PROBABILISTIC MODELLING OF WETLAND CONDITION

Report to the
Water Research Commission

by

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EXECUTIVE SUMMARY

Projections on growing water quantity and quality stress within South Africa are becoming more frequent. One approach in mitigating risks of water supply failure is to safeguard wetlands, with the corollary that degraded wetlands fail to provide ecosystem services, including baseflow recharge and maintenance. Wetland assessments continue to be undertaken, although still at the scale of individual wetlands, and ultimately a regional assessment is required. Costs and logistics prevent conservation managers from assessing the condition of all known wetlands regionally or nationally. The growing levels of catchment degradation occurring nationally necessitate a higher-level approach be taken to assess wetland conditions per wetland type, to begin prioritizing conservation actions at a regional and national level. Given constraints in regional data collection, models which describe relationships between wetland conditions and remotely sensed land use, become useful in achieving regional planning and evaluation of ecosystem health. Such models with region-wide predictive ability are necessary to inform management, planning and policy decisions. The aim of this research was to develop models for statistically predicting the ecological status and probability of degradation per wetland type, based on landscape-scale drivers.

Field and existing data for a total of 463 seep, floodplain and valley bottom (channelled and unchannelled) hydrogeomorphic (HGM) units was used to develop predictive models of HGM unit condition. Data on land cover, infrastructure and catchment physiography covering 21 potential explanatory variables, were linked to individual HGM units. Multiple linear regression models were developed to estimate A-F Present Ecological Status scores, while multiple logistic regression models were developed to estimate the probability of degradation per HGM type. Models were developed at tertiary and quaternary catchment scales, and fine scale using a 1000 m buffer radius around each HGM unit centroid.

Statistical models were successfully developed to estimate both condition and probability of degradation for all four HGM types. Different landscape predictors were linked to the condition status of the different HGM types. Percent natural vegetation, road density, number and area of dams, and altitude, together with anthropogenic land use (cultivation and plantation area) were important predictors of wetland condition. Average wetland conditions per HGM type per quaternary catchment were successfully mapped to provide images on spatial patterns of degradation within KwaZulu-Natal.

The models show promise, and are useful at a regional planning level to identify priority catchments where field assessments of wetland condition would be required. Models are preliminary and should be refined using more data, and covering a wider area. Development of such models is in line with current international wetland research. The models in no way make existing initiatives in South African wetland planning redundant, and rather complement these. There is potential to refine and extend the models for application at a national level.

INTRODUCTION

Wetland status and catchment context

Projections on growing water quantity and quality stress within South Africa are becoming more frequent. Such projections implicitly assume the compounding issue of ongoing catchment degradation, a factor which is intimately linked to national water supply assurances. One approach in mitigating risks of water supply failure is to safeguard wetlands, with the corollary that degraded wetlands fail to provide ecosystem services, including baseflow recharge and maintenance. Kotze (2004) describes a wetland as 'lost' if it has been degraded to a point where it has lost most of its wetland properties and ecosystem services. Already over twenty years ago, Begg (1988), recorded that 33% of wetlands had been lost in the Mfolozi catchment of the province of KwaZulu-Natal. Ten years later, and already more than a decade old, the situation of wetland loss was even more dire, with figures of 45-65% recorded in catchments in KwaZulu-Natal (Whyte and Shepherd 1999). Wetland assessments continue to be undertaken (Macfarlane et al. 2011), although still at the scale of individual wetlands, and ultimately a regional assessment is required.

Wetland classification and condition

Two steps towards regional assessments of wetland condition are to use a suitable classification, and then to assess condition per type. The first step in estimating levels of degradation is to begin with a classification of wetland types. Currently, there are usable hierarchical wetland classifications available to conservation planners. A wetland classification suitable to the project objectives is central to their evaluation, management and conservation, as well as their prioritization for protection and rehabilitation (Ewart-Smith et al. 2006; Lindemayer et al. 2008). The need for a classification to group wetlands by abiotic features that drove their functionality was recognised in South Africa, with biotic features further down the hierarchy providing more detailed levels of the classification (Ewart-Smith et al. 2006). Since landform and hydrology are the two fundamental features which determine the existence of all wetlands, it is logical to use a hydrogeomorphic classification as a robust system, as also adopted in New Zealand (Clarkson et al. 2004).

The current classification system (SANBI 2009) is strongly aligned to the earlier classification of Ewart-Smith et al. (2006), where functional units are referred to as hydrogeomorphic (HGM) units. Furthermore, Kotze et al. (2006) found that a wetland classification by vegetation type, and linking this to potential spatial predictors, was unclear. The adoption of existing classification systems, rather than attempting to "reinvent the wheel" is perceived as an important consideration, especially when existing methods and methods currently being developed are aligned with HGM units, including:

- Assessment of present ecological state (DWAF 2007; MacFarlane et al. 2008; Kotze et al. 2011);
- Assessment of ecological importance and sensitivity (Kotze et al. 2007); and

- Buffer Zone Determination – appropriate buffer zones should be delineated depending on the associated surrounding land use.

The second step is to assign condition to HGM type. One benefit of linking wetland condition to type is that different types of wetlands perform different types of functions (Tiner 2003), and allows for setting differential conservation targets based on wetland services by type. For example, in a catchment where water quality is a priority, the conservation of the HGM units considered to contribute significantly towards reducing sediment, toxicants and nutrients would be prioritised for protection.

Wetland health is defined as a measure of the deviation of wetland structure and function from its natural reference condition (Kotze et al. 2006; Ellery et al. 2009). Wetland condition systems adopted by numerous state agencies all share the approach of using scoring metrics to rate condition, and apply the approach using a chosen classification system (for example, US EPA 2003; Lane et al. 2003; Clarkson et al. 2004; DWAF 2007; Macfarlane et al. 2008; Kotze et al. 2011). Condition assessments are typically multi-metric, with some only applying only biotic metrics (diatoms, algae, macrophytes, and aquatic macroinvertebrates – see US EPA 2003; Lane et al. 2003), and other using additional metrics (hydrological integrity and ecosystem intactness or geomorphology – Clarkson et al. 2004; DWAF 2007; Macfarlane et al. 2008; Kotze et al. 2011). The various indices of biotic integrity are based on species composition, diversity and functional organisation comparable to natural habitats (Stoddard et al. 2006).
Historically, the assessment of wetland ecosystems in South Africa were descriptive studies based on expert opinion, rather than on formal assessment methods. The recent development of formal multi-metric assessment techniques has provided wetland practitioners with frameworks to assess present ecological state and the delivery of ecosystem goods and services. These frameworks included:

- WET-Health (MacFarlane et al. 2008; Kotze et al. 2011) (with links to WET-EcoServices (Kotze et al. 2007)
- Wetland Index of Habitat Integrity (DWAF, 2007)

The first, the WET-Health system (Macfarlane et al. 2008; Kotze et al. 2011), uses a ten-point scoring system to assess the integrity of the hydrology, geomorphology and vegetation of individual wetlands, where a value of 0 is for pristine condition, and a value of 10 is for critically altered. While metrics for each individual component are usually calculated, it is typical that such scores are combined into a final, averaged condition score. Such practices of generating a single measure of wetland condition for HGM units are consistent with other studies (Jacobs et al. 2010), where a wetland condition qualitative score is a proxy for biogeochemical process and function (Jordan et al. 2007). Assessments are based on extent, intensity and magnitude of impacts. This condition assessment tool is flexible in that it can be applied at different levels of assessment intensity. For example, Kotze et al. (2006) used a Level I condition assessment in a stratified random sampling approach to assess the health of 104 individual wetlands in six catchments in the Drakensberg region of South Africa, representing wetlands from a range of altitudes and geological provinces, and hydro-geomorphic types. In this study, the hydrological regime was identified as the critical physical determinant of wetland composition and structure, and that a change in catchment land use translated into a change in water quantity, resulting in a change in the wetland.

Ellery et al. (2009) used the WET-Health scoring system at a landscape scale to assess the cumulative impacts of human activities on wetlands. At this scale, an important driver was land cover change, which assumed that wetland structure and function are fundamentally affected by the hydrological regime, and land use changes in a catchment affect the timing and amount of runoff flow into a wetland, and within a wetland, land use changes impact on the pattern and residency time of flow in a wetland.

A second wetland condition assessment tool is the Wetland Index of Habitat Integrity (Wetland-IHI) currently used by the national Department of Water Affairs (DWA) (DWAF 2007). This is acknowledged as a rapid assessment approach, and only applies to meandering floodplain and channelled valley bottom wetland types. The assessment is based on driving processes (hydrology, geomorphology and water quality) and modifiers (changes in land use affecting changes in vegetation), which are scored to provide a final PES rating using the DWA's A-F ecological categories. A mean value is calculated from metrics of hydrology, water quality, geomorphic and biotic, where categories E-F fall outside an acceptable range (Dickens et al. 2003).

In the provincial wetland community of practice, *ad hoc* feedback from various wetland specialists suggests that the Wetland Management Series documentation (WET-Health and WET-Ecoservices) is currently more widely used amongst wetland specialists within the province than the Wetland IHI technique (Cowden 2010). This is likely to be linked to two factors:

- the Wetland IHI technique is limited to floodplain and channelled valley bottom wetland systems, and
- a large amount of the assessments undertaken by wetland specialists are used to inform environmental impact assessments or water use license applications, where detailed rather than rapid assessments are perceived as being necessary.

Towards landscape assessments

Costs and logistics prevent conservation managers from assessing the condition of all known wetlands regionally or nationally. Given the growing levels of catchment degradation occurring nationally, it is critical that a higher-level approach be taken to assess wetland conditions per wetland type, to begin prioritizing conservation actions at a regional and national level.

Over twenty years ago, the need to define conservation priorities for aquatic systems as a necessary first step, followed by the need to communicate these to conservation managers, was already recognised (O'Keeffe et al. 1987). This has not been achieved because of the limitations described above, and landscape-scale tools show promise in assisting planners working at a provincial scale to assess wetland condition. Thus, for example, within the conservation planning sector, conservation actions for identified priority wetlands cannot be refined because there is very little knowledge on their level of degradation, even though the need to incorporate conservation and management of wetlands into catchment management planning has been recognised for almost a decade (Dickens et al. 2003). To inform conservation planning, studies that attempt to quantify the loss of wetland habitat should include descriptions of the percentage loss, the types of wetlands lost, the major factors contributing towards the degradation, and physiographic and geological features of the areas identified as having significant levels of loss (Kotze et al. 1995).

The amount of natural vegetation at catchment scale has been found to be a good predictor of river habitat integrity (Amis et al. 2006). Conversely, streams in agricultural catchments usually remain in good condition until the extent of agriculture exceeds 30-50% (Allan 2004). Similarly, for every 10% of altered catchment land use, a correlative 6% loss in freshwater diversity was noted, as a linear relationship (Weitjers et al. 2009). The cumulative impact of small dams within a catchment has been shown to impact on water quality and quantity (Mantel et al. 2010).

Given such relationships between catchment condition and system response, various studies have made use of such catchment scale relationships to estimate ecosystem health. Stein et al. (2002) developed indices to calculate river and catchment disturbance, which included settlement, land use and infrastructure (roads and railways and a simple distance decay function). For other studies, used or suggested metrics include percentage urban area, population and road densities, and land cover from selected categories including agriculture, forestry and natural vegetation, which have all been proposed as affecting system condition (O'Keeffe et al. 1987; Lopez 2006). Such metrics can be complemented with metrics describing physical catchment characteristics, which include basin length and drainage shape (Frimpong et al. 2005).

Wetland condition modelling

Given constraints in regional data collection, and relationships between condition and catchment-scale predictors, models become useful in achieving regional planning and evaluation of ecosystem health (Frimpong et al. 2005). Such models with region-wide predictive ability are necessary to inform management, planning and policy decisions (Gutzwiller and Flather 2011). A modelling approach begins to address the problem of a large focus of data collection at individual wetland scale, with decisions taken at catchment scale that don't depend on site-specific assessments. Whigham et al. (2007) demonstrated that site-specific approaches can be successfully applied at catchment scale, provided that sampling does not have geographic bias.

