Environ Monit Assess (2009) 156:343–360
DOI 10.1007/s10661-008-0489-x

Multi-scale landscape factors influencing stream water quality in the state of Oregon

**Maliha S. Nash · Daniel T. Heggem ·
Donald Ebert · Timothy G. Wade ·
Robert K. Hall**

Received: 14 January 2008 / Accepted: 24 July 2008 / Published online: 30 August 2008

Abstract Enterococci bacteria are used to indicate the presence of human and/or animal fecal materials in surface water. In addition to human influences on the quality of surface water, a cattle grazing is a widespread and persistent ecological stressor in the Western United States. Cattle may affect surface water quality directly by depositing nutrients and bacteria, and indirectly by damaging stream banks or removing vegetation cover, which may lead to increased sediment loads. This study used the State of Oregon surface water data to determine the likelihood of animal pathogen presence using enterococci and analyzed the spatial distribution and relationship of biotic (enterococci) and abiotic (nitrogen and phosphorous) surface water constituents to landscape metrics and others (e.g. human use, percent riparian cover, natural covers, grazing, etc.). We used a grazing potential index (GPI) based on proximity to water, land ownership and forage availability. Mean and variability of GPI, forage availability, stream density and length, and landscape metrics were related to enterococci and many forms of nitrogen and phosphorous in standard and logistic regression models. The GPI did not have a significant role in the models, but forage related variables had significant contribution. Urban land use within stream reach was the main driving factor when exceeding the threshold ($\geq$35 cfu/100 ml), agriculture was the driving force in elevating enterococci in sites where enterococci concentration was $<$35 cfu/100 ml. Landscape metrics related to amount of agriculture, wetlands and urban all contributed to increasing nutrients in surface water but at different scales. The probability of having sites with concentrations of enterococci above the threshold was much lower in areas of natural land cover and much higher in areas with higher urban land use within 60 m of stream. A 1% increase in natural land cover was associated with a 12% decrease in the predicted odds of having a site exceeding the threshold. Opposite to natural land cover, a one unit change in each of manmade barren and urban land use led to an increase of the

---

M. S. Nash (✉) · D. T. Heggem · D. Ebert
U.S. EPA, P.O. Box 93478, Las Vegas,
NV 89193-3478, USA
e-mail: nash.maliha@epa.gov

D. T. Heggem
e-mail: Heggem.Daniel@epa.gov

D. Ebert
e-mail: Ebert.donald@epa.gov

T. G. Wade
U.S. EPA, E243-05, Research Triangle Park,
NC 27711, USA
e-mail: Wade.Timothy@epa.gov

R. K. Hall
U.S. EPA Region 9, 75 Hawthorne Street,
San Francisco, CA 94105, USA
e-mail: Hall.robertk@epa.gov

likelihood of exceeding the threshold by 73%, and 11%, respectively. Change in urban land use had a higher influence on the likelihood of a site exceeding the threshold than that of natural land cover.

Keywords Nutrients · Nitrogen · Phosphorus · Enterococci · Grazing potential index · GPI · Logistic regression · Oregon

Introduction

Composition of land use and land cover in a watershed has a direct effect on the condition of streams. Surface water quality is influenced by the extent of anthropogenic and animal (domestic livestock, feral animals and wildlife) inputs within a watershed. Commercial livestock grazing comprises approximately 70% of the land use in the arid western United States (Fleischner 1994). Over the years, lack of adequate grazing management has damaged more than 80% of the stream and wetland riparian ecosystems (Belsky et al. 1999, 2002). Livestock tend to be located in areas where food and water are available (Kauffman and Krueger 1984). Extreme disturbance (both from grazing and trampling) and livestock waste can be found adjacent to the water with disturbance (primarily from grazing) progressively decreasing as distance increases away from the water source. This pattern of disturbance has been shown an effect on vegetation characteristics (Lange 1969; Graetz and Ludwig 1978; Andrew and Lange 1986; Fusco et al. 1995), vegetation patterns (deSoyza et al. 1997; Nash et al. 1999), and soil microtopography (Nash et al. 2003).

The amount of nutrients and pathogens originating from livestock is dependent on terrestrial hydraulic transport mechanisms to surface water. In the arid part of Eastern Oregon, the water table can be contaminated by infiltration of nutrients and pathogens through the overlying soil. This infiltration process is a function of the vegetation cover and structure, evapotranspiration, soil properties, and the number of grazing animals (e.g., horses, cows, wildlife). In the western part of Oregon, streams are more abundant, and livestock tend to concentrate in areas adjacent to streams depending on season, age class and type (VanWagoner et al. 2006). Input of nutrients and pathogens from livestock delivered to surface water is a function of livestock abundance and the proximity of their forage to stream (MacLusky 1960; Marsh and Campling 1970; Betteridge et al. 1986).

A need has been recognized to develop a method(s) for the assessment and evaluation of surface water nutrient and pathogen levels in the Western United States as a result of urbanization, livestock grazing and agriculture activities. Because information on location and number of cattle grazing is limited, this effort focuses on creating a grazing potential index (GPI) map of Oregon as a case study (Wade et al. 1998). The GPI uses, a combination of land cover, land ownership, distance to water, and slope to map the relative likelihood of presence of grazing cattle. GPI values range from 0 to 100, with higher values representing greater grazing potential.

Fecal bacteria (coliforms and streptococci) live in human and animal digestive systems and are used as indicators of possible sewage and nonpoint source contamination. Drinking or coming in contact (i.e., ingested and/or enter the skin through a cut or sore) with surface water contaminated with fecal material can infect humans with many diseases (e.g. skin rashes, urinary tract infections, meningitis). Certain types of bacteria are indicators of fecal material in the water column. Fecal coliform bacteria is normally measured in surface water to determine the level of contamination. Recently, the United States Environmental Protection Agency (USEPA) recommended using enterococci bacteria to indicate the presence of human and/or animal fecal materials (USEPA 2000). Water is safe for drinking when a single sample contains no more than 104 colony formed units (cfu) per 100 ml, or for multiple samples (from at least 24 h time intervals) a geometric mean of 35 cfu/100 ml for freshwater (APHA 1998).

The concentration of nitrogen and phosphorous in surface waters are also found to be related with human or animal presence. Nitrogen and phosphorous in water may come from sources such as runoff from agriculture land, rural and urban areas, waste water and auto exhausts, as well as livestock. The role and magnitude of each of these processes varies mainly with a source's proximity to streams. While, there is no precise estimate of nitrogen and phosphorous inputs from each source into streams, one study indicated that the contribution of ground water (45%) to nitrogen level in Lake Mendota, Wisconsin was much higher than from precipitation (17%) and from rural runoff (9%) (Olson et al. 1971). A different pattern was found for phosphorous, where 52% came from rural runoff due to the use of manure. In industrial populated areas, nitrogen from wet and dry deposition is much higher than from other processes. Phosphorous is highly correlated to sediment loads (Jarvie et al. 2006; Oenema et al. 2005; Mitchell et al. 2004), because of its affinity to bind to fine grained sediment. Twice as much, nitrogen comes from farmland as from forestland near Coshocton, Ohio (Taylor et al. 1971).

The purpose of this study was to relate the concentration of enterococci, nitrogen and phosphorous in surface waters to anthropogenic and livestock activities on stream and river systems within the State of Oregon's watersheds using landcover as the primary input. Objectives of this study are to investigate the following:

- Does the enterococci concentration in streams reflect the condition and composition of the surrounding landscape?
- Will sites with geometric mean of enterococci less than 35 cfu/100 ml be correlated to different constituents than sites with a geometric mean of more than 35 cfu/100 ml?
- What is the likelihood that a site with a high concentration of enterococci is due to land use and anthropogenic sources?

To answer the above questions, surface water enterococci and many forms of nitrogen and phosphorous, for years 1990 through 1994 (STORET, EPA's Main Repository for Water Quality, biological and Physical data. http://www.epa.gov/STORET/), were related with watershed landscape and forage metrics and the GPI using standard and logistic multiple regressions. The contribution of each predictor to the response (enterococci, nitrogen or phosphorous) was determined by their significance and weight in each model. This can identify the level of human, animal or natural effects. Prior to the regression analyses, changes of enterococci concentration with time were analyzed to determine the temporal trend. This retrospective monitoring determines the degraded, improved or no-change status of a watershed condition in biotic or abiotic levels.

