We did not have access to farm-specific details regarding process and application methods of manure, so we could not estimate nitrogen loss through ammonia volatilization for individual farms. We recognize that this may impact the LSTOCK-N estimate for some facilities.

Estimating and distributing fertilizer-N

Fertilizer-N estimates were based on reports of fertilizer expenditures from the 2007 Census of Agriculture. First, 2007 fertilizer sales information for Oregon was obtained

from the Association of American Plant Food Control Officials (AAPFCO) and converted from tons of product to kilograms nitrogen based on chemical composition for each product containing nitrogen. AAPFCO lists most fertilizer products with their chemical composition, e.g., ‘82-0-0 Anhydrous Ammonia’ is 82% nitrogen by weight. When a product was not listed with its chemical composition, percentage of nitrogen in the product was assigned based on review of the literature. Following the approach by Ruddy *et al.* (2006), nitrogen totals for each fertilizer product were summed to give a fertilizer statewide sales total (FFS) for 2007, which was then used in Equation (3) to estimate county-level nitrogen inputs from fertilizer (FERT-N) as proportional to fertilizer expenditures in each county:

$$\text{FERT-}N_i = \text{FFS} \times (\text{FCE}_i / \text{FSE}) \quad (3)$$

where FERT- N_i is estimated nitrogen from fertilizer use in county i (kg N), FFS denotes total fertilizer sales in Oregon (kg N), FCE $_i$ is fertilizer expenditure for county i (\$), and FSE is total fertilizer expenditures for Oregon (\$).

To distribute county-level FERT-N to crop fields represented in the OR-CDL, thus improving spatial resolution, we used crop-specific fertilizer expenditure estimates derived from Enterprise Budget Sheets (EBS) published by Oregon State University (OSU) Extension. An agricultural EBS is a detailed account of revenues and expenses related to farming a specific crop. EBS from OSU-Extension provide recommendations for amount and type of fertilizer to purchase for growing specific crops in the state (OSU 2009). While some EBS quantified the amount of nitrogen for a crop annually, others only provided a general cost estimate. Furthermore, some crops had more than one EBS available depending on the year of establishing or maintaining the crop. To derive a crop-specific dataset for fertilizer nitrogen (EBS-N), we used the most recent EBS available for a crop and averaged recommended nitrogen use across establishment and maintenance years when relevant. For crops with EBS that only provided cost estimates, the recommended quantity of nitrogen for the crop was requested directly from an OSU-Extension agent or obtained from EBS provided by Washington State Extension. If the recommended nitrogen was given as a range, the average between lowest and highest values was used in the EBS-N dataset.

Substitutions were used for crops represented in the OR-CDL for which no EBS was available (Table 2).

Crop-specific EBS-N data (converted from lb/acre-year to kg/cell-year) were distributed to relevant cells according to crop type as represented in the OR-CDL, and a sum was derived for total nitrogen from fertilizer predicted by EBS-N data for each county. These county EBS-N totals were then subtracted from the county FERT-N totals derived from Equation (3). Any residual was uniformly distributed across all cropland within the county. For all counties but three (Gilliam, Morrow, and Sherman), residuals were positive, suggesting that more fertilizer was purchased and used in most counties than amounts recommended in OSU EBS. For the three counties with negative residuals, we assumed that FERT-N estimates had underrepresented nitrogen from fertilizer that is likely being applied, given specific crops identified by the OR-CDL and nitrogen use recommended by OSU EBS, and no further adjustments were made.

Soil sensitivity to nitrate leaching

Soil sensitivity refers to a soil’s tendency to allow a chemical to move through to groundwater (Huddleston 1998). DEQ developed a geospatial dataset to predict SSNL for large areas of the state (Seeds 2011) using methods of the Oregon Water Quality Decision Aid (OWQDA; Huddleston 1998) and data from Natural Resources Conservation Service’s Soil Survey Geographic Database.