Models for linking condition to catchment-scale predictors provide the means to spatially map wetland trends across regions, and often fall within the multiple regression family of models. Multiple linear regression models use quantitative data, and facilitate the development of a correlative relationship (which may or may not be causal) between wetland condition and various causative landscape variables. In such a model, wetland condition is assumed to respond linearly to catchment variables, such that condition could be predicted from a scale of 0 (total degradation) to 100% (pristine condition). Multiple regression analyses have previously been to model stream health from spatial surrogates. In these studies, the extent of agricultural land at the sub-catchment scale (and land use in general) was the best single predictor of stream conditions (Allan 2004; Allan et al. 1997). Weller et al. (2007) used multiple regression models to estimate wetland condition using landscape indicators. In all instances, the models do not assign an ecological threshold to a particular condition, and the decision of whether to classify a wetland as degraded or not is left to the model user.

Probability models are developed using multiple logistic regression models using binary data as a response variable, but use explanatory variables in the same way as multiple linear regression models (McConway et al. 1999). A logit function is used to calculate the probability of wetland type *x* being degraded; using such a modelling approach facilitates the calculation of odds ratios, such that it is possible to make statements like "wetland type *x* is *n*-times more likely to be degraded than wetland type *y*". The key difference between this model type and multiple linear regressions is that in this case the modeller chooses an ecological threshold which classifies a wetland as degraded (value = 0) or not (value = 1), and transforms the data from quantitative to qualitative. While it may be argued that there is some data loss in this process, the predictive power of assigning odds ratios provides a powerful predictive tool for prioritizing one wetland type above another for conservation action. For example, Royle et al. (2002) developed a probability model to predict whether wetland basins contained water or not, based on landscape predictors. Such a model provides a statistical framework for directing management and conservation activities.

The aim of this research was to develop models for statistically predicting the ecological status and probability of degradation per wetland type, based on landscape-scale drivers. Thus, using the current wetland type classification, together with a sample of field-assessed wetland types, it will be possible to model wetland condition on a landscape scale using suitable approaches, thereby filling this large gap in current freshwater conservation plans.

METHODS

Study area

The study area was defined in terms of natural rather than administrative boundaries. The primary catchments which cover KwaZulu-Natal were used to define the broad study area, while the study area was refined according to the quaternary catchments which fall entirely within the province (Figure 1). The final study area reflected the need to calculate percentage land cover per catchment for which the entire catchment had data.

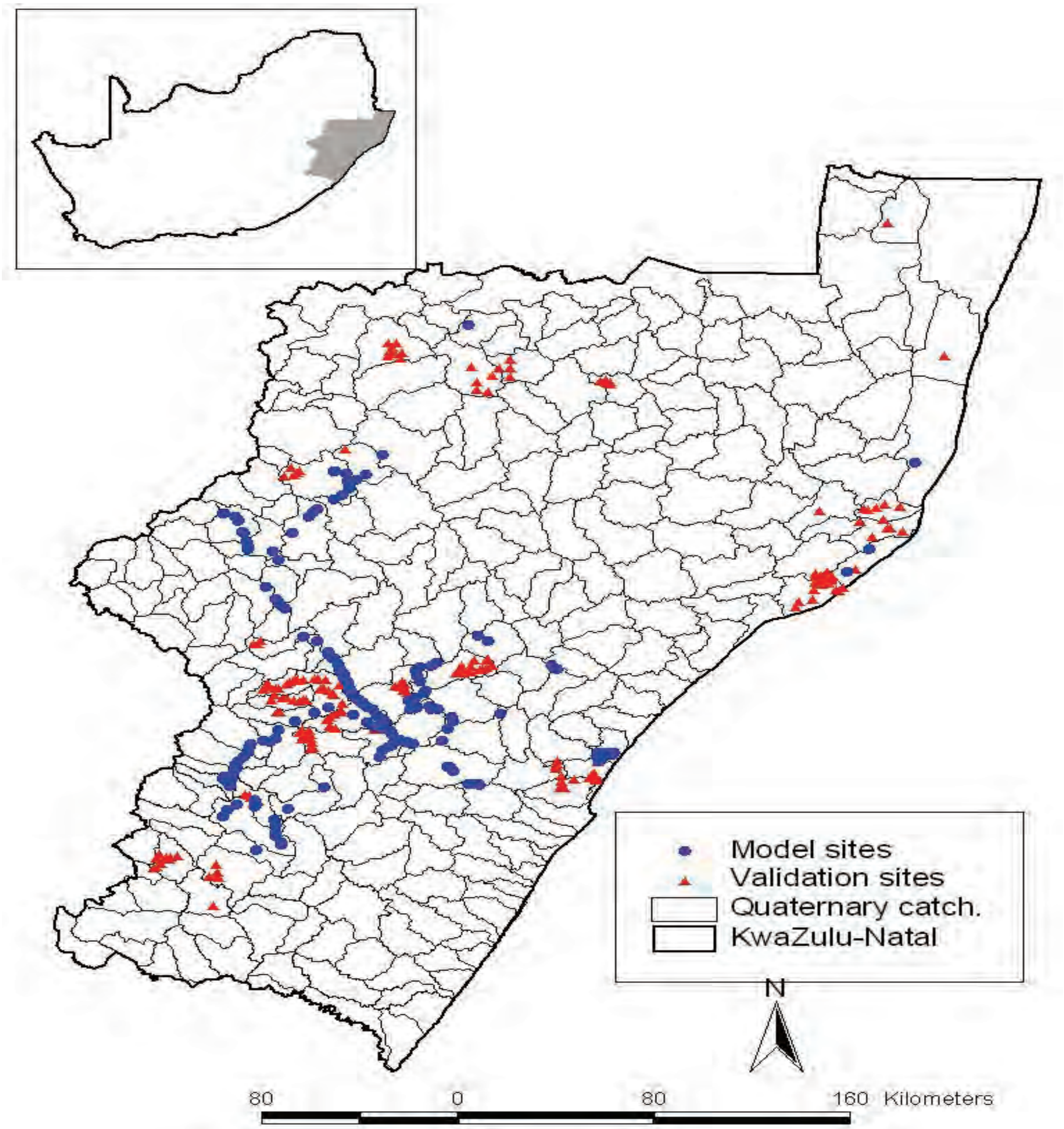


Figure 1 Study area defined using quaternary catchments falling entirely within KwaZulu-Natal. Centroids for wetlands used in the model development and model validation are indicated.

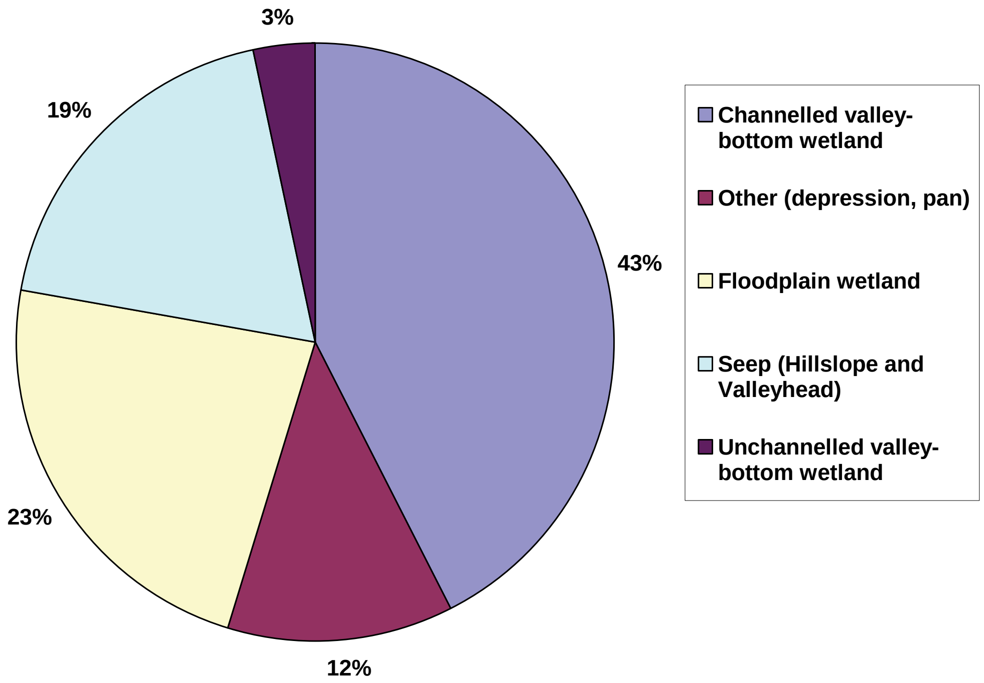
Wetland data collection

For development of wetland condition models which would be consistent with national-level wetland classification systems, we used hydrogeomorphic (HGM) units, which fall into Level IV of a 5-level functional hierarchical wetland classification (SANBI 2009), an extension of the previous classification by Ewart-Smith et al. (2006). For the classification used, levels I-III refer to degree of connectivity to the ocean, ecoregion and landscape setting respectively, while Level V describes hydrological regimes (perenniality and periodicity of inundation). While wetlands in KwaZulu-Natal have been classified using both vegetation types (Rivers-Moore and Goodman 2010) and HGM units, the latter classification has been incorporated into a national wetland coverage where all wetlands have been classified according to the SANBI (2009) classification using automated procedures. For consistency with national initiatives, and for pragmatic purposes of how wetland condition is typically assessed in the field using existing assessment tools, our model has been developed using a HGM unit classification. Of the eight HGM units recognised by this classification, this research focussed on four HGM units:

- Valley bottom (unchannelled)
- Valley bottom (channelled)
- Seep (Valleyhead and Hillslope seep types combined)
- Floodplain

While depression and pan HGM units also occurred within the province, they were excluded from the models because of unavailability of data. Of a conservative total in excess of 420 000 ha of wetlands in KwaZulu-Natal, the greatest area (*ca*. 180 000 ha – 43%) is comprised of channelled valley bottom types, with depressions contributing the least (6 400 ha – 1.5%). By count, valley bottom wetlands (channelled and unchannelled) make up the largest number of wetlands in the province (ca. 15 000 polygons – 40%), while seeps make up the next largest number of wetlands (ca. 11 500 – 31%), with floodplains contributing least to the province by number (178 – < 0.5% %) (Figure 2).

To collect wetland condition data, the most suitable approach was to undertake rapid assessments of as many HGM units as possible, rather than focussing on a few selected systems for more detailed assessments. This was achieved by rapid assessments of HGM units visible from a vehicle along main and district (major and minor) roads. Owing to this constraint, wetland centroids were recorded, and wetland area was not used. While the WET-Health assessment framework does assign scores for each of the components assessed, i.e. hydrology, geomorphology and vegetation, for the purposes of the model it was necessary to utilize the composite score for the assessed wetland, which derives a score by weighting the components 3:2:2 respectively, and align the ten-point scoring system with the A-F scoring system used by the national Department of Water Affairs, where A is natural and F is totally degraded. Wetland condition (A-F Present Ecological Status (PES) score) per HGM unit for 105 sites was collected from 17-18 and 21-22 February 2011 for wetland modelling, and 12 August 2011 where 39 HGM units were assessed to collect data for model validation. An example of the data collected is provided in Appendix I. These data were



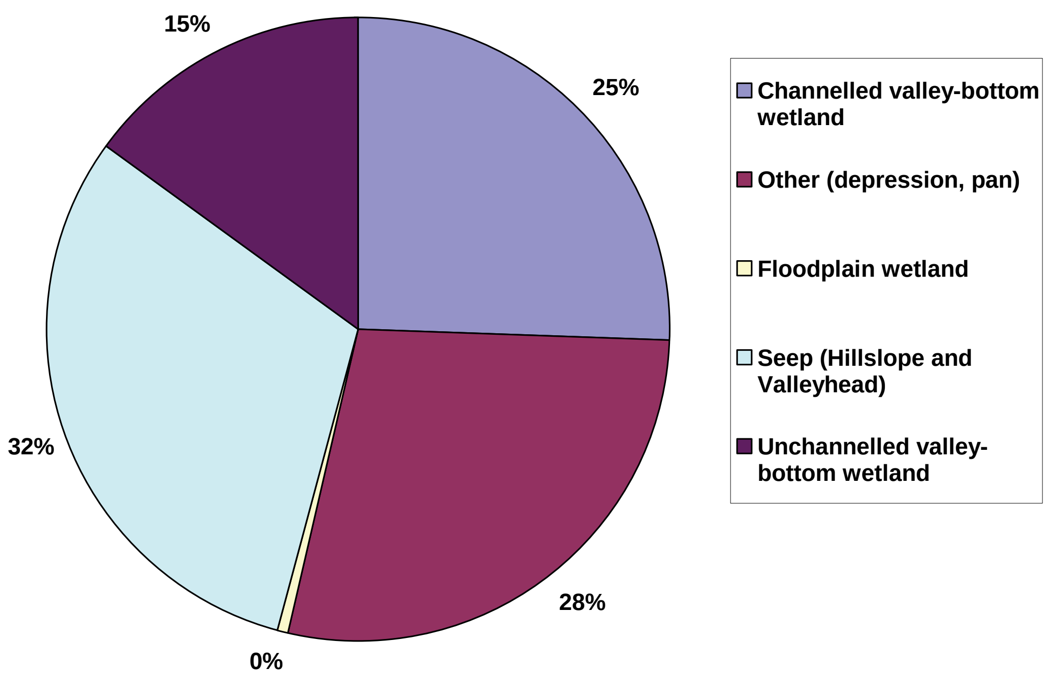


Figure 2 Pie chart of HGM units by area (top) and number (bottom) in KwaZulu-Natal

supplemented using wetland condition assessments from previous provincial vehicular surveys of linear features, such as pipeline and road alignments (Table 1). The nature of the fieldwork undertaken to derive the abovementioned information was considered to have limitations, with the derived PES scores being a localised ‘snapshot’ of the conditions within the identified wetlands, which may impact on the results obtained.