Study area

The State of Oregon is one of several states which participated in the USEPA Western Environmental Monitoring and Assessment Program (Western EMAP) (Jones et al. 2000). Oregon encompasses 251,415 km^2 in surface area with a wide range of elevation and vegetation cover from the coast on the west to the dry land (xeric landscape) in the east. Elevations ranged from sea level at the coast to 3,426 m at Mount Hood. Climate data for the state of Oregon, spanning the last 100 years, indicates cycles of 20–25 years of wet/dry periods. Dry periods were noted in the years from 1920 through 1945 and from 1975 through 1994. A wet cycle appears to have begun in 1994 and continues to date. Oregon spans nine climate zones (Fig. 1) (OCS 2008). Winter months experience the most rainfall, except for zone 8 where the rainfall is uniform throughout the year and in the eastern part of zone seven (South central) where the highest rainfall is in spring and summer. Annual rainfall ranges from <250 mm in the eastern portion of the state, which is comprised mainly of shrubland and dry land farming, to ≥4,600 mm in the forested west. For the time period from 1990–1994, rainfall in the eastern part of the state is almost a tenth of the western part (Table 1). Annual rainfall and rainfall in a growing season showed no increasing or decreasing trend between 1990 and 1994. The growing season rainfall constitutes higher proportion of the annual rainfall in the eastern part than that of the western (Table 1). Oregon's annual temperatures range from −1° to 114° F, with a mean of 55° F.

Vegetation cover is a gradient from mountain forest and valley agriculture in the west to dry land dominated by shrubland in the east central and woodland in the northeastern corner. Forested upland covered approximately 44% of the area (2% deciduous forest, 38% evergreen forest, and 4% mixed forest) and shrubland covered 35%.

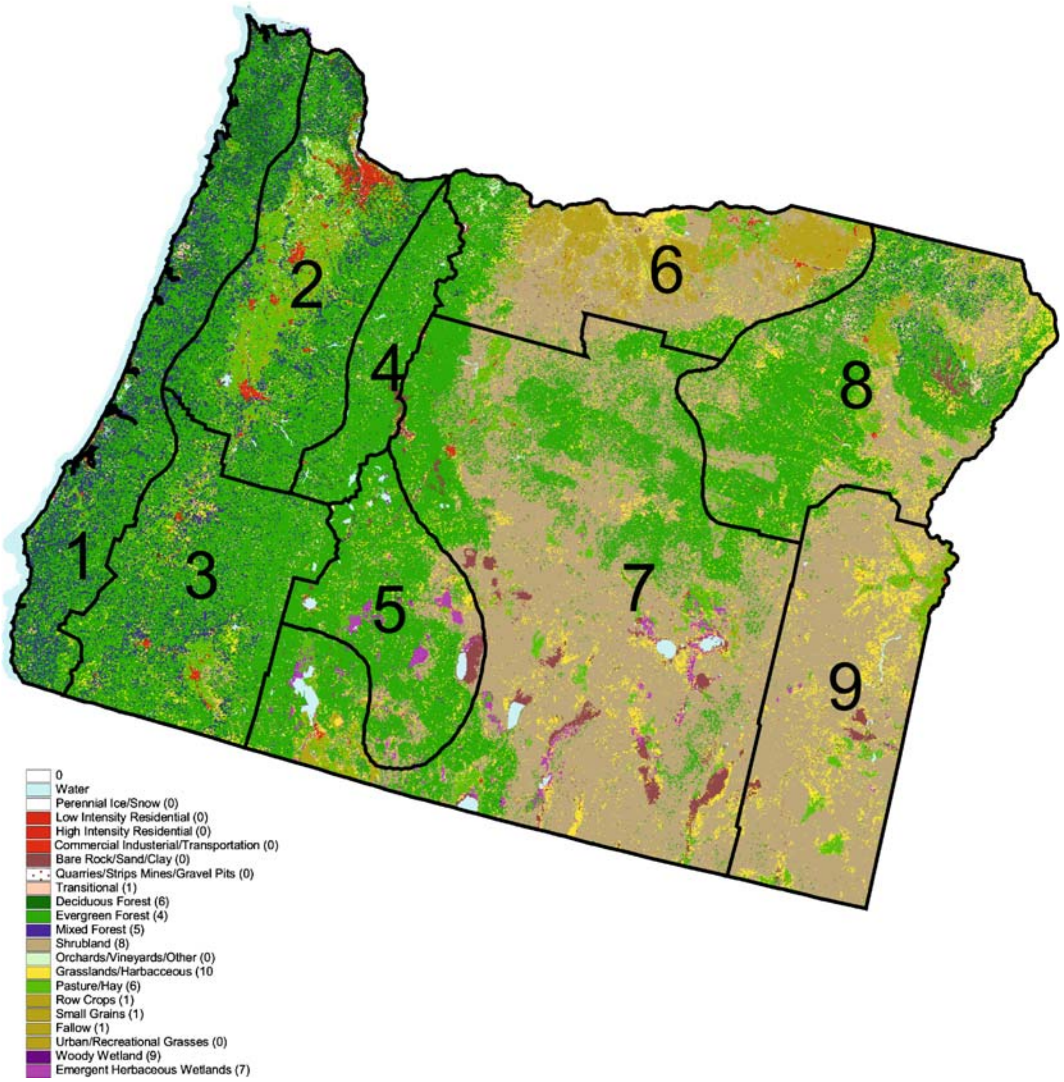

Fig. 1 1992 national landcover (30 m) for the Oregon state. *Number* insides parentheses next to landcover classification denotes forage availability ranks. Marked polygon represents the climate divisions (Table 1, OCS 2008)

Fifteen percent of the land is agricultural (Fig. 1). Land cover proportions were derived from National Land Cover Data (NLCD) (Vogelmann et al. 2001) for 1992.

Aridity increases from west to east across the study area with corresponding changes in dominant land cover and land use activities. Forest dominates along the coast and gives way to shrub-dominated areas toward the east. Most of the agriculture in Oregon (fruits, nuts, berries, mint, grains, and hay) is located in the Willamette Valley where the three largest cities (Portland, Salem, and Eugene) are located. Livestock and agriculture (alfalfa, grass, legume seed crops, and wheat) occur east of the Cascade Mountain Range. The south central area, comprising the high desert prairie interspersed with a number of mountain ranges and isolated peaks, is mainly used for livestock (cow, sheep, horses, and pigs), dry land farming, and to a lesser extent, irrigated–supported agriculture. Crops grown include potatoes, alfalfa, hay, mint, wheat, oats, barley, and onions. The northeastern corner of the state is comprised of large volcanic mountain

Table 1 Annual and growing season (June–September) precipitation for each climate region in Oregon State for 1990–1994

Region	*N*	Dominant landcover	Precipitation mm/year (mm/4 months)		Seasonality
			Mean	Std	
1	8	Forest	2,263 (146)	754 (60)	Winter
2	37	Agriculture	1,135 (87)	306 (35)	Winter
3	3	Forest	1,218 (112)	317 (30)	Winter
4	8	Forest	1,728 (139)	417 (37)	Winter
5	4	Forest	421 (58)	202 (5)	Winter
6	7	Shrubland	179 (33)	33 (9)	Winter
7	15	Shrubland/dryland farming	179 (40)	64 (10)	Winter/spring
8	8	Forest	224 (58)	67 (9)	Uniform monthly
9	15	Agriculture/farming	215 (45)	65 (13)	Uniform yearly

Number in parentheses is the precipitation for the 4 months in the growing season (June–September). Rainfall of more than 6.4 mm per event was included in the data below

n Number of rainfall stations per each climate region, *Std* standard deviation

ranges surrounding fairly broad valleys. Several million acres of Federal land are being utilized by ranchers for raising livestock.

Data

Water quality variables

Water quality data were obtained for the years 1990 through 1994 (STORET, EPA's Main Repository for Water Quality, biological and Physical data. http://www.epa.gov/STORET/), which were within 2 years (1992) of the Natural Land Cover Data (NLCD; Vogelmann et al. 2001) for the study area. For landscape water quality relationships, only data for the growing season (June–September) were used. To ensure adequate temporal and spatial coverage, sites with a minimum five measurements during the study period of 1990–1994 are used. Some sites were sampled more frequently than others. A total of 485 sample sites were used in the analyses. Biotic and abiotic measurements (enterococci and forms of nitrogen and phosphorous) from surface water samples were used in the analyses. Nitrogen forms were total nitrite + nitrate, total dissolved ammonia + ammonium, total Kjeldahl nitrogen (TKN = organic nitrogen + ammonia) and total nitrogen (TN = TKN + NO_2 + NO_3). Phosphorous forms were total Phosphorous (TP) and dissolved orthophosphate as phosphorous (DOP). Nitrate (NO_3^-) and ammonium (NH_4^+) are the inorganic forms that are water soluble and are available for plant uptake.

Enterococci were sampled frequently and number of visits per site which ranged from 1 to 206. For samples collected on an hourly basis, a daily arithmetic mean for enterococci concentration was calculated. Sites with at least five means for (1990–1994) were used to derive the geometric mean for the analyses and to quantify responses of enterococci to GPI and landscape metrics. The geometric mean is recommended to reduce the influence of extreme values. The arithmetic mean of nitrogen and phosphorous as an average value is used in the analyses.