The OWQDA was developed as a screening tool for determining the likelihood that a specific chemical, including

Table 2 | Substitutions for OR-CDL crops when EBS was unavailable

OR-CDL crop missing EBS	Substituted fertilizer-N value
Rapeseed, safflower, sunflower	Canola
Sorghum	Average for all grain crops
Soybeans	Dry beans
Misc. veg and fruits	Average for all fruit, vegetable crops
Other crops	Average for all crops
Other small grains	Average for all grain crops
Other tree fruits	Average for all tree fruit crops
Other tree nuts	Average for all tree nut crops

nitrate, applied to any Oregon soil would affect groundwater. OWQDA models leaching potential of soil as a function of throughflow potential, runoff potential, and hydraulic loading. Throughflow potential is an empirical rating of time required for water to pass through soil to groundwater. Runoff potential is an empirical assessment based on three variables: surface soil permeability, depth to restrictive layer, and soil slope. Regarding hydraulic loading, two corrections were necessary to account for widely divergent climates of Oregon: one for amount of rainfall in the region, and one for change in hydraulic loading due to irrigation.

The OWQDA tool distinguishes five classes of groundwater vulnerability ranging from very low to very high, and these ratings are carried over into the SSNL dataset. DEQ used the SSNL dataset to identify factors influencing nitrate risks for public water systems (Seeds 2011). We used the SSNL dataset to identify private wells located in soils that have 'very high' or 'high' sensitivity to nitrate leaching (Figure 3). Ratings of soil sensitivity for irrigated soil were used in the model. While the SSNL data layer covers the majority of the state, the State Soil Geographic (STASGO2) data were used to determine sensitivity ranking where SSNL data were unavailable.

New well construction

Oregon Water Resources Department (OWRD) permits new well construction and maintains this information in a publicly accessible well log database (OWRD 2007). Well logs for new wells drilled for domestic use between 2000 and 2007 were used to locate wells across Oregon and within GWMA (Figure 4).

Numerous studies demonstrate that buffer size around a sampled well is significant when evaluating groundwater quality in relation to land use (Cain *et al.* 1989; Barringer *et al.* 1990; Tesoriero & Voss 1997). We chose a buffer radius of 1,000 meters for establishing an area of influence following work by Greene *et al.* (2005). This buffer radius was applied to each well and the total sum of manure-N and fertilizer-N allocated to cells within the buffer area (approximately 314 hectares) was derived using an algorithm created with Python programming language.

Datasets from Table 1 were uploaded to a single GIS database in order to generate summary information and

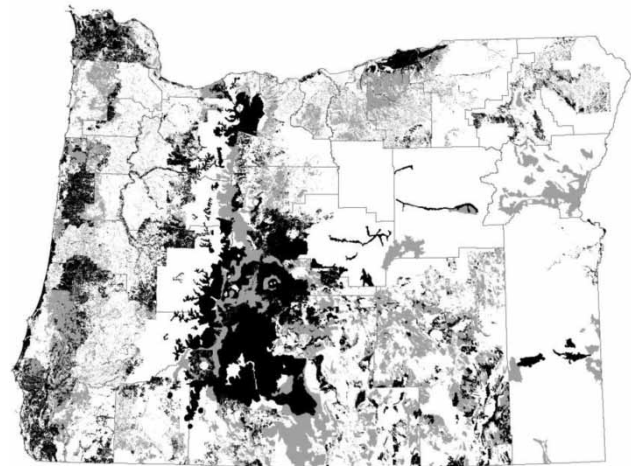


Figure 3 | Map of 'very high' (black) and 'high' (gray) SSNL.



Figure 4 | New wells drilled between 2000 and 2007 and locations of GWMA.

produce maps associating nitrogen distribution (contaminant source) with new wells (potential exposure points) for understanding health risks related to consumption of nitrate in drinking water.