Table 1 Wetland condition data available for KwaZulu-Natal. Data from Macfarlane et al. (2011) were used for model validation

Data Source	Region Covered	No. HGM units with PES Scores
KZN State of Wetland Reporting 2011 (Macfarlane et al 2011)	Large-scale wetland systems throughout the province	204
Mondi State of Wetland Reporting	Mondi Forests properties throughout KZN	6
EIA, WULA Studies	Coastal regions of KZN, Midlands and Coast to Mountains,	36 + 73 respectively

Spatial information

A 20 m 2005 land cover image was used as the basis for calculating % land cover per category at tertiary and quaternary catchment scales, as well as at a 1000 m radius buffer for wetland centroids (EKZNW 2010). This date of land cover image was chosen in preference to more recent (2008) land cover images for two reasons: some of the wetland condition assessments being used for this study pre-date 2008, and because a lag between land cover change and wetland condition would be better correlated with an earlier land cover image. This land coverage had 43 categories, some of which were aggregated into broader categories according to Appendix II. The percentage coverage per category was calculated at three different scales (1000 m buffers, quaternary and tertiary catchments).

Population density (number of people per km^2) per tertiary and quaternary catchment was calculated using the 2001 South African Census data as the most accurate and available population density data to use for the model (Statistics South Africa 2002). Data were available as number of people per municipal ward. These data were converted to density values (number of people divided by ward area) per catchment (Figure 3).

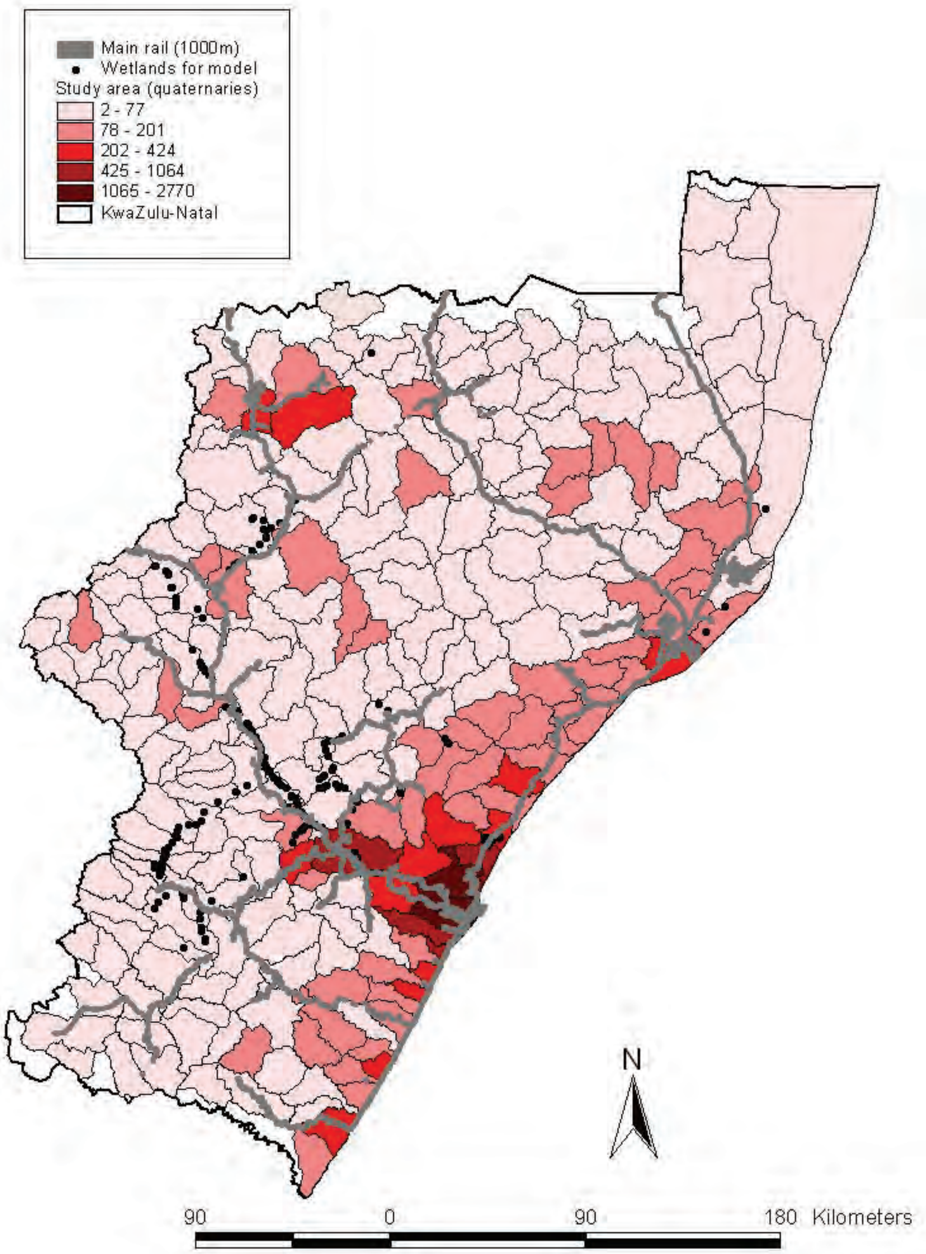


Figure 3 Population density (by class intervals) per quaternary catchment for the study area, based on the 2001 Census data, and main rail network in KwaZulu-Natal, buffered by 1000 meters, and showing relative positions of wetlands with PES data available.

Main road and rail corridors typically result in ribbon development adjacent to them. Potential land use changes and disruptions to wetland hydrology and geomorphology are associated with such corridors, which become less marked with distance from roads and rails. To include the effects of main roads and rails on wetland condition as a function of distance, these were included into the model by buffering the 1:50 000 provincial main roads and rails by 1000 m (ESRI 1999) (Figure 3) (CD: SM 2010). Wetlands were assigned a binary value based on how they were located relative to the buffer distances.

Additional data per catchment (tertiary and quaternary) derived from 1:50 000 topo-cadastral maps for the province were number of dams, and road density, where tracks/informal roads were excluded as these features are generally characterised by direct disturbance and limited infilling or alterations to surface flow characteristics (CD: SM 2010). Stream network density was calculated from a 1:500 000 scale rivers coverage (DWAF 2005a), as the 1:50 000 rivers data were irregular between map sheets. Further information on model variables is included in the model development section. Topographic data (slope, altitude, and basin characteristics) were calculated using a raster-based GIS (Clark Labs. 2009) and a 90 m elevation model (Schulze 2007) as input. Further detail on model inputs is provided in the section below.

Model development

In this study, we developed and tested linear and non-linear multiple regression models. A total of twenty one variables were used in model development (Table 2). Where necessary, data were transformed using suitable transformations – percentage data using square root transformation, and large numbers using log transformation (Steel and Torrie 1981). For both modelling approaches, we estimated percent land cover at a range of spatial scales, viz. tertiary catchment, quaternary catchment, and for a 1000 m buffer around each wetland point.

The linear models were fitted in the statistical package R using stepwise multiple regression routines (R Development Core Team 2009) for four wetland HGM types (seeps, floodplains, channelled and unchannelled valley bottoms) using raw, single PES values, where categories A-F were re-coded as 1-6. A linear response to land use change was assumed, and the models provide a direct condition score.

Next, models were developed to estimate the probability of a HGM unit being degraded for four HGM unit types. All wetlands with a PES score of D, E, F were defined as degraded and assigned a score of 1, where D is defined as “Largely modified = a large change in ecosystem processes and loss of natural habitat and biota has occurred” (Kotze et al. 2011). All A, B or C wetlands were classified as being in good condition, and assigned a value of 0. This threshold was slightly more conservative than a threshold of E recommended by DWAF (2002, cited in Dickens et al. 2003). Logit binomial models in the form of Equation 1, together with 95% confidence intervals, were fitted in the statistical package R using the binary condition values (0, 1) as the response variable (R Development Core Team 2009), to estimate the probability of degradation, together with 95% confidence intervals. Stepwise regression was used to reduce the maximal models to the minimal adequate models (Crawley 2007). Models were assessed using Akaike’s information criterion (AIC), which assesses goodness of fit but penalizes models for too many extra parameters (Crawley 2007), such that the lower the AIC, the better the model. Models were tested for statistical significance using Chi-square tests ($p < 0.05$) on residual deviance on degrees of freedom. From these models, odds ratios linking the odds of a wetland being degraded based on changes in the input variables were calculated (Equation 2).

$$p = \frac{e^{\alpha+\beta x}}{1+e^{\alpha+\beta x}} \qquad [1]$$

where α is a constant, and β is a coefficient for variable x, and where x could be, for example, % land use type per quaternary catchment

$$\Psi = e^{\beta} \qquad [2]$$

where Ψ is odds ratio

Table 2 Predictor variables for wetland condition models

Variable	Term	Units	Description
x1	Stream order	N/A	Calculated by assigning the highest stream order to occur in each quaternary catchment
x2	Population density	people/km2	Population density per catchment (2001 census)
x3	Road-1000	T/F	Wetlands falling within 1000 m buffer of main road
x4	Rail-1000	T/F	Wetlands falling within 1000 m buffer of main rail
x5	Slope	degrees	Slope of pixel on which wetland centroid located, calculated from a 90 m elevation image
x6	Altitude	m amsl	Altitude where wetland centroid located, as well as modal altitude per catchment using a 90 m image
x7*	Plantation	%	Forestry plantation (includes clearfelled areas)
x8*	Sugarcane	%	Sugarcane (commercial and emerging farmer)
x9*	Dense_sett	%	Built up dense settlement
x10*	Low_sett	%	Low density settlement
x11*	Agric_dry	%	Agricultural cultivation (dryland)
x12*	Agric_irri	%	Agricultural cultivation (irrigated)
x13*	Natural	%	Natural land classes
x14*	Degraded	%	Degraded land classes
x15*	Dams	%	All inland dams
x16	Dams-No.	N/A	Number of dams per quaternary catchment (1:50 000 scale)
x17	Basin length	km	Straight distance from basin outlet to farthest point on drainage divide (Frimpong et al. 2005)
x18	Drainage shape	N/A	Area divided by square of basin length (Gordon et al. 1992)
x19	Relief ratio	N/A	Difference in altitude divided by basin length (Gordon et al. 1992)
x20	Drainage density	km/km^2	Density of rivers per quaternary catchment (1:500 000 scale)
x21	Road density	km/km^2	Density of roads per quaternary catchment (1:50 000 scale; no footpaths)

Model validation

The second half of the HGM unit condition data was used to validate the models. Expected PES scores, and probabilities of degradation, were calculated at the most appropriate spatial scale (quaternary catchment) within the study area, using the models for all four HGM types. HGM units with an expected probability of degradation of >50% were assigned a value of one (i.e. degraded). Statistically significant differences between expected and observed PES scores were tested for using a Chi-square test ($p < 0.05$). Additionally, modal (± standard deviation) differences between observed and expected PES scores were calculated, together with bar charts of the frequencies of deviations per PES class (%). A similar approach for the binary data (degraded versus non-degraded) was not possible, and instead the prediction success (%) was calculated per wetland type for the validation data.