Landscape variables

Watersheds, or upstream contributing areas, were delineated for each of the 485 sample sites. 1992 NLCD was used with the Analytical Tools Interface for Landscape Assessments (ATtILA; Ebert and Wade 2004) ArcView® extension to generate landscape metrics for each watershed. Metrics included percentage of crop land (agc), percentage of pasture (agp), percentage of all agricultural use (agt), percentage of forest (for), percentage of man-made barren (mbar), percentage of natural barren (nbar), percentage of natural grassland (ng), percentage of shrubland (shrb), percentage of urban (urb), percentage of wetland (wetl). Identical metrics were calculated for riparian zones using 0, 30 and 60 m stream buffers for all streams in the watershed and for areas within 300 and 600 m of the sample site (pour point). Total stream length and stream density were also calculated for each watershed. High resolution (1:24,000 scale) National Hydrography Data (NHD; USGS 2008a) was used for all stream metrics.

Grazing potential index

Availability of spatially explicit data for cattle grazing is limited. The number of cattle grazing in a given area often changes from year to year, or even seasonally. The GPI map was developed by the EMAP landscape team to rank the relative potential for cattle grazing (Wade et al. 1998), but does not quantify the number or duration of cattle on the range.

Water availability was used as a mask to isolate areas where cattle were likely to be located and have a significant impact on water quality. Other areas were excluded from the analysis. Spatial coverage of water sources were derived from three primary data sets. These three data sets have some redundant features, which did not affect the results. These data were used because water availability is a crucial factor in cattle location, particularly in the arid and semi-arid west. High resolution NHD (USGS 2008a) was used to define streams and shorelines line segments. Point locations for lakes, reservoirs and springs were derived from the US Geological Survey's Geographic Names Information System (GNIS; USGS 2008b), and surface water features from the 1992 NLCD (Vogelmann et al. 2001). Together, the three data sets capture most permanent water sources. These data were not readily available in 1998 and represent a significant improvement to the method in Wade et al. (1998).

The water sources data is used as starting points to generate a cost path using distance and percent slope. Cattle are known to avoid steep slopes, but will travel further from water on flat or low slopes to obtain forage (Roath and Krueger 1982; Cook 1966). A cost path assigns a value to each cell representing the cost to traverse that cell. A maximum cost is set as a threshold, beyond which no further travel is allowed. The maximum cost used in the index was 211. Table 1 shows cost assigned based on slope. Crossing a cell in a cardinal direction costs 30 (the length of the cell in meters) times its slope weight. Crossing diagonally costs approximately 42 (the diagonal length of the cell) times the slope weight. For example, on flat ground, cattle could travel at most seven cells (30 [distance in meters] × 1 [slope cost] × 7 [number of cells] or 210 m) from a water source before reaching the cost threshold. Cells located less than 211 cost units from a water source defined the area of analysis. All cells beyond the threshold were excluded from the analysis.

Slope weights were assigned based on literature. Roath and Krueger (1982) found cattle did not use slopes over 60%, while Gillen et al. (1984) found the limit to be 20%. Pinchak et al. (1991) observed cattle on slopes up to 40%, but 90% of use was on slopes less than 7%. Mueggler (1965) recorded relative amount of time spent by cattle at different distances from slope bottoms. Even moderate slopes (10% or more) greatly curtailed cattle presence just 100 yards upslope. To allow a less restrictive distribution of cattle, 60% was chosen for the maximum slope. Slopes above 60% were considered non-traversable.

The two primary components in the GPI index are forage availability (WT's; Table 2) and land ownership. Availability of forage is estimated using 1992 NLCD (Vogelmann et al. 2001). Each land cover type is assigned a weighting factor (i.e., value) ranging from zero to ten representing potential quantity and quality of forage. Values are based on the experts' knowledge (i.e., professional judgment) of the area. Higher values represent better forage (Table 3). Values of zero were considered ungrazeable.

US Geological Survey National Atlas (USGS 2008c) is used to determine land ownership. The Federal Lands layer is used as a surrogate for management practice. GPI weighting factors are based on experts' knowledge of the area where higher values more likely to be grazed (i.e., 1 = very unlikely to be grazed, or only grazed under limited conditions; 5 = grazing restrictions varied greatly between units, with no way to distinguish between them; 10 = few or no restrictions on grazing). For example, national parks are assigned a one, national forests were assigned fives and Bureau of Land Management and private lands were given a ten.

Table 2 Weights of land cover for forage availability for livestock used for GPI

Land cover	Abbreviation	Weight
Open water, ice/snow, mining, low & high intensity residential, commercial/industrial/transportation, bare rock/sand/clay, quarries/strip mines/gravel pits, orchard/vineyards/other, and urban recreational grasses	W0	0
Transitional, row crop, small grain, and fallow	W1	1
Evergreen forest	W4	4
Mixed forest	W5	5
Deciduous forest, pasture/hay	W6	6
Herbaceous wetland	W7	7
Shrub land	W8	8
Woody wetland	W9	9
Grasslands/herbaceous	W10	10

Weight values ranged from 0 (not available) to 10 (highly available)

The GPI is a simple multiplication of the two input layers (NLCD and land ownership), resulting in scores of 0 to 100. Higher values represent higher grazing potential (Fig. 2). Using ATtILA (Ebert and Wade 2004), average, standard deviation (Fig. 2b, c), and coefficient of variability of GPI per watershed was calculated to be used in the regression model. Forage availability related land cover variables were also used in the models (W0, W1, W4–W10; Table 3).

Statistical analyses

A site was identified as "BT; below threshold" if the geometric mean of the enterococci concentration was less than 35 cfu/100 ml (<35 cfu/100 ml). A site was indentified as "AT; above threshold" when the geometric mean concentration was greater or equal 35 cfu/100 ml (≥35 cfu/100 ml). Hereafter, "AT" and "BT" will be used for group identification.

The temporal trend in enterococci demonstrates present surface water status as compared with the past based on the condition of surrounding areas. Hence, temporal trend of enterococci levels were determined using the general linear model (Genral Linear Model, proc GLM in SAS® 9.1) to follow changes in enterococci level in selected sites. There are 279 sites which had enough temporal samples to be used for these analyses.

Table 3 Land cover derived from ATtiLA used in standard regressions in Table 4

Metrics	Description
mbar	Percent of reporting unit that is man made barren
nbar	Percent of reporting unit that is natural barren
agp	Percent of reporting unit that is pasture
agc	Percent of reporting unit that is crop land
agt	Percent of reporting unit that is all agriculture use
for	Percent of reporting unit that is forest
ng	Percent of reporting unit that is natural grassland
nat	Percent of reporting unit that is all natural land use
wetl	Percent of reporting unit that is wetland
urb	Percent of reporting unit that is urban
shrb	Percent of reporting unit that is scrubland
_0	Percent of a stream length adjacent to a specific land cover (Riparian zone metric)
_60	Percent of a specific land cover in 60 m stream buffer area
_300	Percent of a specific land cover within 300 m circular segment of a sample
_600	Percent of a specific land cover within 600 m circular segment of a sample
strmdens	Stream density reported as km of streams/area of reporting unit in km^2
strmlen	Total stream length in map units