RESULTS

Manure-N

Using the approach by Ruddy *et al.* (2006), we calculated that approximately 88 million kilograms of manure-N were applied across Oregon in 2007. This is an increase of approximately 10 million kg from 1997, the most recent

year for which Ruddy *et al.* (2006) calculated manure-N input across Oregon from confined and unconfined farms. Alternatively, using data from state CAFO permits, we calculated that approximately 54 million kg of manure-N would have been applied in any one year between 2000 and 2007. For 75% of counties, manure-N estimates were larger using the Ruddy approach. The Ruddy approach relies on reports of animal type and number from the 2007 Census of Agriculture, while the CAFO permit approach relies on information submitted to the state under a regulated program. Given the vague definition in statute for the type of facility considered a CAFO (ORS468B 2009) and that the CAFO approach did not account for ranches or farms with free-ranging animals, it is likely that the CAFO approach underestimates the true impact of manure-N across Oregon. For example, based on the CAFO dataset used in this study, approximately 750 farms were used to estimate and distribute annual manure-N inputs. Conversely, nearly 20,000 Oregon farms reported livestock or poultry inventory in the 2007 Census of Agriculture. However, over 80% of these farms produced cattle, which are often free-ranging and not likely to be a significant localized source of manure-N. Yet the discrepancy suggests that there are farms missing from the CAFO dataset, either because they do not meet state criteria for permitting, or because operators are out of compliance with statutory requirements.

After distributing manure-N estimates across the state, values representing annual manure-N loading ranged from approximately 0 to 605 kg per receiving cell. Each cell is approximately one-third of a hectare. Areas with the highest values of manure-N were in Tillamook County, which contains 20% of the state's dairy farms. Cells with the highest manure-N values were located within a 12 km² area containing more than 90 dairy or beef farms in operation between 2000 and 2007. The high density of farms in this small area is the likely cause of high manure-N loading.

Fertilizer-N

Using the approach by Ruddy *et al.* (2006), we calculated that approximately 178 million kg of fertilizer-N were applied across Oregon in 2007. This is an increase of approximately 70 million kg from 2001, the most recent year for which Ruddy *et al.* (2006) calculated fertilizer-N input across

Oregon. Alternatively, using EBS data and crop type identified in the 2007 OR-CDL, we calculated that approximately 117 million kg of fertilizer-N would have been applied during that year. For over 80% of counties, fertilizer-N estimates were larger using the Ruddy approach. The Ruddy approach relies on reports of fertilizer sales and use provided by regulatory agencies and farmers, while the ESB approach relies on expert guidance for crop-specific fertilizer use. The discrepancy between these two estimates may be evidence of the widely recognized problem of fertilizer application rates that exceed well-researched agronomic needs (Hatfield & Prueger 2004; Hansen *et al.* 2011), in particular as the amount of agricultural land in production decreased by more than 240,000 hectares from 2002 to 2007 (USDA 2009).

Alternatively, the difference may be a consequence of misidentification of cropland within the OR-CDL. After distributing fertilizer-N estimates across Oregon, values representing annual fertilizer-N loading ranged from approximately 0 to 1,598 kg/ha, with mean and median values of 197 and 182 kg/ha, respectively, when all 0-valued cells were excluded. Such values are typical amounts of nitrogen applied to most crops grown in Oregon. Areas with extremely high fertilizer-N values were located mainly within Hood River County, which accounts for approximately 2% of Oregon's gross farm and ranch sales (NASS 2013). The Ruddy approach for estimating fertilizer-N indicated that this county used less than 1% of the state's total fertilizer use in 2007. While a small percentage, by our calculations the 2007 OR-CDL identifies approximately 150 hectares of cropland in Hood River with the result that this relatively small amount of cropland received all the fertilizer-N estimated for the county. While we only saw these extreme fertilizer-N values in one county, the situation underscores the importance of accurately identifying land use and crop type for estimating and distributing nitrogen from agricultural sources.

The CDL and NLCD datasets are commonly used for modeling land use impacts to drinking water sources (Greene *et al.* 2005; Mehaffey *et al.* 2011). While the CDL borrows non-agriculture land use data from the NLCD, it uses a separate, unique process to identify agricultural land use, particularly for cropland (Mueller & Ozga 2002). We were interested in how cropland estimates for Hood River compare

between the NLCD and CDL, and compared cropland estimates for each Oregon county in order to understand how these two datasets match regarding estimation of agricultural land use. We calculated total square kilometers of land identified as pasture or cropland between the 2007 OR-CDL and the 2006 NLCD and found that for all but four counties (Gilliam, Linn, Morrow, Sherman), the 2006 NLCD identified on average nearly 183 km² of land as agricultural over the amount estimated in the 2007 OR-CDL.