RESULTS

A total of 463 HGM units were used in this study, where 220 records provided model input, and 243 points were used in the validation. Valley bottom (unchannelled) were the most numerous HGM units (*n* = 80), followed by channelled valley bottom HGM units (*n* = 76), seeps (*n* = 35) and floodplain HGM units (n = 26). Data on depressions and pans were too limited (*n* = 2 for each) to use in the model development. For the data used to develop the models, seep and valley bottom wetlands all had a median condition of 4 (D), while floodplains had a median condition of C (3) with outliers (Figure 4). Validation data for floodplains and channelled valley bottoms had the same median values as the data used for the modelling, while seeps from the validation data were in better condition (median = 1) and unchannelled valley bottom types had a median value of 3 versus 4 for the model development data set (Figure 5). Local impacts on wetlands surveyed included a range of the following: alien invasive vegetation, developments (power-lines, roads and rails, housing), drains and channels, and land transformation (agriculture, pasture, erosion).

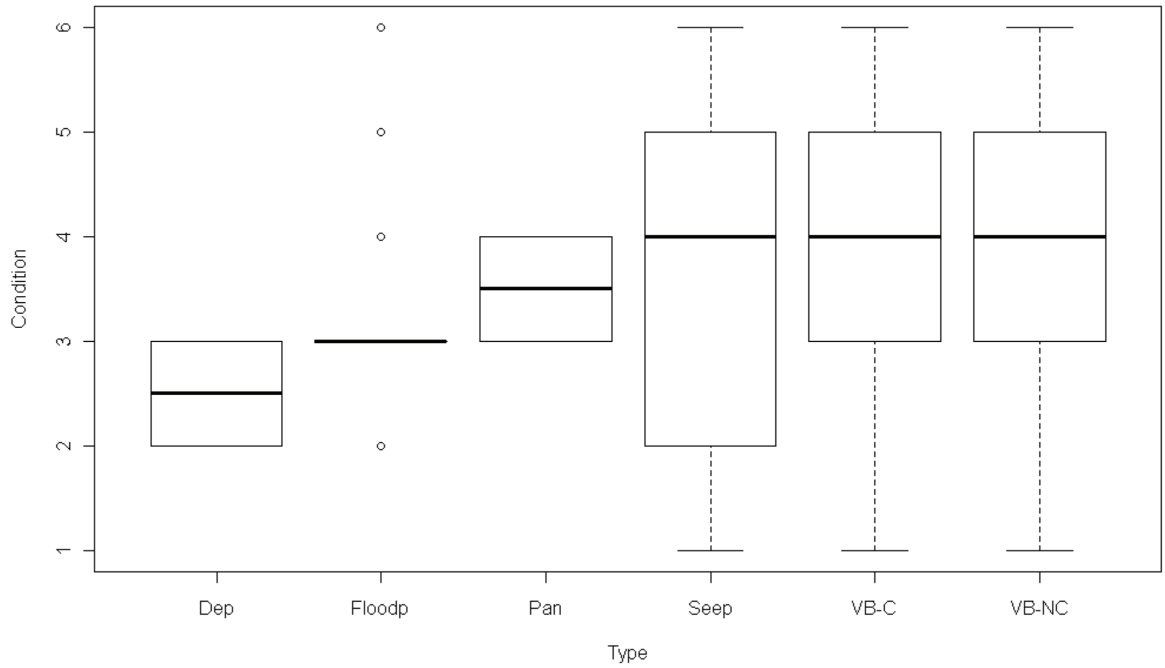


Figure 4 Box-and-whisker plot of wetland condition per type for HGM units (Dep = depression; Floodp = floodplain, VB-C and VB-NC are channelled and unchannelled valley bottoms respectively) used in model development. Bold horizontal lines indicate median PES values, while boxes show 25th and 75th percentiles respectively. Whiskers indicate maximum values, while outliers are shown individually

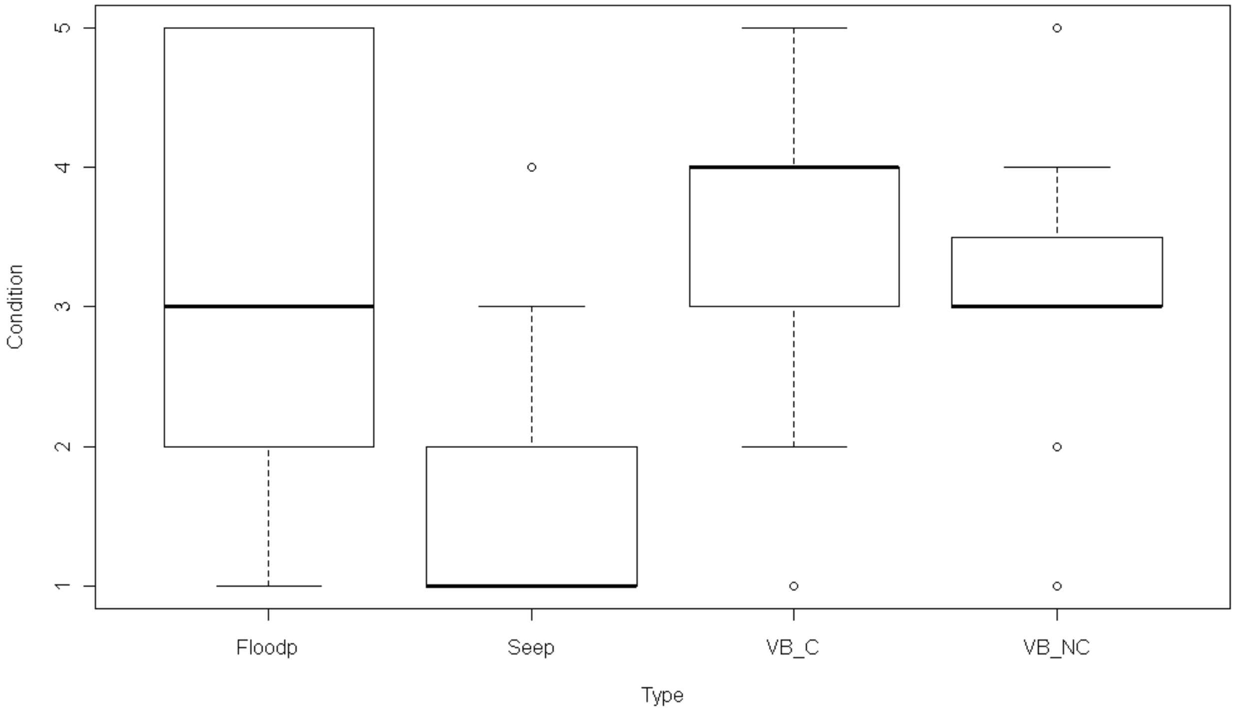


Figure 5 Box-and-whisker plot of wetland condition per type for HGM units (Dep = depression; Floodp = floodplain, VB-C and VB-NC are channelled and unchannelled valley bottoms respectively) used in model validation. Bold horizontal lines indicate median PES values, while boxes show 25th and 75th percentiles respectively. Whiskers indicate maximum values, while outliers are shown individually

Model assessment

Resolution of data for models at a tertiary catchment was too coarse to be of use in modelling wetland condition. The best model fits were achieved using data at a quaternary catchment scale, with models using land cover data at a 1000 m radius around wetland centroids providing slightly worse R^2 values than for models at the quaternary scale.

Multiple linear regression models to estimate average PES scores showed a range of explanatory power, with R^2 values ranging from 0.20 to 0.45. Models for three of the four HGM unit types were statistically significant. The model to estimate PES scores for unchannelled valley bottoms, in spite of having the highest R^2 value, being non-significant ($p > 0.05$) (Table 3.1, and see Appendix III for worked example). The standard error of PES score residuals was approximately one for all four models. In general, natural vegetation and number/ area of dams emerged as strong predictors of PES scores for HGM units. More specifically, the best predictor of PES scores for seep HGM units was percent natural vegetation remaining per quaternary catchment. For floodplain HGM units, percent land cover converted to irrigated agriculture, and number of dams, were identified was strong predictors of PES score. Predictor variables of both types of valley bottom differed, with altitude, percent natural vegetation and percent area of dams as predictors for unchannelled valley bottom HGM units. Conversely, for channelled valley bottom HGM units, percent plantation, natural cover were suitable land cover predictors, while basin characteristics (basin length and relief ratio) also contributed to the final model. PES scores across the province for the four HGM unit types modelled exhibited spatial heterogeneity, and distinct differences in spatial patterns between types (Figure 6). Based on the models, the majority of catchments in KwaZulu-Natal were most likely to have PES scores of 3-4, with unchannelled valley bottoms having the highest number of degraded scores (PES = 5-6) (Table 4).

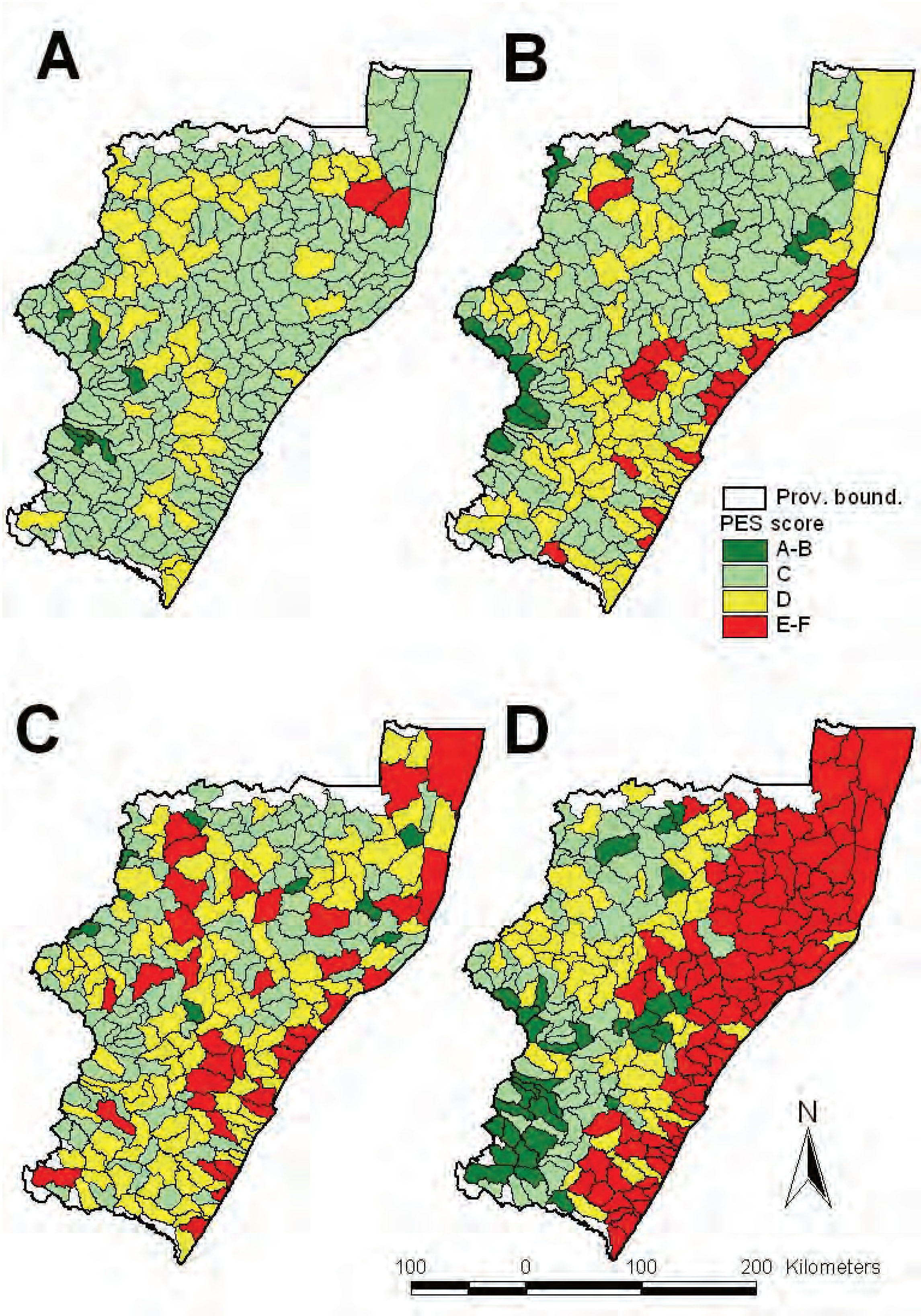


Figure 6 PES scores by quaternary catchment of floodplain (A), seep (B), channelled valley bottom (C) and unchannelled valley bottom (D) wetlands per quaternary, based on multiple regression models

Table 3.1 Multiple linear regression models to calculate PES for four HGM unit types

Type	Model	R^2	P	Residual S.E.
Valley Bottom (Chann.)	$PES = 4.663 - 0.035(x7) - 0.044(x13) + 0.055(x17) + 17.58(x19)$	0.27	< 0.001	1.09 on 71 d.f.
Valley Bottom (Unchann.)	$PES = 3.690 - 0.003(x6) + 0.046(x13) + 0.231(x15)$	0.45	0.079	1.05 on 75 d.f.
Floodplain	$PES = 3.181 - 0.134(x12) + 0.005(x16)$	0.20	<0.001	1.01 on 23 d.f.
Seep	$PES = 5.839 - 0.040(x13)$	0.29	<0.001	1.02 on 33 d.f.