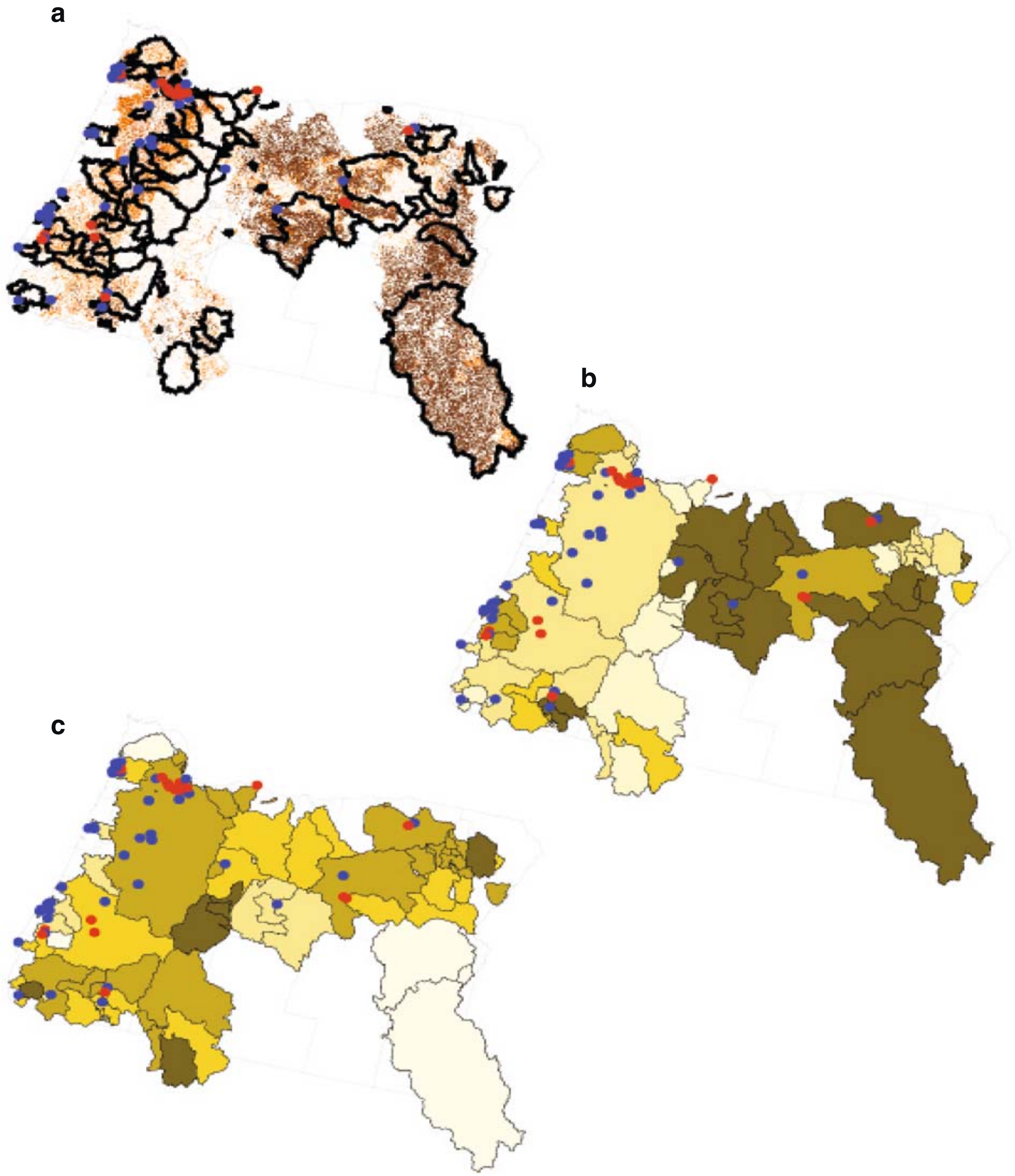

Fig. 2 Distribution of **a** GPI, non-nested watershed and locations of sample sites for the enterococci, *red* dots denote the exceeding level of that sample site in the enterococci (geometric mean $\geq$35 cfu/100 ml). *Blue dots* denote the enterococci geometric mean <35 cfu/100 ml). Mean of GPI (**b**) and variability of GPI (**c**) per watersheds (higher order nested watersheds) based on the five quantiles, *darker color* denote the higher value

For the standard multiple regression, we used forage related variables (WT's; Table 2), the landscape metrics derived from ATtILA (Ebert and Wade 2004), stream length, stream density and three measures for the GPI (mean, standard deviation and coefficient of variability) as predictors. Prior to regression analyses, collinearity between predictors was calculated, and a threshold of 0.80 was used as a criterion to select predictors for the regression model. Predictors retained in the model were further tested to determine whether predictor values calculated near a sample point had stronger correlation with the response variable than the predictor derived from the whole watershed. Pair-wise correlations of landscape metrics at each scale (linear buffers, proximal segments, whole watershed) with the response variables were examined. A predictor with highest correlation replaced the one that was derived from the whole watershed in the model. If this replacement increased predictive power of the model then that predictor was retained.

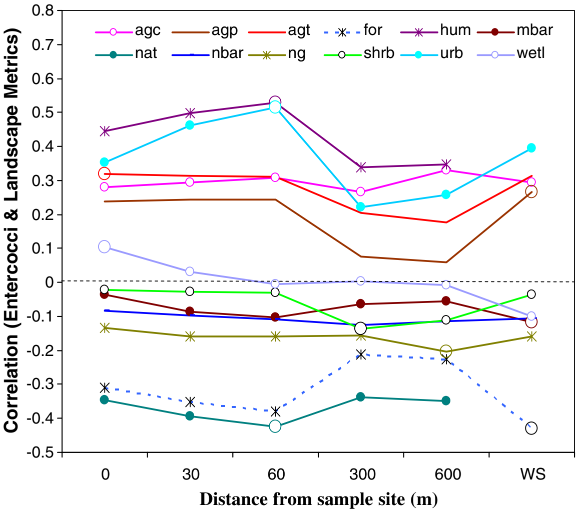

Fig. 3 Pair-wise correlation of enterococci with each of the landscape metrics ($n = 74$). The *large open circle* indicates the highest correlation value for the landscape metric. Landscape metrics entered the logistic regression model were; for, mbar, nbar, urb, wetl, agc600, agt300, for600, nat600, ng60, ng300, ng600, shrb300, hum60, urb60, urb600, mbar300, nbar60, nbar300, nbar600, wetl0, wetl600

Correlation of response with each of the predictors varied with scale in magnitude and direction. As an example and regardless to site condition (AT or BT), the pair-wise correlation between enterococci and each of the landscape metrics at different scales is given in Fig. 3. The close proximity of lines at a specific scale value indicates the high collinearity between landscape metrics at that scale (e.g. human and urban within 60 m of streams; Fig. 3). For each landscape metric, the strength of correlation with enterococci is scale dependent and not uniform over the scale, except for natural barren. Highest correlation with the enterococci may be at the watershed scale (pasture, manmade barren, forest), at 600 m (natural grasses), at 300 m (shrub land), at 60 m (human, urban, natural) or adjacent to stream (wetlands, total agriculture). While human, urban, agriculture and wetlands (within 60 m of stream) correlated positively with the enterococci concentration, and hence, may enhance enterococci concentration, remaining landscape metrics may have an opposite effect. Wetlands had a mixed behavior; Wetland's highest positive correlation with enterococci was adjacent to streams. Correlation decreased with distance to become the lowest (and negative) at the whole watershed scale. Inclusion of wetlands in the model will have an increasing or decreasing effect dependent on scale. An examination of Fig. 3 may help to mark predictors that are expected to be selected by the model (test your selection with that in logistic regression model in results).

Standard multiple regressions were performed on two groups of sites defined by the geometric mean of the enterococci concentration. Model residuals were tested for normality. Outliers from the analyses were identified and were not retained in the final model analysis. Logistic multiple regression with stepwise selection (Proc Logistic, SAS®) was used to determine the probability of a site exceeding the threshold in enterococci concentration "AT" being related to land and anthropogenic elements. Sample sites from the non-nested sites ($n = 74$; Fig. 2a) were used in building the predictive model. Predictors were landscape metrics, stream characteristics, forage availability related land cover and GPI measures. To test lack of fit for the model we used Hosmer–Lemeshow goodness-of-fit test where the higher probability value test statistics suggests that the fitted model is an adequate. Cross validation (Jackknife; leave one site out) was used in for prediction of the remaining watersheds (n = 411). When available,rainfall data and other ancillary information were used to empirically or conceptually explain anomalies or trends in surface water variables.

Results

Spatial and temporal characterization of enterococci

A total of 279 sample sites were analyzed for enterococci, 34% (94 sites) of these sites exceeded the standard geometric mean (35 cfu/100 ml; maximum = 6,672). These 94 sites were further investigated for the enterococci behavior with time (increasing/decreasing). One site (402767), located approximately 1 mile downstream of McKay Creek Dam, in the Umatilla basin, had one value exceeding the 35 cfu/100 ml of enterococci concentration, which occurred on July 5, 1990. Rainfall, irrigation runoff and/or releases from the reservoir could be possible causes for this high value. Major land use and land cover at this site is grazing and agriculture (ODEQ 2007). To further understand the relationship between weather patterns and increased enterococci concentrations, daily arithmetic mean enterococci concentrations were correlated to daily precipitation rates to locate synchronized high or low values. Six sites were identified. Three of these sites had weather stations (i.e., rain gages) within 2 km of the sampling site. The other three ranged in distance from 2 and 13 km. In all cases the amount of rainfall was below 4 mm per event, which is deemed insufficient to generate runoff to match the observed enterococci concentrations.

For those sites exceeding the geometric mean of 35 cfu/100 ml, monthly geometric means with time (1990–1994 for month of June through September) were further investigated for trend direction. An increase or decrease in the enterococci concentration at a site may have a link to the GPI. From the 93 sites exceeding the 35 cfu/100 ml, 32 had data for less than 5 months, all within 1 year. The remaining sites (61) were analyzed for trends, 11 sites exhibited a decreasing trend in enterococci concentration; three sites (402246, 402063, 402673) were equal to or just below the geometric mean of 35 cfu/100 ml. Sites 3827011 and 402104 decreased significantly ($p < 0.047$) with time. Fifty sites had a positive trend with 14 sites significantly ($p < 0.042$) increasing enterococci concentration with time.