The average percent difference was greater between the two estimates of cropland (20%) than between the two estimates of pastureland (0.05%). With regard to Hood River County, the 2006 NLCD identified approximately 29,500 more hectares of cropland than the 2007 OR-CDL. Using the NLCD to distribute fertilizer-N to cropland would have resulted in approximately 17 kg N per cell (approximately 56 kg N/hectare-year). This is a more realistic estimation compared to an average of 2,550 kg N per cell (approximately 8,333 kg N/hectare-year) using the 2007 OR-CDL, especially given that the majority of cropland in Hood River County is identified in the 2007 OR-CDL as tree crops (fruit, nut, holiday trees), which require low levels of nitrogenous fertilizer.

CDL crop areas are based on sample observations of what has been planted and then extrapolated to remaining areas for which training data are missing. Overall accuracy for the 2007 OR-CDL is 83%, with an error value of 17% (USDA NASS 2007). NLCD bases crop estimates on satellite imagery and does not use field-level training data. Thus, in most cases the CDL would have more accurate data for hectares actually in production and the specific type of crop grown. However, a possible exception would be an area with relatively little data for training (J. Dewitz, NASS, personal communication, September 26, 2010), such as cropland in a mostly forested area, which may be the case in Hood River given that over 70% of the county is forested. At this time, it is difficult to determine the reason for extremely high fertilizer-N loading on cropland in Hood River. Our experience using the OR-CDL for this study underscores the need to be cautious about taking land use designations at face value. While extreme anomalies, such as we encountered in Hood River, may be more the exception than the norm, it is especially important to explore and report on these exceptions when downscaling large datasets to small area units.

We combined the final fertilizer-N and manure-N rasters to derive a map representing total nitrogen loading from agricultural inputs across Oregon (Figure 5). We capped the total amount of nitrogen from fertilizer and manure that could be attributed to any single cell at 600 kg, a conservative estimate. Loading rates that exceeded this cap were only present in a few pixels near groups of multiple CAFOs in close proximity (Tillamook County). In the absence of 'ground-truthing' locations and periods of operation for these CAFOs, we assumed that reported information on these farms in the CAFO dataset was accurate, but that animal farms in such close proximity would need to transport manure further afield than 16 km from the facility, a noted assumption of our model.

Identification of wells susceptible to nitrate contamination

A total of 30,893 new private wells were drilled between 2000 and 2007, based on well logs with duplicates removed (Figure 4). This result reflects state agency estimates for annual gains in new wells (OWRD 2008). DEQ estimates that there are over 350,000 private wells in Oregon, which suggests that our model captured 9% of active wells in the state (DEQ 2011). Of this total, 1,307 (4%) were drilled within a GWMA, areas proven through extensive sampling to have elevated levels of groundwater nitrate. After combining all datasets in a GIS database, only wells with a buffer area including soil with 'very high' or 'high' sensitivity to nitrate leaching and a nonzero sum for combined manure-N

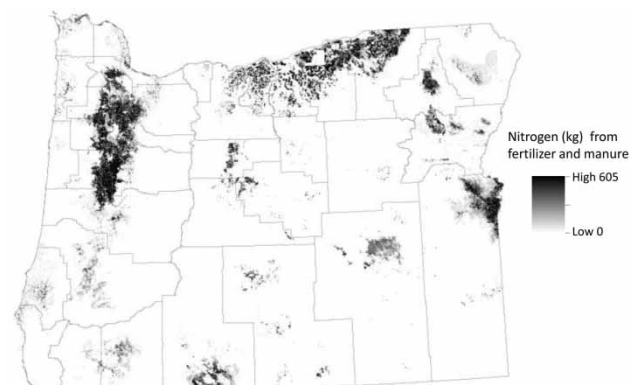


Figure 5 | Total annual nitrogen (kg) from manure and fertilizer applications.