Table 3.2 Logistic regression terms for models to estimate probability of degradation for four HGM types.

Type	Model	Residual deviance	AIC
Valley Bottom (Chann.)	$10.859 - 0.221(x7) - 0.200(x13) - 0.157(x14) - 0.008(x16) + 0.176(x17) + 38.360(x19)$	66.91 on 69 d.f.*	80.919
Valley Bottom (Unchann.)	$15.797 - 0.783(\ln(x2)) + 2.826(x4True) - 1.837(\ln(x6)) - 0.072(x7) - 0.966(x10) + 0.179(x14) + 0.371(x15)$	63.03 on 71 d.f.*	79.04
Floodplain	$-3.315 + 2.969(x21)$	26.96 on 24 d.f.	30.96
Seep	$2.728 - 0.003(x6)$	40.01 on 33 d.f.	44.01

* Model fit significant (χ^2 test, $p < 0.05$)

Table 4 Percentages of quaternary catchments falling within PES scores A-F for floodplain, seep, channelled (VB-C) and unchannelled (VB-NC) valley bottom HGM unit types. Mean (± standard deviation) probabilities of degradation for each of these types for the province are shown, based on probabilities of degradation calculated per type per quaternary catchment using the logistic regression models

PES score	Floodplain	Seep	VB-C	VB-NC
A	0.00	0.00	0.00	1.19
B	2.37	5.93	3.56	11.07
C	77.87	53.75	39.13	24.90
D	18.97	32.02	43.08	23.32
E	0.79	8.30	13.04	19.76
F	0.00	0.00	1.19	19.76
p(degrad)	0.30±0.27	0.56±0.28	0.58±0.34	0.44±0.36 (0.73±0.35)

The logistic regression models to estimate probability of degradation (i.e. PES = D, E or F) were significant for both valley bottom types ($p < 0.05$) and non-significant for seeps and floodplains ($p > 0.05$) (Table 3.2). However, the small number of terms and associated relatively low AIC values for the floodplain and seep types added value to these models. Explanatory terms differed in many cases from the multiple regression models. The best predictor of floodplain condition was road density network and a road density of > 3km.km^{-2} resulted in a probability of 1 of floodplain wetlands being degraded (Figure 7). Using the β coefficient of 2.969 to calculate odds ratios based on road density, and rescaling by a factor of ten, it can be generally said that each tenfold reduction in road density results in the chances of a floodplain wetland being in a good condition increasing by 1.35 times. For seeps, the best predictor was altitude, where HGM units at an altitude of less than 1000 m are likely to have a >50% chance of being degraded (Figure 8). Converting to odds ratios, for every 100 m rise in altitude, seeps are 0.77 times more likely to be in a good condition. Ninety-five percent confidence intervals for both models were wide. Models for the channelled and unchannelled valley bottom types required six and seven explanatory terms respectively, giving these models higher AIC values. Interestingly, percent degradation (bare ground) was a predictor of wetland degradation in both models, while proximity to main railway lines increased the probability of degradation for unchannelled valley bottom types.

Using the probability models, probabilities of degradation for HGM units for KwaZulu-Natal could be spatially represented (Figure 9). Floodplain wetlands showed a high probability of degradation in the coastal regions of the province, and in the more developed southern and central provincial regions. Seeps were most likely to be intact in the higher escarpment regions. Channelled and unchannelled valley bottom types showed very different spatial probabilities of degradation, with the latter showing a greater likelihood of degradation in the north of the province, and the former having a heterogeneous pattern based on catchment-specific patterns of land use. Valley bottom HGM units (channelled and unchannelled) were the most degraded in the province, according to the models (Figure 10 and Table 4).

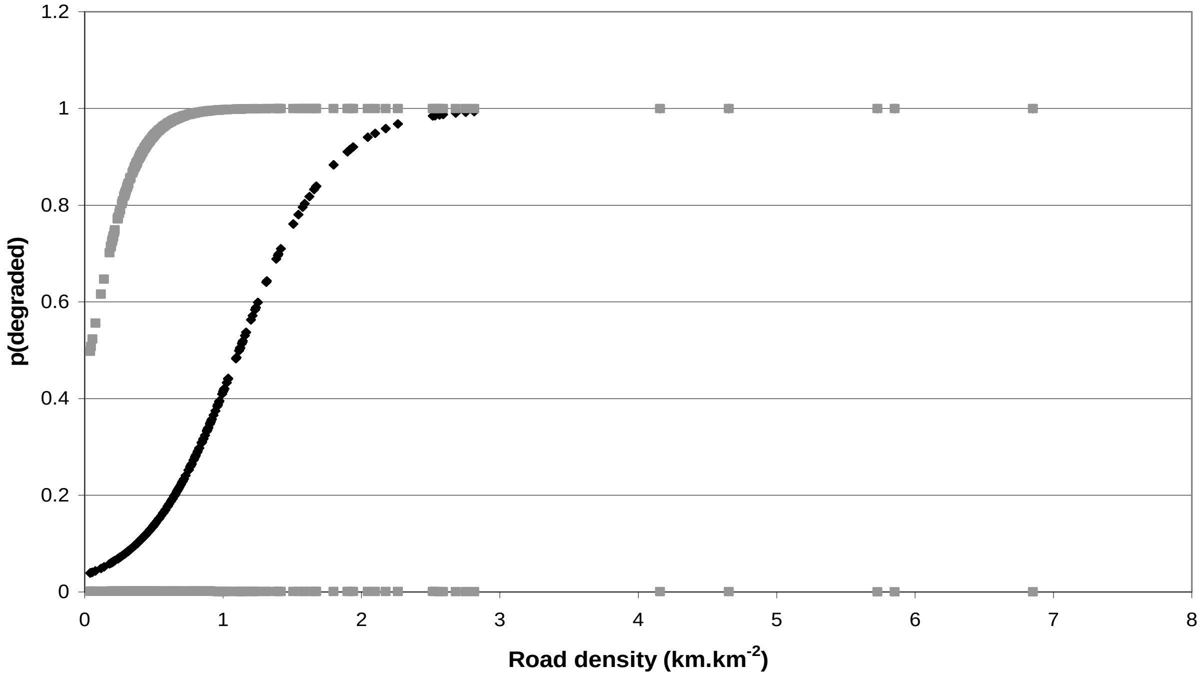


Figure 7 Probability of degradation for floodplains using road density as a predictor, with 95% confidence intervals indicated in grey

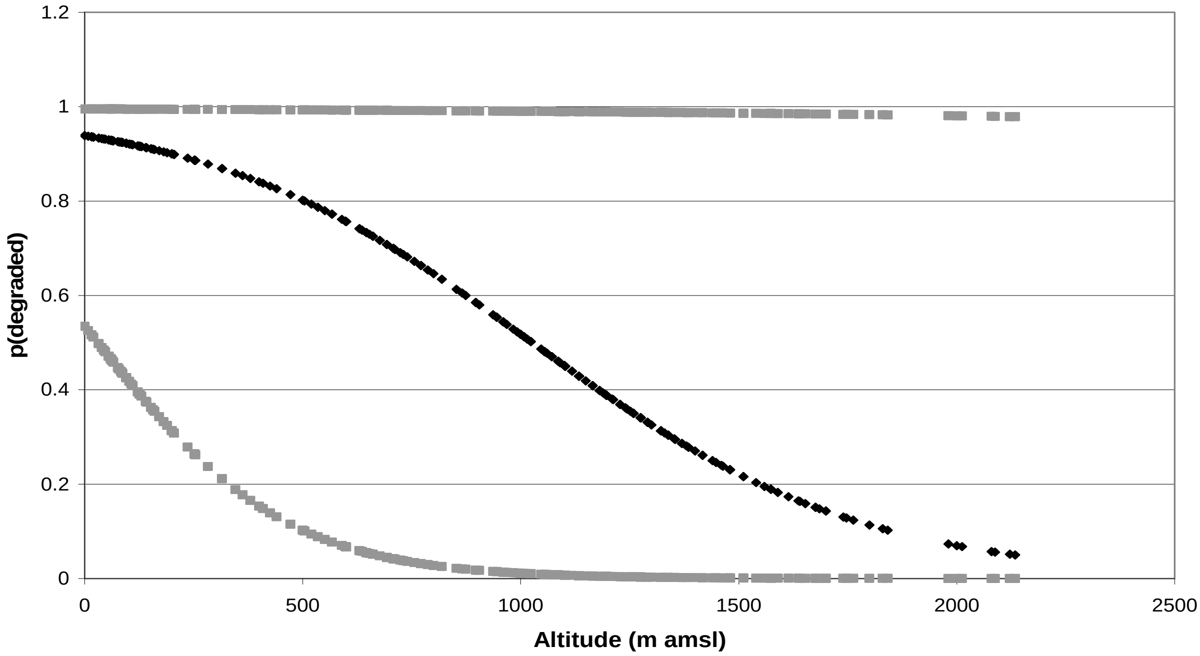


Figure 8 Probability of degradation for seep wetlands using altitude as a predictor, with 95% confidence intervals indicated in grey.

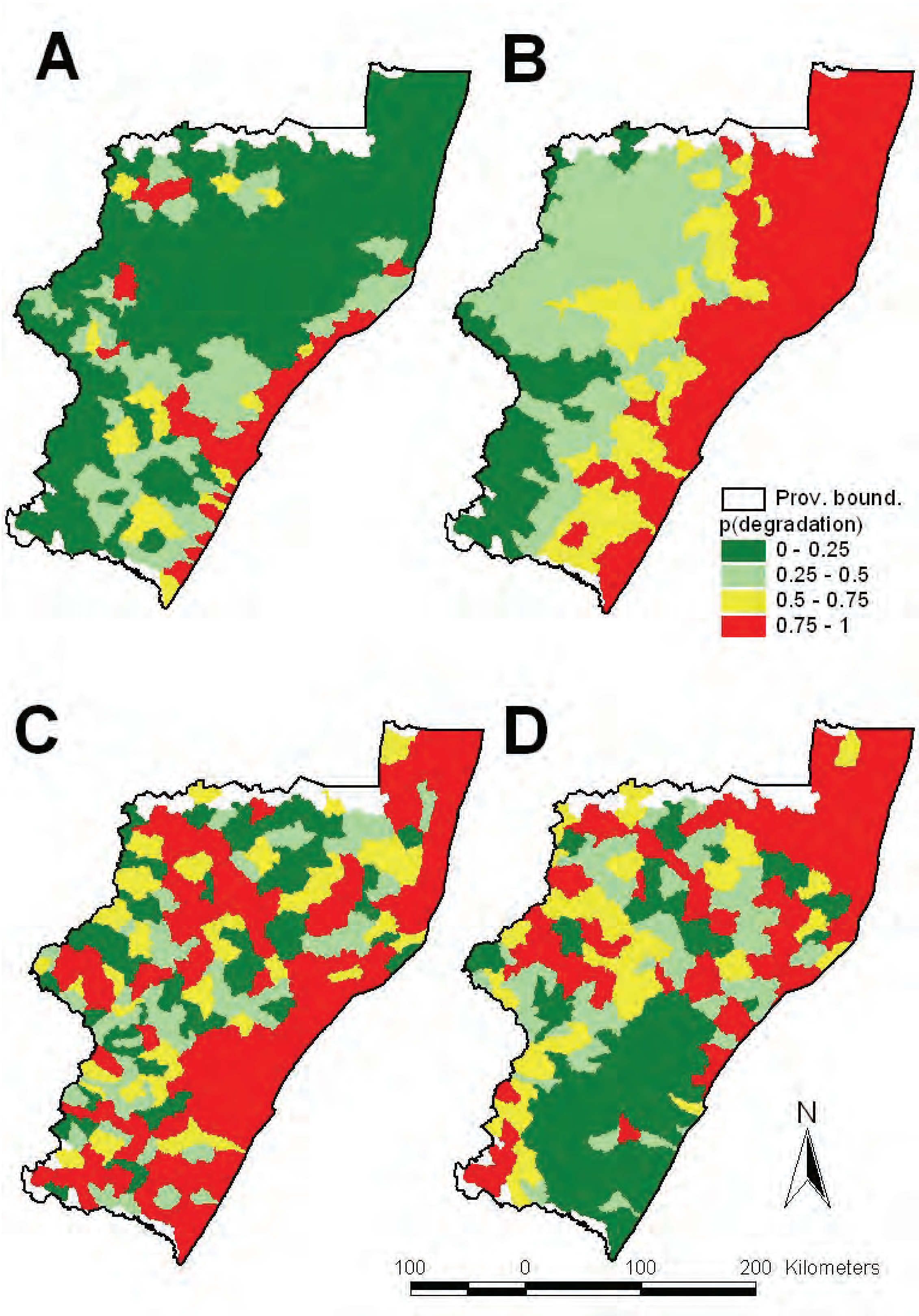


Figure 9 Probability surface of degradation of floodplain (A), seep (B), channelled valley bottom (C) and unchannelled valley bottom (D) wetlands in KwaZulu-Natal, based on logistic regression models