Standard multiple regressions

Models accounted for 71% to 94% of variability in enterococci and nutrients. Neither grazing potential index measurements nor forage related variables appear to have a significant correlation with enterococci. Urban land cover adjacent to streams (urb0) appears to be the most important landscape metric in elevating surface water enterococci concentrations in “AT” sites ($\geq$35 cfu/100 ml; Table 4). A similar effect but with lower magnitude was observed in the density of streams in a watershed, natural grasses (ng) adjacent to streams and cropland (agc) within 600 m from the sample point. While manmade barren and shrub land cover decreased the enterococci concentrations in “BT” sites, agriculture in cropland within 300 m of the sample site increased the level of concentration in a significant contribution to explain 75% of the variability.

Forage related metrics (WT’s; Table 2) were not contributing factors to enterococci, but they appear to have a significant contribution to surface water nutrients. Forage related variables in forested land (WT4, WT5; Table 4) and all forest land cover in watershed had a significant role (partial R^2 = 15% to 86%) in decreasing the level of surface water nitrogen and phosphorous. Forage variables related to transitional/agriculture land (WT1), herbaceous wetland (WT7) and woody wetland (WT9) increased the concentration of nutrients in surface water.

Landscape metrics from different scales appear to have a combined effect on surface water nutrients. Landscape metrics derived from areas that are smaller than that of the whole watershed had higher contributions (except for DOP/”AT”).

Table 4 Partial regression coefficient and sign of each predictor per response model in "AT" and "BT" sites

Response	Predictors					Adj_R^2	SW_test
	WS_LS	Riparian	Circular segment	Forage	Stream		
Enterococci							
AT ($n = 17$)		**urb0 (+59)** ng0 (+3)	agc600 (+3)		Density (+17)	94	0.50
BT ($n = 45$)		mbar0 (−6) shrb60 (−6)	**agc300 (+60)** agt300 (+15)			83	0.13
NH_3NH_4_N (Ammonia_N)							
AT ($n = 17$)	mbar (−14)		**Shrb300 (−43)**	WT1 (+31)		85	0.15
BT ($n = 18$)		**agc0 (+74)**		WT4 (−15)		87	0.38
NO_2NO_3_N (Nitrate_N)							
AT ($n = 18$)	mbar (−27)	shrb60 (−10)	**Shrb300 (+44)**	WT5 (−4)	Length (+7)	89	0.71
BT ($n = 16$)			urb600 (+29) mbar600 (+22)	**WT7 (+36)**		83	0.52
TKN							
AT ($n = 18$)	agp (+26)		ng600 (+23) nat300 (+9)		Density (−22)	75	0.41
BT ($n = 16$)		**for0 (−86)** urb60 (+4)				89	1.00
TN							
AT ($n = 19$)		mbar60 (−21)	ng300 (−11)	**WT1 (+48)**		75	0.055
BT ($n = 16$)	mbar (−5)		urb600 (+7) Wetl300 (+2)	**WT4 (−73)** WT7 (+9)		94	0.45
TP							
AT ($n = 19$)		mbar60 (−20)	ng300 (−8)	**WT1 (+49)**		72	0.98
BT ($n = 18$)		**nat60 (−58)** wetl0 (+19)	agc600 (+7) nat600 (+4)			85	0.25
DOP							
AT ($n = 18$)	for (−47)		wetl600 (+32)			75	0.15
BT ($n = 18$)		**agc60 (+36)**		WT9 (+17)	Length (−23)	71	0.46

"AT" is geometric mean ≥35 cfu/100 ml; "BT" is geometric mean <35 cfu/100 ml. WS_LS is landscape metrics for the watershed. "Adj_R^2" is the adjusted value for the regression coefficient of determination. "SW_" is the normality Shapiro Wilk test of model residuals. Bolded text denotes the most contributing variable in the model. Description of forage variables is in Table 2 and description of landscape metrics is in Table 3

Watershed landscape metrics (manmade barren, pasture and forest; Table 4) contributed significantly to nitrogen but were not as frequent and strong as that from smaller areas (i.e. 0, 30, 60, 300, 600 m). The combined contribution of landscape metrics from smaller areas ranged from 1.2 to 3.1 times of that for the whole watershed, except for DOP/"AT" where the forest from the whole watershed was 1.5 times of that of the smaller areas.

Watershed pasture, crop, urban and wetland within 600 m of the sample site enhanced the concentration of nitrogen and phosphorous in surface water. Whereas, percent manmade barren, shrub lands, natural and natural grass had a mixed effect on nutrients. As expected, forest had a significant role in decreasing the concentration of DOP and TKN in "AT" and"BT" sites, respectively. The concentration of Nitrate_N, TKN, and TN in "BT" sites were all enhanced by urban land cover within 600 m of the sample site, but the influence was much stronger for Nitrate_N. Urban was not a contributing factor in "AT" sites. Total pasture in watershed increased TKN concentration in "AT" sites. In "BT" sites, TP, DOP, and Ammonia_N concentration were enhanced by higher amounts of percent of cropland within 600 m of the sample site, cropland in areas adjacent to streams (agc0) explained 74% of the variability in Ammonia_N.

Stream density and stream length contributed significantly and accounted for 7–23% in variability of the enterococci and nutrients. Stream density increased the concentration of enterococci but decreased the TKN concentration in “AT” sites. Stream length increased Ammonia_N in “AT” sites but decreased of DOP in “BT” sites.

Logistic multiple regression

The probability of having sites with enterococci concentrations with a geomean of more than 35 cfu/100 ml was dependent on landscape metrics from different scales; whole watersheds to areas within 60 m of a stream (Fig. 4). The model had a good fit to the data set with percent concordance of 95.2 ($P = 0.99$ for the Hosmer & Lemeshow goodness of fit) (Figs. 5 and 6). The proportion of all natural land cover within 600 m of the sample site decreased the probability of having a site with high concentration (AT) of enterococci. Conversely, urban cover within 60 m of a stream and manmade barren for the whole watershed enhanced the likelihood of having sites with enterococci levels higher than 35 cfu/100 ml (Table 5).

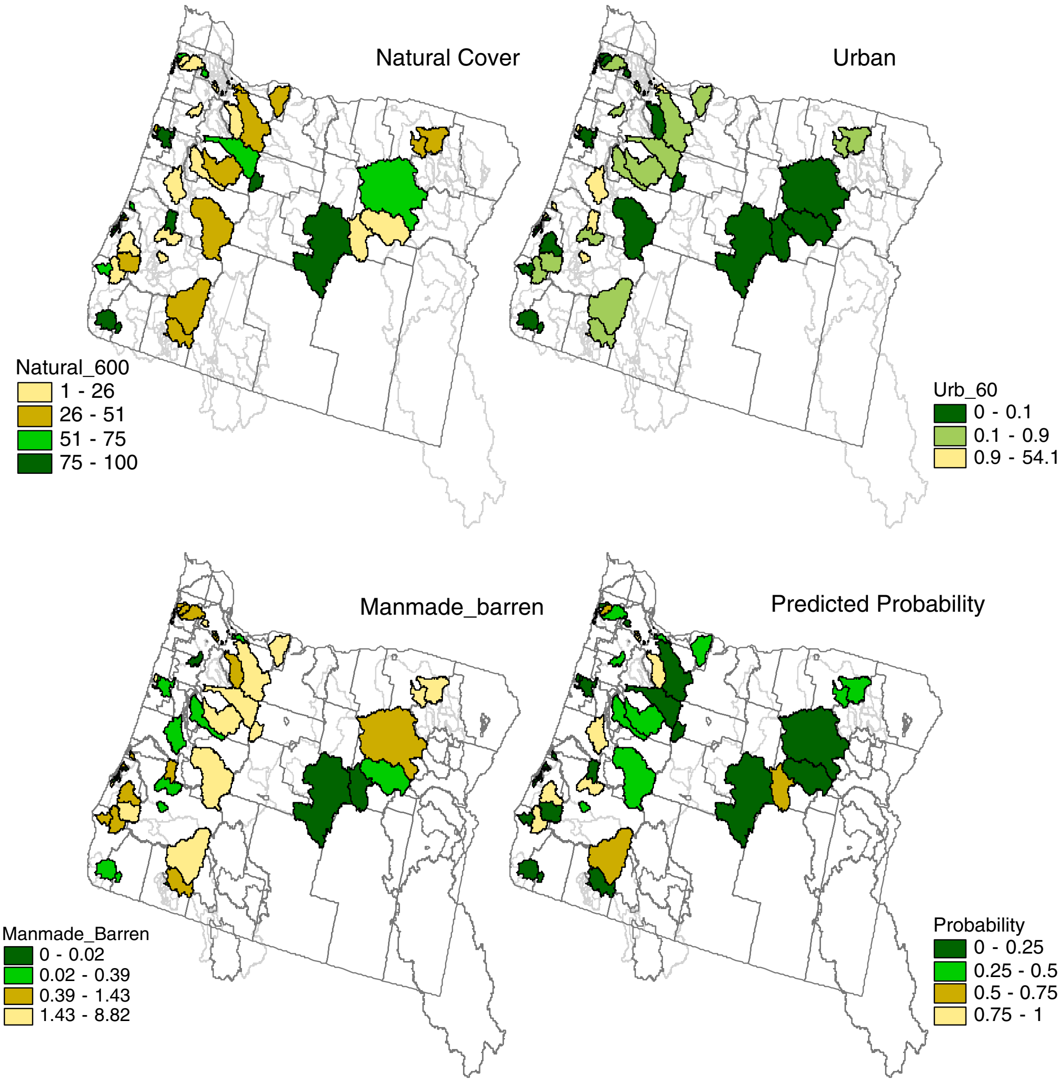

Fig. 4 The selected landscape metrics of natural (nat_600), urban (urb_60) and manmade barren (pmbar) land cover/land use by the logistic regression model to predict the probability (*lower right figure*) of enterococci exceeding the threshold “AT”