and fertilizer-N ('selected' wells) were retained ($n = 20,488$) (Figure 6). The average amount of annual nitrogen loading within buffers of selected wells ranged from negligible to 789 kg N/ha. The highest estimate was associated with a well located in Tillamook County in an area of intensive animal farming. We assessed the distribution of annual nitrogen loading to selected wells according to quartiles. The highest quartile included values of 47 to 789 kg N/ha within well buffers. To identify 'hot spots' of susceptible wells, we focused on ranges within this quartile, aggregating values into three groups (47–99, 100–199, 200–789), based on the premise that annual nitrogen loads up to 200 kg N/ha are common agronomic practice. Figure 6 shows the distribution of selected wells across Oregon color-coded according to average annual nitrogen loading. We considered wells associated with soils with very high or high sensitivity to nitrate leaching and average annual nitrogen loads surpassing 200 kg N/ha as potentially susceptible to nitrate contamination. These susceptible wells totaled 713 or 4% of selected new wells drilled between 2000 and 2007. Notably, the two GWMAs located in the center and eastern parts of the state appear to 'capture' a large number of susceptible wells in the area. However, the GWMA located in the southern Willamette Valley is far

too restricted in area to include all wells identified as susceptible to contamination.

GWMAs are outlined in black. Wells in black are considered susceptible to nitrate contamination, wells in dark gray are cautionary, and wells in light gray are least susceptible. Selected wells in the Willamette Valley and their relationship to the GWMA are highlighted in the small map on the left.

DISCUSSION

This study demonstrates an approach for increasing spatial resolution of data representing agricultural nitrogen inputs affecting nitrate contamination in private wells at a statewide scale. GIS modeling is a valuable, low-cost alternative to identifying areas of concern related to drinking water exposures (Swartz *et al.* 2003) and can facilitate the 'risk ranking' promoted by WHO guidelines for drinking water system safety assessments (WHO 2011). Results of this modeling work can assist land use planners, water quality managers, and public health specialists in decision-making regarding well permitting, source water protections, and implementation of well stewardship programs (Rowe *et al.* 2007). For example, our

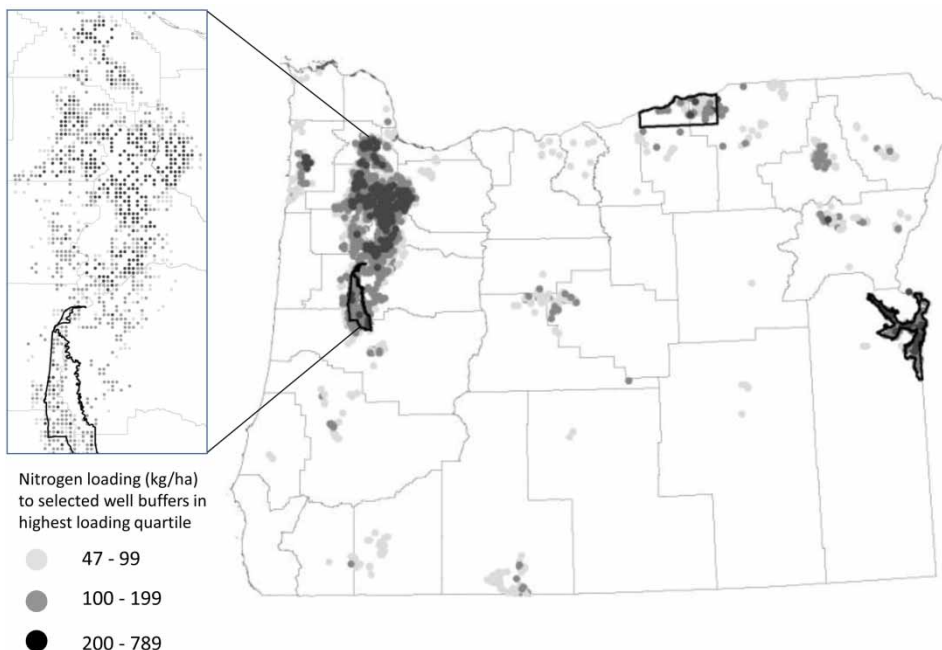


Figure 6 | Selected wells in 'hot spots' of average annual nitrogen loading (kg/hectare).

analysis indicates that the existing boundaries of the GWMA located in the southern Willamette Valley do not include all susceptible wells in the area. Given that designation of a GWMA triggers activities to educate affected residents and mitigate contamination, it may be advantageous from a risk-reduction perspective to expand the boundaries of this GWMA, specifically to include susceptible wells to the north.