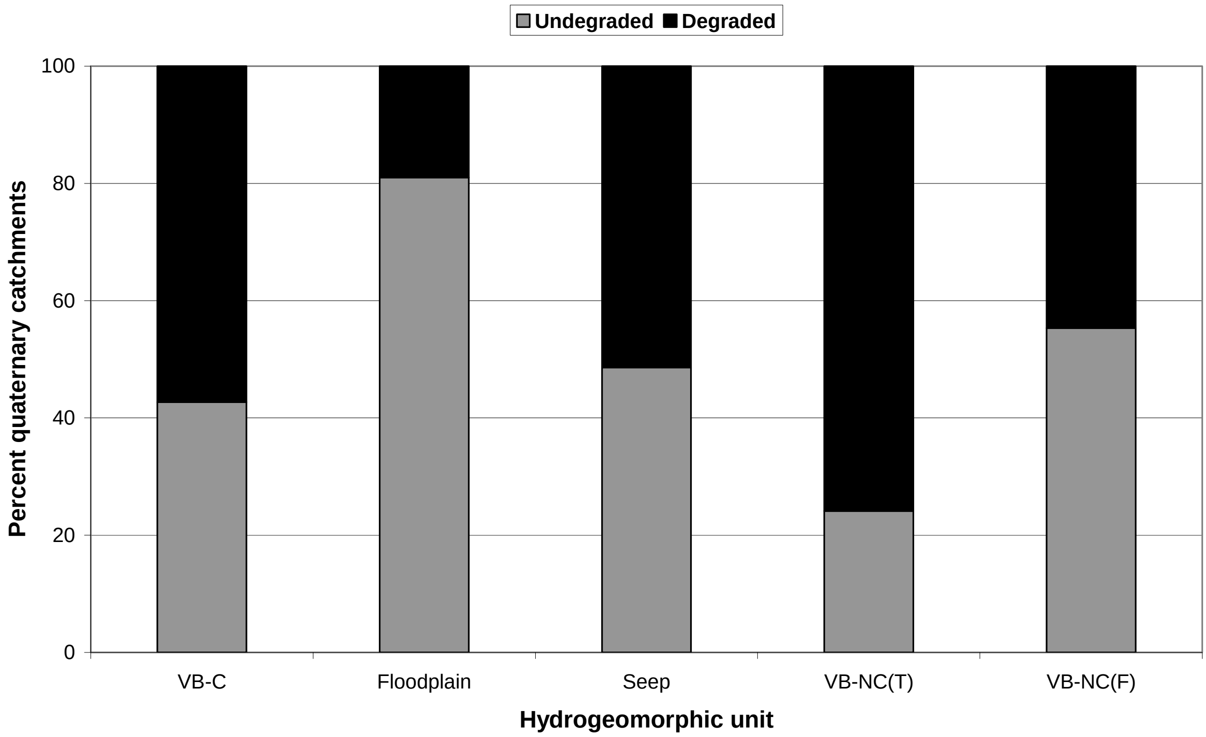


Figure 10 Percentage quaternary catchments with degraded versus non-degraded wetlands per HGM unit

Model validation

Model validation results indicated that models should be used with caution to predict condition of individual wetlands. Multiple regression models to estimate PES score class had a standard deviation of 1, and a modal difference between observed and estimated PES class of less than (unchannelled valley bottom and Seep) or greater than (Floodplain) 2 (Table 5). The model with the highest number of zero deviances between observed and expected PES scores was for channelled valley bottoms (Figure 11). The models used to predict the probability of wetland degradation were more robust than the former models, with approximately 90% of floodplain and seep HGM units being correctly classified as degraded or non-degraded, while almost one in every two valley bottom wetlands were correctly classified (Table 5 – Logistic regression models % correct).

Table 5 Validation for logistic regression models (showing % correct classification as degraded or not), and multiple regression models, where modal (± standard deviation) differences between expected and observed PES scores are shown. Chi-square tests in significance of difference between expected and observed scores per HGM type are shown (* = $p < 0.05$). Sample sizes per HGM type are illustrated

		Logistic regression models	Multiple regression models		
Type	***n***	**% correct**	**Mode (PES)**	**SD (PES)**	**χ^2 - PES**
VB (chann.)	72	55.56	0	1.30	34.45*
VB (unchann.)	118	44.07	-2	1.09	49.15
Seep	27	96.30	-2	1.04	37.58*
Floodplain	26	88.46	2	1.64	23.25*

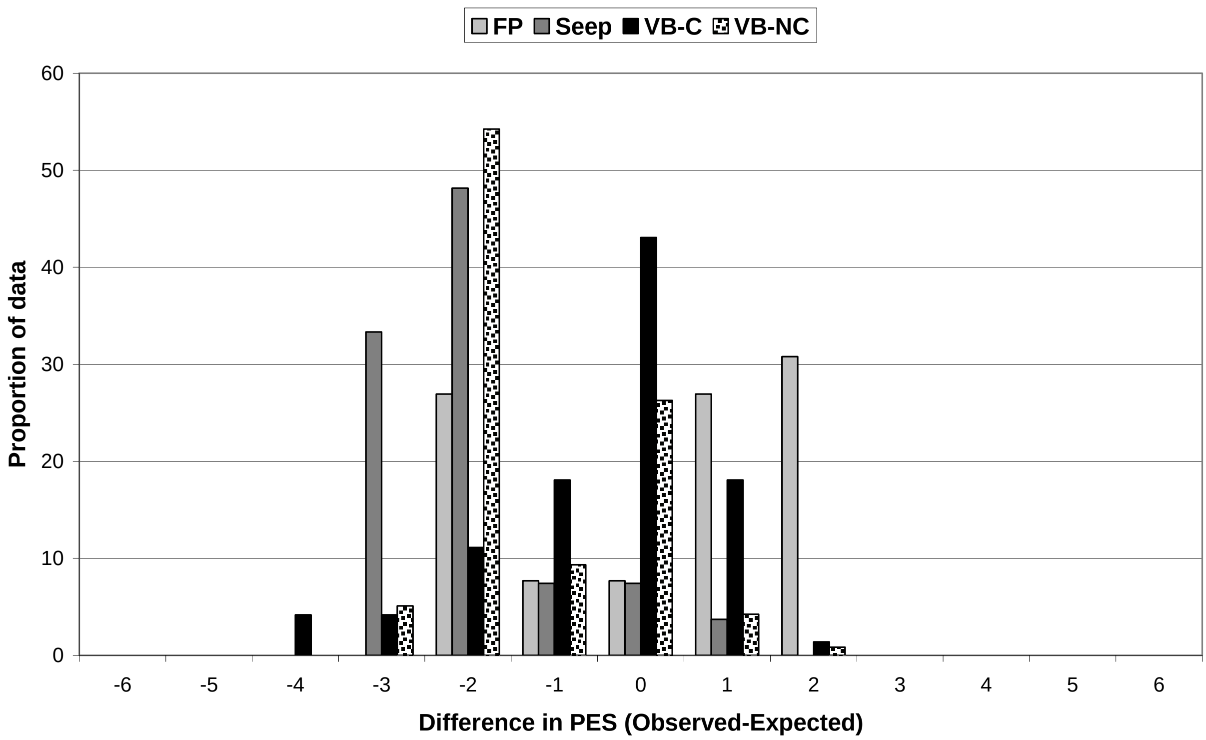


Figure 11 Proportion of data for different levels of error (differences between observed and expected PES scores) based on multiple regression models for floodplain (FP), seep, channelled (VB-C) and unchannelled (VB-NC) valley bottom HGM types

DISCUSSION

Impacts of scale on modelling success

A number of studies have illustrated that the scale of analysis often dictates whether relationships between data appear as interpretable patterns versus meaningless noise (Johnson and Gage 1997; Weller et al. 2007; Gutzwiller and Flather 2011). This makes intuitive sense as drivers of wetland condition occur at specific scales: habitat structure and organic matter inputs are local, while nutrient supply, sediment and hydrology are regional. Different environmental variables of streams can be expected to vary in their responsiveness to large versus small-scale environmental factors (Allan 2004). Furthermore, modelling at different spatial extents will potentially result in different strengths of observed relationships (Frimpong et al. 2005).

For the models developed to estimate condition of HGM units in KwaZulu-Natal, relationships at a tertiary catchment scale were largely meaningless, while relationships at a quaternary catchment scale provided the most meaningful relationships. Strength of relationships between HGM unit condition and land cover predictors within a 1000 m buffer of a HGM unit centroid were intermediate between the previous two scales. Similar findings of the influence of scale on data patterns have been reported elsewhere, which were not entirely comparable in terms of specifics with the findings of this study, but similar in terms of the message of how important it is to develop and test models at a range of scales. For a series of circular buffers (570, 1710 and 5130 m radius) around wetlands, where land cover data was extracted, the risk of habitat loss was most strongly associated with conditions with 570 m, least at 1710 m and intermediate at 5130 m. (Gutzwiller and Flather 2011). Similarly, Weller et al. (2007) found that landscape indicators at 100 and 1000 m radii gave different model results.

Understanding scale impacts on wetland condition has not only important theoretical importance in terms of success in modelling condition, but also has important implications for effective management. While most decision making occurs at the local level (Allan et al. 1997), a number of studies have attributed more influence to catchment than to local land use (Allan 2004). The obvious management implication here is that decision making, while being local, should take cognisance of the appropriate broader spatial context.

Model Assessment

HGM unit conditions were successfully linked to landscape predictors, although model success varied between different HGM unit types. This is not atypical when compared with similar studies – Weller et al. (2007) linked wetland condition to remotely sensed data using stepwise multiple regression models with differing levels of success for different wetland types (R^2 of 63-85% variance of scores explained in riverine wetlands versus 48-54% for flat wetlands). While one potential limitation of our models is that relationships were based on an averaged single PES score, other studies have found that models developed for each component of HGM final condition (hydrology, biogeochemistry, habitat, plant community and landscape) provided results which were not as good as those using mean conditions (Weller et al. 2007).

Based on the verification statistics, the probability models were more robust than the multiple regression models to predict a PES score. In the development of the probability model, there was a need to determine thresholds for the PES scores recorded for wetland systems to dictate when a system was 'Degraded' versus 'Intact'. The setting of a threshold value is strongly linked to the purpose of setting this threshold. For example, if the study was focused on wetland rehabilitation, there may well be two threshold values implemented, where systems with 'A-B' and 'E-F' PES scores would be excluded from the prioritization process, as the first two classes have limited opportunity for rehabilitation and the last two classes are 'lost causes' in terms of rehabilitation. In this instance, the model is to be developed to inform conservation planning processes, to assist in identifying which wetlands are more prone to degradation and hence more likely to require additional protection within the region. Based on this objective, the thresholds utilized in the model will be based on scores 'A-C' reflecting an intact wetland system and scores 'D-F' reflecting a degraded wetland system. This breakpoint is slightly more conservative than the E/F threshold recommended by DWA (Dickens et al. 2003). This is primarily linked to the description of PES classes, with 'C' still representing a system which is relatively intact – "Moderately modified – a moderate change in ecosystem processes and loss of natural habitats has taken place but the natural habitat remains predominately intact". The understanding of ecological thresholds is critical, but it is not always clear how to identify thresholds before they occur (Lindenmayer et al. 2008). The concept of ecological thresholds remains a contentious issue for a number of reasons, as it is not only a departure from a baseline condition, but can only have an action assigned to it with an associated knowledge of measured vs. true change (i.e. is change in condition due to natural events or due to direct human intervention?) (Ladson et al. 2006), and the statistical power to detect change, linked to adequacy of sampling strategy.