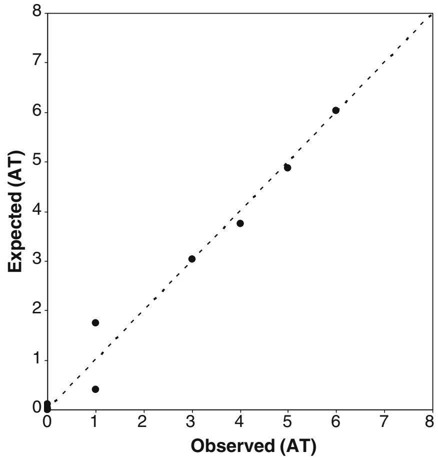

Fig. 5 Observed and predicted AT (enterococci geomean ≥35 cfu/100 m) sites by deciles of site condition in the training data set from Hosmer_Lemeshow goodness of fit ($\chi^2 = 1.78$, $P = 0.99$). *Dashed line* is 1:1 relationships

None of the grazing metrics or the forage availability related metrics (WT's) contributed to the variance in site condition (i.e. "AT").

The relative importance of each predictor measured by the standardized coefficient (Table 5) indicated that natural land cover within 600 m of sample site was the most important predictor affecting the condition of the site (i.e. AT or BT).

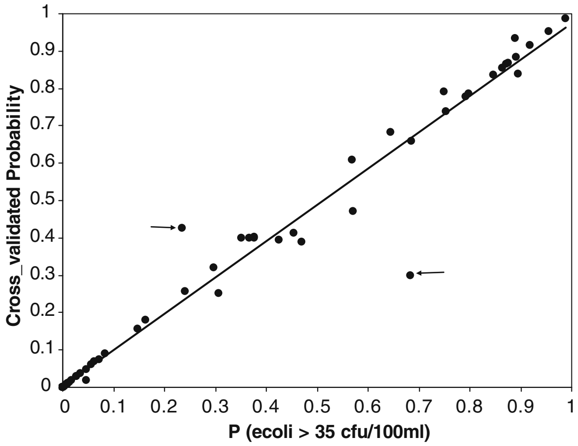

Fig. 6 Cross validation (Jackknife) and full data probabilities for the fitted model. The two points (*marked with arrows*) that are most distant from the 1:1 line from *right* to *left* are for sites (stn in STORET, EPA's Main Repository for Water Quality, biological and Physical data. http://www.epa.gov/STORET/): 412006 and 404171

The predicted odd ratios of "AT" for natural land cover was 0.876 times the odds of "BT". That is, the likelihood of "AT" decreased by 12% with a unit (1%) increased in percent of natural land cover within 600 m of a sample site (nat600; odds ratio, Table 5). The predicted odds of having "AT" sites increased by 73%, and 11% with a unit (1%) increased in percent of all manmade barren land use (mbar) and urban land (urb60) use within 60 m of a stream, respectively.

The predicted probability of having a site that exceeds the threshold for the remaining 411 watersheds (Fig. 7) was carried out using the above model. Watersheds with higher probability that exceed the threshold in enterococci were mostly located in the western side of the state (Fig. 7a). Because of high nesting in watershed structure, we presented only non-nested watersheds in Fig. 7a. When the nesting structure was ignored, higher ranking nested watersheds or larger watersheds took precedent showing the overall probability of exceeding the threshold in Fig. 7b.

Discussion

Information on forage availability, distance to water, slope and landform were included in the GPI calculations. An area with a high GPI was expected to be favored and utilized by livestock more than an area with a low GPI score. While scattered patches of land with high potential for grazing are found in the Willamette Valley, higher value patches are denser in eastern Oregon indicating an increased potential for grazing (Fig. 2). None of the GPI metrics appeared to be a significant factor in the enterococci and nutrient models. However, forage related variables (WT's) had a significant role in nutrient models. While evergreen and mixed forest (WT4 & WT5) decreased the nutrient concentration in surface water, herbaceous and woody wetlands increased the concentration on nutrient in surface water (Table 4). Livestock prefer stream and wetland riparian areas for their abundance of water, forage and shade (DelCurto et al. 2005). The presence and concentration of organic and inorganic nitrogen is an indicator of human and/or livestock input into surface water. High levels of ammonia (NH_3NH_4_N)

Table 5 Probability, standard estimate and odds ratio of predictors estimate from logistic regression ($n = 74$ sites; 20 "AT" sites and 54 "BT" sites)

Predictor	Estimate (β)	*P*	Standard estimate	Odds ratio	Variable description
Intercept	2.2604				
nat600	−0.1324	0.0003	−2.4596	0.876	Percent of all natural land use within 600 m circular segment of a sample site
urb60	0.1064	0.0224	0.6949	1.112	Percent of urban in 60 m stream buffer
mbar	0.5460	0.0351	0.5319	1.726	Percent of manmade bare land in watershed

at "BT" sites were mainly associated with the presences of cropland by the stream (partial $R^2 = 74\%$). Pasture land use patterns, natural grasses and all natural land cover within 600 m of a sample site played a major role in nitrogen concentrations, predominantly TKN, in "AT" sites. Stream density and all forest in designated "AT" and "BT" watersheds, respectively, were factors in decreasing of TKN concentrations in surface water. At "BT" sites, however, the

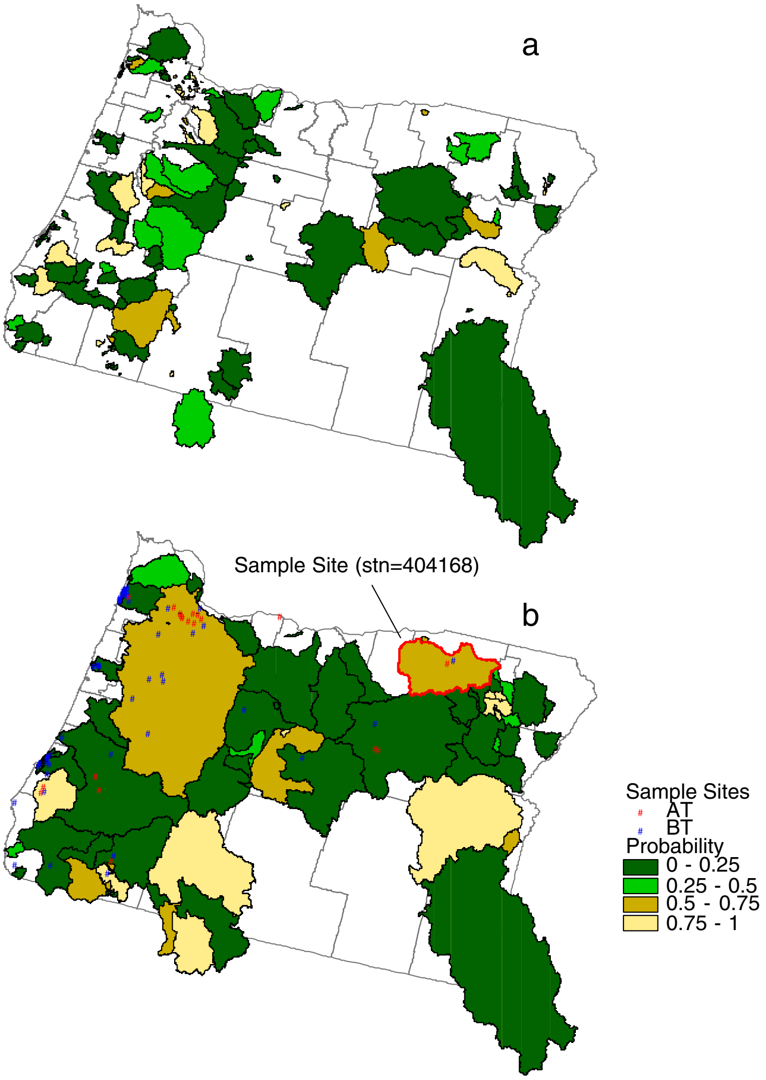

Fig. 7 Predicted probability of a site exceeding the threshold concentration of enterococci for **a** non-nested watersheds, and **b** all watersheds, higher order nested watershed appears in the figure. The whole Willamette Valley is within the area of exceeding the threshold

Nitrate_N concentrations increased as the percentages of herbaceous wetlands, urban and manmade barren within 600 m of sample site increased. As the ecological function of urban and barren land covers decreases, overland flow patterns become altered and associated plant communities are lost. Excess nutrients load interactions with soil and plants are reduced and this may results in a net increase of nitrates and bacteria reaching surface waters. Herbaceous wetlands can be sinks or sources of nitrogen and are often cited as sinks for nitrate via the process of denitrification. When acting as sources, wetlands typically exporting nitrogen as ammonia or dissolved organic nitrogen. It is possible that the relationships established with Nitrate_N resulted from denitrification of exported ammonia but further study would be needed.