While we considered developing maps showing total manure-N and fertilizer-N per census tract in comparison to new well construction, we were concerned that this would provide a misleading view of susceptible wells at this level. Aggregating contaminant sources to various areal units is a common approach for characterizing population-level environmental exposures and exploring links to health outcomes (Elliott & Wartenberg 2004; Nuckols *et al.* 2004; Parenteau & Sawada 2011). This approach facilitates linking exposures and health outcomes to socioeconomic and demographic (SED) information available at various census levels (Teschke *et al.* 2010; Balazs *et al.* 2011). An understanding of SED characteristics of Oregonians relying on susceptible wells would assist public health professionals in effectively targeting education efforts to these populations. However, aggregating contamination that can vary substantially over short distances and assigning a sum to a large areal unit, like a census tract, may wrongly infer that all individuals living in the unit are equally exposed to the same contamination. This is referred to as the Modifiable Area Unit Problem (MAUP) and is a common challenge for studies on health and the environment (Elliott & Wartenberg 2004; Nuckols *et al.* 2004; Parenteau & Sawada 2011). Our efforts to use state-level data to characterize fine-scale variability in agriculturally sourced nitrogen allow for understanding contamination and potential exposures at point locations across Oregon. Aggregating all manure-N and fertilizer-N within a census tract and inferring an association with all wells located within the tract is a step backwards from the fine-scale modeling of our data and may demonstrate a mishandling of MAUP.

The model developed in this study was used to evaluate validity of a state-level dataset of private well tests for supporting a sentinel public health surveillance system, such that a report of contamination in a single well could be interpreted as representing contamination in proximal wells. A major assumption underlying this study is that nitrate contamination in wells is temporally and spatially linked to

agricultural nitrogen inputs. However, studies report a wide range of spatial and temporal delays between agricultural practices and groundwater contamination (Cain *et al.* 1989; Barringer *et al.* 1990; Tesoriero & Voss 1997). We acknowledge that because groundwater nitrate may persist for long periods of time under certain aquifer conditions and may travel long distances from source areas, agricultural inputs that are outside the temporal and spatial consideration of the model may not be adequately represented in the results. In addition, we did not test sensitivity of the model results to variations in nitrogen threshold or buffer size. Our goal was to develop methods for high resolution exposure characterization, and we recommend further model developments to refine the method for sensitivity to inputs. Given that our modeling efforts were focused on nitrate, the approach presented here may not be transferable to other contaminants that are a concern for private wells but may not be as closely linked to agricultural land use, such as radon or arsenic.

The likelihood that private well owners and their families will be exposed to high levels of agricultural contaminants through drinking water increases as residential development encroaches on historical farmland, especially given that groundwater nitrate can persist and accumulate over decades, even centuries (Showers *et al.* 2008; Schlesinger 2009; Dubrovsky *et al.* 2010). However, research shows that regulation and technical improvements in farming can succeed in decreasing nitrogen surplus in groundwater while maintaining crop yields and increasing animal production (Kladivko *et al.* 2004; Hansen *et al.* 2011). Most construction regulations do not address the problem of existing groundwater contamination when agricultural lands are converted to residential developments. Our modeling results suggest that more research is needed to characterize the quality of source water before homes dependent on private wells are built. An immediate action to protect private well owners is to require water quality testing of newly constructed wells as part of the permitting process, a policy that currently does not exist in Oregon at the state level (Showers *et al.* 2008). The cost to a home owner to install an effective treatment system to remove nitrate from well water can easily exceed \$2,500, not including annual maintenance. The decision to transition long-standing arable land from agricultural to residential use should include an open assessment of the short- and long-term costs inherent in treating affected groundwater tapped for new homes.

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