Even after assigning a threshold, the response of a system to impact gradients may be linear or non-linear (Allan 2004; Lindenmayer et al. 2008). A linear approach is used between wetland condition and degree of land use disturbance for studies in the United States (US EPA 2003). In a similar approach, Ellery et al. (2009) assign wetland services per wetland type, and assume a threshold at the "moderately modified" (3/10) wetland condition score, after which there is a linear decline in ecosystem services. The non-linear approach inherently assumes the existence of ecological thresholds, following which there is a significant system change. For example, analyses by Allan (2004) found that streams in agricultural catchments usually remain in good condition until the extent of agriculture exceeds 30-50% the land use in the catchment. Similarly, for every 10% of altered catchment land use, a correlative 6% loss in freshwater diversity was noted, as a linear relationship (Weitjers et al. 2009).

For both modelling approaches, the high standard errors and wide confidence intervals of the models make prediction of actual, individual wetland conditions less useful. Model validation further illustrated that the risk of misclassifying HGM unit condition on a case-by-case basis is relatively high. According to the models, PES scores for channelled valley bottoms were correct, on average, per quaternary catchment. PES scores for the remaining three HGM types assessed (unchannelled valley bottom, seep, floodplain) were, on average, over-/underestimated by two categories i.e. a B would be modelled as a D or vice versa. The probability models were more robust than the multiple regression models, as a consequence of data resolution loss in reclassifying PES scores into degraded versus non-degraded categories. Consequently, for groups of wetlands assessed per quaternary catchment, one in ten would be incorrectly classified as degraded for seep and floodplain HGM types, while one in every two valley bottom HGM types would be misclassified. Risk in using these models to estimate condition is therefore higher when applied at the scale of individual wetlands, and lower when applied at a regional level to describe the average condition of HGM types at a catchment scale.

One inescapable issue in modelling wetland condition is that impacts on wetlands are usually a result of both within system and catchment activities, and the variable range of factors affecting the degradation of wetland varies between wetlands and between catchments (Kotze et al. 1995). Such issues alone make it difficult to develop models which account for high levels of data variance at a regional scale. Additionally, the quality, differing spatial resolutions and differing ages of the various input datasets, and different sources of wetland condition data, could all have introduced modelling error. The assessment framework used in this study accounts for impacts associated with catchment but not in-system impacts (Kotze et al. 2011). During the development of the condition models it was noted that in many instances, onsite impacts were the overriding impact on the ecological integrity of the selected wetlands. This trend is supported by Macfarlane et al. (2011), as on-site impacts were considered to score slightly higher than catchment-based impacts in the provincial assessment of the priority wetlands. If onsite impacts contribute significantly towards wetland degradation in the province, the model is unable to take this into account in its current form.

In spite of these criticisms, these models are useful regional planning tools, with the models enabling generation of spatial patterns of average wetland condition per planning unit (quaternary catchment). Weller et al. (2007) noted that while uncertainty in predicting condition scores for two wetland types was high, the models still helped to prioritize field visits to select sites for management action. The models additionally serve to indicate which land cover data to focus on in the future when assessing wetland condition, such as better data on number and areas of dams, and density of road networks. Another useful result emerging from these models was that channelled versus unchannelled valley bottom types appear to be subject to different combinations of drivers, and should be managed in distinctly different ways. While for both HGM types, the extent of plantation, natural vegetation and degraded land in any catchment were common predictors of degradation from the regression models (Table 3.1), other unique drivers were relevant to each type. For channelled valley bottom types, in addition to the drivers listed, catchment characteristics (basin length and relief ratio) and number of dams were significant predictors of PES score and probability of degradation. Conversely, for unchannelled valley bottom types, anthropogenic factors (population density, area of low density settlement, proximity to main railway lines) were significant predictors of condition, together with altitude and area of dams.

Landscape predictors and trends of wetland condition

In this study, it was assumed that the structure and dynamics of the wetland habitat and condition was determined by the surrounding catchment (Hynes 1975). Consequently, a major driver of HGM unit condition was assumed to be alteration of natural catchment land use into agriculture, industry or urban areas, and associated eutrophication (Weitjers et al. 2009). To account for such changes, Gergal et al. (2002, cited in Allan 2004) notes the expanding role of landscape analysis in catchment management. Anthropogenic landscape disturbances shift the structural and functional relationships among the landscape elements and the stability of the environment, by changing terrestrial to aquatic energy transfers (Schlosser 1991). Frimpong et al. (2005) cites a number of studies that correlate vegetation cover, geology, geomorphology and hydrology to ecosystem health.

Because aquatic systems are usually affected by multiple and interacting disturbances, matching a response to the responsible stressor can be very difficult (Allan 2004). Covariance between anthropogenic and natural gradients makes attributing relative influences of drivers of aquatic system health difficult (Allan 2004), and therefore correlative relationships may not be causative. There is strong evidence of the importance of the surrounding landscape and human activities to a stream’s ecological integrity (Allan 2004).

The catchment-scale drivers of HGM unit condition which have been statistically identified using a modelling exercise confirm a number of trends previously identified based on field assessments and expert opinion within the province. These include catchment impacts by afforestation and irrigated agriculture, urban development and erosion, with the coastal belt being subject mainly to agriculture (sugarcane) and urban development, and the inland margin being subject to erosion (semi-arid areas) and drainage in high rainfall areas (Kotze 1995; 2004; Macfarlane et al. 2011). Dams have been identified as having a significant catchment impact on geomorphic integrity of HGM units (Begg 1989; Kotze 2004, Macfarlane et al. 2011). Begg (1989) records grazing by livestock as one of the most significant degradation factors, followed by burning and drainage activities. Further to these drivers, specific land uses have been directly linked to levels of degradation – approximately 65% of the wetland areas in the upper Mgeni catchment were considered to have been highly transformed, linked primarily to current and historical cultivation and drainage (72% of the degraded systems), with dams being the next significant impact on these systems (19% of the degraded systems). Timber and urban infrastructure were the remaining degradation factors in the catchment (Kotze 2004). In other countries, similar predictors of wetland loss have been observed, with land cover and road density being the best predictors for models of wetland habitat loss in the southern United States (Gutzwiller and Flather 2011), and, *inter alia*, slope, elevation, proximity to roads in Costa Rica (Daniels and Cumming 2008).

Given the clear links between landscape drivers and wetland condition, it also becomes possible to make regional generalisations on conditional trends. The distribution of the predicted high probability of wetland degradation is aligned with a common trend of greatest loss at lower altitudes, described by Kotze (2004). Specifically, in the southern Kwazulu-Natal, Kotze (2004), states that where altitudes are greater than 1800 m, wetland loss was considered to be very low but wetland loss increased considerably with decreases in elevation, with more than 90% loss recorded within the 300-400 m zone of elevation. Kotze (2004) describes this trend being repeated in central KwaZulu-Natal, a trend also evident in this study.

Model application and integration with existing studies

Kotze et al. (1995) highlight that the term “wetland loss” implies that the loss in ecosystem functioning and integrity is irreversible, but rehabilitation is often achievable if hydrological conditions are restored. The modelling of wetland degradation could therefore be used to inform rehabilitation planning efforts within KwaZulu-Natal. Currently, the rehabilitation planning process adopted by the national “Working for Wetlands” programme relies on broad-scale information, such as conservation plans, to identify the focus areas for wetland rehabilitation efforts. Unfortunately, this approach sometimes identifies priority areas, which on the ground have limited opportunities for rehabilitation as these areas are not subject to degradational factors. The use of our models would provide an additional layer of information to overlay and therefore provide the planning process with more focussed areas to groundtruth.

Wetland condition models with a user manual fit into the critical path guidelines for including wetlands in a catchment management strategy (Dickens et al. 2003). This model will therefore address issues regarding defining future desired states and facilitate setting thresholds of potential concern. The wetland condition model provides an objective tool to set priorities and define management objectives on a catchment basis by providing probabilistic information on condition per type. Such knowledge feeds directly into the process of engaging with stakeholders and integrating with catchment management strategies to address wetland issues. The probabilistic models developed here are distinct from a recent modelling exercise as part of the National Freshwater Ecosystem Priority Areas (NFEPA) project, where wetland condition was estimated using the proportion of natural vegetation in and immediately surrounding wetlands (50, 100 and 500 m buffers, and using the best available wetland coverage) (Driver et al. 2011). The advantages of a probabilistic approach over a deterministic approach are numerous:

- They enable the calculation of odds, allowing conservation planners to make comments such as "floodplain wetlands in catchment x are three times as likely to be degraded than channelled valley bottom wetlands" and "wetlands of type *x* are likely to be five times more threatened within KwaZulu-Natal than wetlands of type *y*, which allows for the prioritization of one type of HGM unit over another;
- Wetland condition per quaternary catchment can be estimated independently of a wetland coverage, thus allowing conservation planners to prioritize areas where wetland data are poor;
- Wetland condition can be estimated using a range of statistically significant variables which are broader than only land use;
- Future scenarios of changes on wetland condition per type based on projected land use and population growth can be assessed.

Areas of predicted high levels of wetland loss generated by these models are strongly aligned with the hypothesised extent of provincial wetland degradation identified by Kotze et al (1995). In terms of NFEPA, Macfarlane et al. (2011) identify that virtually all wetland types within the Indian Ocean Coastal Belt are classified as critically endangered. Once again there appears to be strong correlation between NFEPA's results and those generated by these models. In terms of the models, valley bottom HGM units, the most numerous in the province and most likely key in ecosystem services, were the most degraded. Regional assessments such as this allow planning decisions to be made, such as to focus on valley bottoms, and in defining reference condition wetlands.

Although the concept of condition is subjective and value-laden, and various measures of reference condition exist, it is increasingly used to describe the standard or benchmark against which current condition is compared (Stoddard et al. 2006). There is no scientifically credible way of defining what is acceptable versus what is not, and remains a societal decision involving complex trade-offs among human values and benefits (Richter 2009). Reference condition describes a statistical distribution rather than a single absolute value i.e. temporal and spatial variability that is inherent in any measure chosen to represent the natural state of environmental systems (Stoddard et al. 2006). Because of the value in capturing the natural range of variability, system variance, and the value of assigning confidence intervals, Ladson et al. (2006) regard random sampling as a more beneficial approach than selecting representative areas.

RECOMMENDATIONS

Recommendations on the future use and application of these models fall into two co-dependant themes – firstly, exploring options of the uptake of these models regionally and nationally by suitable agencies, such as conservation authorities, and secondly, on recommendations for future research to improve the models' predictive power. For the first theme, there is potential for defining hypotheses on differential application of management interventions and testing these. For example, different wetland types could potentially have different buffers applied depending on their classification. The delineation of buffers assumes that a wetland itself has been accurately delineated, and tools exist for this (DWAF 2005b). There is further scope and opportunity to test the effectiveness of existing buffers (proposed as 20 m as a minimum – DWAF 2005b) on wetland condition, by using buffer distance (when available) as one of the explanatory variables in a wetland condition model. Critical to relating to triggers for management actions for rehabilitation and/or conservation is to re-examine the cut-off point of degradation thresholds. In this study, the threshold was made at the A-C versus D-F split for non-degraded versus degraded HGM types. In terms of management targets used by the DWA and SANBI, the categories A-B (conservation value or management target), C-D (potential for rehabilitation) or E-F (totally degraded, not acceptable as potentially not able to be rehabilitated) could be a more useable split to guide rehabilitation prioritization. Such a split would result in 3-class PES models versus the current 6-class PES models which were less robust than the probability models.