Predictors contributed significantly to each response seen in Table 4. These responses reflect the composition and dominance of forage related variables or landscape metrics related to urban, wetland or other land use pattern. Urban land cover, distribution and concentration, was highly correlated with elevated Nitrate_N, TKN and TN concentrations. Possible sources were fertilization of lawn, leaking septic systems, waste water discharge, etc. Wetland land cover was correlated to an enhanced concentration of Nitrate_N, TN, TP and DOP in "BT" sites. Grazing pattern, perhaps, is one reason for this positive correlation. Grazing pattern is affected by season, climate and type and age of livestock (VanWagoner et al. 2006; DelCurto et al. 2005; Loucougaray et al. 2004). Different types of livestock will have different grazing patterns (VanWagoner et al. 2006; Loucougaray et al. 2004). For example, older (DelCurto et al. 2005), taller, heavier, and different varieties (VanWagoner et al. 2006) of cows will venture further away from a water source. In early summer months livestock spread and forage uniformly. Late summer livestock prefer shady areas, closer to water and riparian forage (DelCurto et al. 2005). Hence, the utilization of riparian areas in the summer months is higher. Bacterialogical and nutrient water quality data within the study area collected over the summer season (June–September, 1990–1994) showed the relationship of loads in the summer rather than in the winter. GPI metrics were not contributing factors in variability of any of the nitrogen forms, but forage related predictors (wetland, pastures) that linked to grazing potential appeared to have a significant role. For example, the isolated wetlands adjacent to the Upper Klamath Lake have been used for cultivated crop and cattle grazing since the turn of the twentieth century. The input from agriculture as well as the oxidation process of organic matter elevated the level of inorganic nitrogen in soil solution (Snyder and Morace 1997). High nutrient concentrations in the lake resulted in eutrophication and subsequent blue-green algal blooms.

The probability of having sites with enterococci concentrations ≥35 cfu/100 ml are affected by landscape metrics that are either in riparian zones or within circular segments of 600 m from the sampling site. In a site (stn = 404168), an increase of 2% only in proportion of the natural land cover within 600 m of sample site (Table 5) decreased the probability of exceeding the threshold, hence improving the condition of a site. Other measure of contribution is the odds ratio; natural land cover within 600 m of sample site reduced the likelihood of having high enterococci concentrations (≥35-cfu/100 ml) by 22%, a relationship opposite to that of urban within 60 m of stream and manmade barren in the whole watershed. Closer proximity of a pollution source to the sample site signaled a proportional effect on surface water biota. Confirming these results on surface water biota, Wickham et al. (2006) reported that in Maryland, bacterial contamination of stream waters is highest in small watersheds dominated by urban landuse close to streams.

Condition of some sites was reversed after probability prediction using logistic regression. For example, a condition of a site at the Umatilla River at the Westland Road, west of Hermiston City (stn = 404168; Fig. 7b; red polygon) was "AT" but changed after prediction to "BT". Close examination to this watershed indicates that this sample site is within areas where the confined animal feeding operations (CAFOs) and leaky septic systems are common source of fecal coliform contaminations, which worsens in summer months when surface water flow rates are low (ODEQ 2007). The proportion of the landscape in this watershed were low in wetlands (0.01%)

and urban (0.69%) within 60 m of stream, medium to low in natural land cover (21%) but high in human use within 600 m (79%). Viewing the landscape in Google Earth®, the distribution and spatial intensity of agriculture, urbanization and city services are more intense around the sample site. Contamination caused by all human use of the surrounding area is more likely the cause of a high concentration of enterococci. Human use and natural cover have a high inverse correlation ($r > 0.8$).

Conclusion

Individual components of forage related variables in the grazing potential index, GPI, proved to be useful. Forage related variables contributed significantly to nutrients but not to enterococci concentration. The consistent contribution of forage related variables in increasing or decreasing nutrient concentrations in surface water makes them valuable for future consideration in grazing analyses. The grazing index, GPI, appears to be a non significant contributor that was not retained by any model. To be useful, further modifications of GPI are needed.

The anthropogenic effect was evident in elevated enterococci concentrations in streams, but the strength of that effect depends on the surrounding landscape at different scales. Urban adjacent to streams and cropland within 300 m of sample sites increased the concentration of enterococci in “AT” and “BT” areas, respectively. In sites above the threshold for enterococci; urban was the primary factor in elevating concentration of enterococci in surface water by as much as 20 times as agriculture. Whereas, in below level threshold sites, agriculture was the dominant factor in higher enterococci concentrations. Enterococci concentration appears to be readily affected by areas within proximity of sample sites and not by whole watersheds.

While only land cover at local scale influenced enterococci concentrations, nutrient concentrations were dependent on the landscape metrics that derived from multiple scales; watershed scale, 100’s m to 0 m by streams. Nitrogen and phosphorous fertilizers, and manure are added to soil for lawn (homes, golf courses, etc), crops, pastures, and in most dairy farms. Plants and microbes utilize the dissolved nitrogen and phosphorous in soil solutions. Excess nitrogen is leached from soil into subsurface waters in the form of nitrate, transported via soil particles (sediments) to streams in runoff, or transformed to the atmosphere via volatilization or denitrification. Phosphorous is often retained by soil particles and is less likely to be leached. Therefore, contribution of phosphorous to streams is due to runoff where sediments are able to reach streams. From an experimental plot study, Minshall et al. (1969) reported that the loss of phosphorus (P) in base flow was only one-tenth of that of surface runoff from plots and watersheds. Measured concentration of phosphorous was as high as 200 ppb in the runoff water from fertile topsoil, whereas this value reduced to 15 ppb as the water moved downstream (Kunishi et al. 1972). This is due to soil particle retention of P (Kunishi et al. 1972). Both N and P are linked to eutrophication of surface water. Concentrations of P within range of 0.01–0.3 ppm are associated with algal growth. Nitrogen sources include not only fertilizers, but can be from degraded organic matter, animal waste, septic tanks, sewage and industrial effluent. Contamination from these sources could be on a local scale. Streams closer to these sources are more likely to be contaminated than streams further away. Though wetlands are typically considered as sinks of nutrients in surface water, our study showed the opposite. Wetlands, natural grasses, and all natural land cover are closer to streams and sample sites contributed to the higher levels of nutrients, it is an indication of livestock utilization.

The probability of having sites exceeding the threshold of enterococci concentrations was much lower in areas of natural land cover and much higher in areas with higher urban land use within 60 m of stream. An increase of natural land cover while holding the proportion of watersheds manmade barren and urban within 60 m stream buffer constant will lower the probability of exceeding the threshold of enterococci concentration. A two unit increase in natural land cover decreased the probability that changed the condition of a site from “AT” to “BT”. While controlling and meeting human need, maintaining and preserving natural land cover by certain level may help in providing healthier surface water quality.

Proper management for each source of contamination may limit and lower the contamination level in watersheds. Removal of microbiological contamination from stream water runoff by establishing certain structural designs is reported by Kurz (1998). Hagedorn et al. (1999) have shown that simple fencing of pastures to exclude cattle from streams or creating narrow water gaps (Sherman Swanson, University of Nevada, Reno, pers. Comm.) can result in significant reduction in fecal coliform concentration in a rural watershed. Non-structural practices include proper waste management at dairies and poultry farms, livestock management (exclusion) near water bodies and streams, good housekeeping and facilities management at wastewater treatment plants, and proper site planning and management of on-site wastewater treatment systems. Both structural and non-structural systems should be evaluated for their potentials to reduce fecal contamination and accurate assessment of the source and nature of the contamination is identified within a given watershed.

Acknowledgements We are very grateful for the inputs of Toney Selle and Dr. Jay Christensen (US EPA). The U.S. Environmental Protection Agency, through its Office of Research and Development, funded the research described herein. Although this work was reviewed by EPA and approved for publication, it may not necessarily reflect official Agency policy. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

References

Andrew, M. H., & Lange, R. T. (1986). Development of a new biosphere in arid chenopod shrubland grazed by sheep 1. Changes to the soil surface. *Australian Journal of Ecology, 11*, 395–409.