For integration with management plans by provincial conservation authorities, the outputs from these models could be used as inputs into conservation planning software. There is also value in assessing the strength of correlative relationships between wetland condition per HGM type versus existing regional freshwater condition indices. Finally, there is an initial incompatibility between wetland classification systems designed for biodiversity conservation and based on vegetation types (for example, Rivers-Moore and Goodman 2010) versus classifications based on hydrogeomorphic types to address wetland rehabilitation (SANBI 2009). Such a disjoint could potentially be solved by breaking the HGM types used for the models in this research into finer HGM types to include vegetation types (for example, tall grassland seep versus short grassland seep). This way, a regional condition assessment of HGM types could more explicitly feed into conservation planning initiatives by guiding differential setting of targets based on levels of threat.

To address the second theme, viz. to improve the predictive power of the wetland condition models and provide greater confidence when being used as a tool to inform rehabilitation planning processes, these models could be refined to focus on the different components of system integrity, namely hydrology, geomorphology and vegetation. This would allow 'lost causes', for example where geomorphology has potentially been significantly degraded, to be eliminated from the prioritisation process. Due to the limitations associated with the project, the use of existing data was necessary to inform the models, which precluded obtaining information relating to the size of the various wetlands. However, size is considered to be an important criteria to ultimately be included in the models as smaller wetlands tend to be steeply sloped, narrow, readily accessible and the edge to area ratio increases susceptibility to alien invasive plants (Kotze, 2004), thereby potentially increasing the probability that these systems would be degraded. In addition to size, and assuming there is ongoing collection of condition data for different HGM units across a wider spatial area, there is scope for refining these models using additional candidate explanatory variables which previous studies and logic suggest might be important. These could include, *inter alia*, additional rainfall parameters (amount, intensity and variability), as rainfall is a key driver of wetland function; soil parameters (types, depth and erodibility); and vegetation types (as different vegetation types would have different levels of resilience to catchment degradation). To illustrate, Kotze (1995) has identified rainfall intensity and slope as potentially contributing towards erosion and deterioration of wetland conditions, as well as making distinctions in the types of land ownership seen as impacting on wetland loss (private, community, state). Subsequent iterations could also consider abstraction ratios and water stress in catchments. Furthermore, this research has identified that the scale at which correlations between HGM type condition and landscape predictors are examined determines the strength of the statistical relationships (i.e. what may appear as noise at one scale appears as pattern at another scale). Subsequent iterations of these models should be run at different scales not considered so far, viz. sub-quaternary catchments and a wider range of buffer distances (2000 m).

Two additional issues not dealt with explicitly in this research, but nevertheless important for future research, are assessing the relative importance of local interventions (for example, digging of furrows) versus landscape drivers in accounting for variability in the models. Different HGM types may exhibit different degrees of likelihood of local interventions. Secondly, the models assume a uniform spread of different HGM types across the landscape, and conditions are assigned accordingly. The spatial assessment of HGM type conditions should be complemented with research on the spatial distribution of the different HGM types. In this way, it becomes possible to make statements such as “the likelihood of floodplain wetlands in this catchment being degraded is x%, although there is only a y% likelihood of finding floodplain wetlands here”. With increased predictive power, the models become the basis for being able to make comments on trends in condition between different HGM types and across landscapes linked to particular drivers. Thus, for example, comments such as “the likelihood of degradation of valley bottom wetlands increases x-fold when the number of farm dams exceeds *n* [as a threshold value]” become useful rules in a decision making process when assessing an application to build a new dam. This presents a potentially powerful tool for conservation authorities when engaging with planners and during negotiations with competing land users.

The modelling approach developed in this research could be applied nationally, as a significant tool in identifying areas of conservation intervention specific to particular HGM units at the quaternary catchment scale. The explanatory datasets used for this study are all available nationally, and the major limitation to achieving this is the lack of a single database for South Africa of HGM units with associated, reliable condition scores. Compiling such a database would provide data for a wider range of HGM unit types over a larger area, and would be key to testing and expanding these models to a national level.

CONCLUSIONS

The models show promise, and are useful for high-end regional planning. Models are preliminary and should be refined using more data, and covering a wider area. Development of such models is in line with current international wetland research. The models in no way make existing initiatives in South African wetland planning redundant, and rather complement these. There is potential to refine and extend the models for application at a national level.

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APPENDIX I

Example of data of wetlands assessed for current model, with type and PES value

No.	Date	Latitude	Longitude	Altitude	PES	Type
1	10-Feb-11	-29.31415	30.14268	998	F	VB_C
2	10-Feb-11	-29.31119	30.14057	997	E	VB
3	10-Feb-11	-29.32073	30.12656	1023	D	VB_NC
4	10-Feb-11	-29.32793	30.12167	1051	C	VB_C
5	10-Feb-11	-29.32893	30.12036	1053	D	VB_NC
6	10-Feb-11	-29.33300	30.11276	1049	F	VB_C
7	10-Feb-11	-29.34202	30.09863	1082	C	VB_C
8	10-Feb-11	-29.35977	30.09067	1313	C	VB_C
9	10-Feb-11	-29.44457	29.54988	956	B	Seep
10	10-Feb-11	-29.50428	29.45892	1460	C	FP
11	10-Feb-11	-29.53403	29.42289	1395	F	VB_NC
12	10-Feb-11	-29.55035	29.42477	1130	C	FP
13	10-Feb-11	-29.57229	29.42666	1100	F	Seep
14	10-Feb-11	-29.58191	29.42736	1176	D	Seep
15	10-Feb-11	-30.00073	29.43847	994	D	VB_C
16	10-Feb-11	-30.00724	29.43803	973	C	FP
17	10-Feb-11	-30.02205	29.37632	1397	C	FP
18	10-Feb-11	-29.49855	29.37580	1487	D	VB_C
19	10-Feb-11	-29.49307	29.37683	1459	B	Seep/ VB_C
20	10-Feb-11	-29.48082	29.37468	1460	B	FP
21	10-Feb-11	-29.48362	29.37398	1458	B	VB_NC
22	10-Feb-11	-29.48497	29.37516	1459	A	VB_C
23	10-Feb-11	-29.49028	29.32810	1500	B	VB_NC
24	10-Feb-11	-29.50746	29.30338	1460	E	VB_C
25	10-Feb-11	-29.52374	29.29424	1533	B	Seep
26	11-Feb-11	-29.43876	29.31116	1523	C	VB_NC
27	11-Feb-11	-29.43311	29.31264	1520	B	FP
28	11-Feb-11	-29.42886	29.30925	1560	C	VB_NC
29	11-Feb-11	-29.42136	29.29694	1574	E	VB_NC
30	11-Feb-11	-29.42082	29.29629	1573	C	VB
31	11-Feb-11	-29.41745	29.29607	1561	E	VB_NC
32	11-Feb-11	-29.41132	29.29650	1561	D	VB_NC
33	11-Feb-11	-29.42029	29.31548	1525	D	VB_C
34	11-Feb-11	-29.41216	29.31614	1517	C	VB_C
35	11-Feb-11	-29.41010	29.31664	1534	A	Seep
36	11-Feb-11	-29.40812	29.31830	1556	A	Seep
37	11-Feb-11	-29.39628	29.32245	1481	B	VB_C
38	11-Feb-11	-29.39540	29.32385	1470	E	VB_C
39	11-Feb-11	-29.38894	29.32839	1470	E	VB_C
40	11-Feb-11	-29.38059	29.32541	1492	B	Seep
41	11-Feb-11	-29.37794	29.32514	1462	E	VB_C
42	11-Feb-11	-29.37737	29.32759	1431	E	VB_NC
43	11-Feb-11	-29.37030	29.33431	1356	C	VB_C
44	11-Feb-11	-29.35731	29.34717	1307	D	Seep
45	11-Feb-11	-29.34533	29.35976	1491	B	VB_NC
46	11-Feb-11	-29.34064	29.35714	1462	F	VB_C
47	11-Feb-11	-29.33024	29.36294	1306	E	VB_NC
48	11-Feb-11	-29.31913	29.36488	1293	E	VB_C
49	11-Feb-11	-29.31286	29.39614	1674	B	Depression
50	11-Feb-11	-29.31518	29.41574	1824	A	Seep

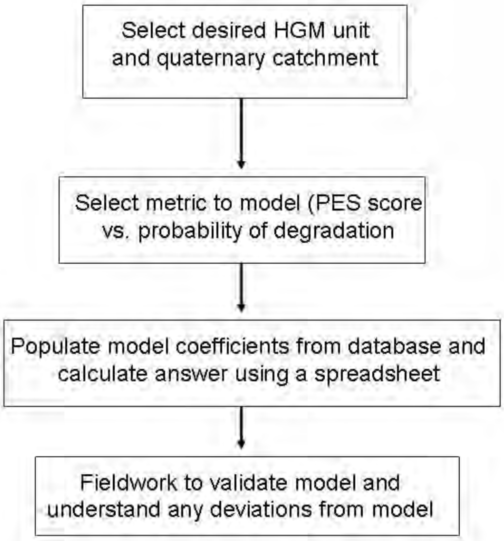
APPENDIX II

Land use codes for the 2005 land cover image in KwaZulu-Natal

Code	Land cover	Model class
1	water natural (NEW)	N/A
2	plantation	Forestry
3	plantation clearfelled	Forestry
4	wetlands	N/A
5	wetlands-mangrove	N/A
6	permanent orchards (banana, citrus)	Commercial Agriculture
7	permanent orchards (cashew) dryland	Commercial Agriculture
8	permanent pineapples dryland	Commercial Agriculture
9	sugarcane – commercial	Commercial Agriculture
10	sugarcane – emerging farmer	N/A
11	mines and quarries	N/A
12	built up dense settlement	N/A
13	golf courses	N/A
14	low density settlement	N/A
15	subsistence (rural)	N/A
16	annual commercial crops dryland	Commercial Agriculture
17	annual commercial crops irrigated	Commercial Agriculture
18	forest	Natural
19	dense bush (70-100 cc)	Natural
20	bushland (< 70cc)	Natural
21	woodland	Natural
22	grassland / bush clumps mix	Natural
23	grassland	Natural
24	bare sand	N/A
25	degraded forest	N/A
26	degraded bushland (all types)	N/A
27	degraded grassland	N/A
28	old cultivated fields – grassland	N/A
29	old cultivated fields – bushland	N/A
30	smallholdings – grassland	N/A
31	erosion	Erosion
32	bare rock	N/A
33	alpine grass-heath	Natural
34	KZN national roads	N/A
35	KZN main & district roads	N/A
36	water dams (NEW)	N/A
37	water estuarine (NEW)	N/A
38	water sea (NEW)	N/A
39	bare sand coastal (NEW)	N/A
40	forest glade	Natural
41	Outside KZN boundary	N/A
42	KZN railways	N/A
43	Airfields	N/A

APPENDIX III Guidelines for using the Wetland Condition Models

The following flowchart represents the steps required to calculate either the probability of wetland degradation or a PES score at a quaternary catchment scale for a chosen HGM unit:



Worked example

In this example, the probability of degradation for a channelled valley-bottom and a seep HGM unit both occurring in quaternary catchment U20B has been calculated. More than half of this quaternary catchment, which is relatively high (modal altitude = 1400 m amsl), is natural, although the number of dams is relatively high (242). Model coefficients are given, which have been multiplied with the model values to give the equation terms. These coefficients are summed and a probability is calculated using logistic regression equations. In terms of the model, a channelled valley bottom HGM unit in this catchment, is twice as likely to be degraded than a seep, with a probability of degradation of 60% (Table A1).

Table A1 Calculation of probability of degradation for channelled valley bottom and seep HGM units for quaternary catchment U20B.

Variable	Variable name	Coefficient		Values (U20B)	Product	
		VB-C	**Seep**		**VB-C**	**Seep**
Constant		10.859	2.728		10.859	2.728
x6	Altitude (m)		-0.003	1400		-3.724
x7	Plantation (%)	-0.221		15.33	-3.388	
x13	Natural (%)	-0.200		55.77	-11.154	
x14	Degraded (%)	-0.157		0.55	-0.086	
x16	No. dams	-0.008		242	-1.936	
x17	Basin length	0.176		28.25	4.972	
x19	Relief ratio	38.360		0.03	1.151	
Coefficient sum					0.417	-0.996
Probability					**0.603**	**0.270**

Using the seep HGM unit as an example of how Equation 1 was applied to calculate probability of degradation, the following model substation steps were followed:

$\alpha+\beta x$: 2.728 - (0.003*1400)

= - 0.996

$$p = \frac{e^{-0.996}}{1+e^{-0.996}}$$

p = 0.27