APHA (1998). *Standard methods for the examination of water and waste water*. Washington, DC: American Public Health Association, American Water Works Association and Water Environmental Federation.

Belsky, A. J., Matzke, A., & Uselman, S. (1999). Survey of livestock influences on stream and riparian ecosystems in the western United States. *Journal of Soil and Water Conservation 54*(1), 419–431.

Belsky, A. J., Matzke, A., & Uselman, S. (2002). What the river once was: Livestock destruction of western waters and wetlands. In G. Wuerthner, & M. Matteson (Eds.), *Welfare ranching: The subsidized destruction of the American West*. Washington, D.C.: Island Press.

Betteridge, K., Andrews, W. G. K., & Sedcole, J. R. (1986). Intake and excretion of nitrogen, potassium and phosphorus by grazing steers. *Journal of Agricultural Science, 106*, 393–404.

Cook, C. W. (1966). Factors affecting utilization of mountain slopes by cattle. *Journal of Range Management, 19*, 200–204.

DelCurto, T., Porath, M., Parsons, C. T., & Morrison, J. A. (2005). Management strategies for sustainable beef cattle grazing on forested rangeland in the Pacific Northwest. *Rangeland Ecology and Management, 58*(2), 119–127.

deSoyza, A. G., Whitford, W. G., & Herrick, J. E. (1997). Sensitivity testing of indicators of ecosystem health. *Ecosystem Health, 3*, 44–53.

Ebert, D. W., & Wade, T. G. (2004). *Analytical tools interface for landscape assessments (ATtiLA): User manual. EPA/600/R-04/083*. Washington, DC, USA: US Environmental Protection Agency.

Fleischner, T. L. (1994). Ecological costs of livestock grazing in western North America. *Conservation Biology, 8*, 629–644.

Fusco, M., Holechek, J., Tembo, A., Daniel, A., & Cardenas, M. (1995). Grazing influence on watering point vegetation in the Chihuahuan desert. *Journal of Range Management, 48*, 32–38.

Gillen, R. L., Krueger, W. C., & Miller, R. F. (1984). Cattle distribution on mountain rangeland in northeastern Oregon. *Journal of Range Management, 37*, 549–553.

Graetz, R. D., & Ludwig, J. K. (1978). A method for the analysis of biosphere data applicable to range assessment. *Australian Rangeland Journal, 2*, 126–136.

Hagedorn, C., Robinson, S. L., Flitz, J. R., Grubbs, S. M., Angier, T. A., & Reneeau, R. B. (1999). Using antibiotic resistance pattern in the fecal streptococci to determine sources of fecal pollution in a rural Virginia Watershed. *Applied and Environmental Microbiology, 65*, 5522–5531.

Jarvie, H. P., Neal, C., Jürgens, M. D., Sutton, E. J., Neal, M., Wickham, H. D., et al. (2006). Within-river nutrient processing in Chalk streams: The Pang and Lambourn, UK. *Journal of Hydrology, 330*(1–2), 101–125, 30 October 2006.

Jones, K. B., Heggem, D. T., Wade, T. G., Neale, A. C., Nash, M. S., Mehaffey, M. H., et al. (2000). Assessing landscape condition relative to water resources in the Western United States: Strategic approach. *Environmental Monitoring and Assessment, 64*, 227–245.

Kauffman, J. B., & Krueger, W. C. (1984). Livestock impacts on riparian ecosystems and streamside management implications: A review. *Journal of Range Management, 37*, 430–438.

Kunishi, H. M., Taylor, A. W., Heald, W. R., Gburek, W. J., & Weaver, R. N. (1972). Phosphate movement from an agricultural watershed during two rainfall periods. *Journal of Agricultural Food and Chemistry, 20*, 900–905.

Kurz, R. C. (1998). A comparison of rapid sand filtration, alum treatment, and wet detention for the removal of bacteria, viruses, and protozoan surrogate from storm water. Technical report. Southwest Florida Water Management District.

Lange, R. T. (1969). The piosphere: Sheep track and dung patterns. *Journal of Range Management, 48*, 396– 400.

Loucougaray, G., Bonis, A., & Bouzillé, J. B. (2004). Effects of grazing by horses and/or cattle on the diversity of coastal grasslands in western France. *Biological Conservation, 116*(1), 59–71, March.

MacLusky, D. S. (1960). Some estimates of the areas of pasture fouled by excreta of dairy cows. *Journal of the British Grassland Society, 15*, 181–188.

Marsh, R., & Campling, R. C. (1970). Fouling of pastures by dung. *Herbage Abstracts, 40*(2), 123–130.

Minshall, N., Starr, N. M., & Witzel, S. A. (1969). Plant nutrients in base flow of streams in southwestern Wisconsin. *Water Resources Research, 5*, 706–713.

Mitchell, C., Brodie, J., & White, I. (2004). Sediments, nutrients and pesticide residues in event flow conditions in streams of the Mackay Whitsunday Region, Australia. *Marine Pollution Bulletin, 51*(1–4), 23–36.

Mueggler, W. F. (1965). Cattle distribution on steep slopes. *Journal of Range Management, 18*, 255–257.

Nash, M. S., Jackson, E., & Whitford, W. G. (2003). Soil microtopography on grazing gradients in Chihuahuan Desert Grasslands. *Journal of Arid Environment, 55*, 181–192.

Nash, M. S., Whitford, W. G., deSoyza, A., & Van Zee, J. (1999). Livestock activity and Chihuahuan desert annual plant communities: Boundary analysis of disturbance gradients. *Ecological Applications, 9*, 814–823.

OCS (2008). Oregon climate service: Climate data. retrieved January 9, 2008 from http://www.ocs.orst.edu/.

ODEQ (2007). Oregon department of environment and quality: Oregon water quality Index for the Umatilla Basin, water years 1986–1995, retrieved August 30, 2007 from http://www.deq.state.or.us/lab/wqm/wqindex/umatilla3.htm.

Oenema, O., van Liere, L., & Schoumans, O. (2005). Effects of lowering nitrogen and phosphorus surpluses in agriculture on the quality of groundwater and surface water in the Netherlands. *Journal of Hydrology, 304*(1–4), 289–301, 10 March.

Olson, R. A., Army, T. J., Hanway, J. J., & Klimer, V. J. (1971). *Fertilizer technology & use*, 2nd edn. Madison, Wisconsin, USA: Soil Science Society of America.

Pinchak, W. E., Smith, M. A., Hart, R. H., & Waggoner, J. W. Jr. (1991). Beef cattle distribution patterns on foothill range. *Journal of Range Management, 44*, 267–275.

Roath, L. R., & Krueger, W. C. (1982). Cattle grazing and behavior on a forested range. *Journal of Range Management, 35*, 332–338.

Snyder, D. T., & Morace, J. L. (1997). Nitrogen and phosphorous loading from drained wetlands adjacent to Upper Klamath and Agency Lakes, Oregon. Water-Resources Investigations Report 97-4097. Portland, OR: U.S. Department of Interior, Geological Survey.

Taylor, A. W., Edwards, W. M., & Simposon, E. C. (1971). Nutrients in streams draining woodland and farmland near Coshocton, Ohio. *Water Resources Research, 7*, 81–89.

USEPA (2000). Beaches environmental assessment and coastal health act of 2000 (p. 9). Public Law 106–284—Oct. 10, 2000.

US Geological Survey (2008a). National hydrography dataset home page. URL: http://nhd.usgs.gov/.

US Geological Survey (2008b). USGS mapping information: Geographic names information system (GNIS). URL: http://geonames.usgs.gov/domestic/.

US Geological Survey (2008c). Federal lands of the United States. URL: http://www.nationalatlas.gov/mld/fedlanp.html.

VanWagoner, H. C., Bailey, D. W., Kress, D. D., Anderson, D. C., & Davis, K. C. (2006). Differences among beef sire breeds and relationships between terrain use and performance when daughters graze foothill rangelands as cows. *Applied Animal Behavior Science, 97* (2–4), 105–121, May.

Vogelmann, J. E., Howard, S. M., Yang, L., Larson, C. R., Wylie, B. K., & Van Driel, N. (2001). Completion of the 1990s national land cover data set for the conterminous United States from landsat thematic mapper data and ancillary data sources. *Photogrammetric Engineering and Remote Sensing, 67*, 650–662.

Wade, T. G., Schultz, B. W., & Wickham, J. D. (1998). Modeling the potential spatial distribution of beef cattle grazing using a geographic information system. *Journal of Arid Environments, 38*, 325–334.

Wickham, J. D., Nash, M. S., Wade, T. G., & Currey, L. (2006). Statewide empirical modeling of bacterial contamination of surface waters. *Journal of the American Water Resources Association (JAWRA), 42*, 1–9.