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Use of GIS and remotely sensed data for *a priori* identification of reference areas for Great Lakes coastal ecosystems

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Identification of reference conditions for ecological assessments of coastal ecosystems poses a challenging problem in highly modified landscapes. A method is described for characterizing disturbance in coastal ecosystems using remotely sensed land classification and other publicly available GIS data. Within ecoregions bordering the US Great Lakes coast, aquatic habitats bordering the shoreline were classified into five ecological types: high-energy shoreline, embayments, open-coast, river-influenced and protected wetlands. Degree of anthropogenic disturbance in contributing areas to these ecosystems was assessed using a watershed approach for wetland types or a moving window approach for high-energy shorelines. Anthropogenic stress variables included proportions of agricultural or residential land use, information on population and road density, and distance from the nearest point source. Polygons (wetlands) or pixels (high-energy shoreline) were categorized as 'reference' if the magnitude of the most severe stressor, based on its cumulative frequency distribution within that ecoregion, placed it within the lowest 20th percentile. For shorelines, adjacent 'reference' pixels were agglomerated into polygons and a final ranking of polygons containing at least 2 km of shoreline was used to identify candidate reference areas. A subset of these sites is currently being sampled for fish, macroinvertebrates and physical habitat attributes. This *a priori* approach to reference area identification will allow managers to identify biological correlates of reference conditions, providing a benchmark for bioassessment and restoration efforts in coastal regions.

1. Introduction

As biomonitoring and biocriteria have become more important to attaining the goals of the US Clean Water Act, there is a need for objective and scientifically-defensible methods to classify aquatic resources and define reference biotic conditions for unique classification units. The concept of 'reference' is integral to biomonitoring efforts and the development of biocriteria (Karr 1991). The reference condition provides a benchmark for gauging the degradation or improvement of aquatic ecosystem health and quantifying the effects of specific management actions (Hughes 1995, Omernik 1995, Yoder and Rankin 1995). Historically, this

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benchmark was based on site-specific ‘control’ locations or upstream–downstream comparisons.

More recently, the concept of ‘regional reference conditions’, which are tailored to specific climatic and physiographic localities, has been advocated (Hughes *et al.* 1986) and refined (e.g. Barbour *et al.* 1992, 1996, Reynoldson *et al.* 1995, 1997). The regional reference condition is determined from a defined population of reference sites and incorporates and describes the range of natural variability of environmental conditions within a region at a particular point in time. As such, a reference condition determined from a population of reference sites strengthens a biomonitoring programme by establishing a benchmark (both a value and some measure of variability) that is indexed to regional factors that reflect the best available abiotic conditions (e.g. water quality, habitat structure) at the time reference sites were identified. An important aspect of the regional reference condition is that its biota represent attainable, current biological conditions, which may or may not reflect historical biological conditions (Hughes and Larsen 1988, Hughes 1995, Omernik 1995, Yoder and Rankin 1995). This paper defines the reference condition as the environmental condition that exists in ecosystems that are the least impacted by anthropogenic stressors.

Inherent in the use of regional reference conditions is the need for a classification system that stratifies aquatic resources over large geographical areas. Gradients of geomorphic, climatic and anthropogenic factors that affect aquatic ecosystems exist across these landscapes. Ideally, a classification system will account for the differential effects of non-anthropogenic factors (i.e. climate and physiography) and partition aquatic resources into units that respond similarly to anthropogenic stress and management activities.

Programmes based on empirical classifications of macroinvertebrate community data have been developed in the UK (Wright *et al.* 1984, Wright 1995), Australia (Marchant *et al.* 1997, 1999, Parsons and Norris 1996, Smith *et al.* 1999, Turak *et al.* 1999) and Canada (Reynoldson *et al.* 1995, 1997). In biomonitoring systems such as RIVPAC-S, BEAST, AusRivAS, macroinvertebrate community data from reference sites are classified using ordination or clustering techniques (e.g. Wright *et al.* 1984, Reynoldson *et al.* 1997, Smith *et al.* 1999). Predictive models based on the physical attributes of reference sites are used to select an assemblage of biota associated with specific reference conditions, which may subsequently be compared to the biota of test sites (e.g. Wright *et al.* 1984, Reynoldson *et al.* 1997, Smith *et al.* 1999). This type of classification system does not make any assumptions about the nature of community composition based on geographical location (Reynoldson *et al.* 1997). However, these approaches use an *a priori* classification to initially identify reference conditions.

Currently, most federal and state biomonitoring programmes in the USA use *a priori* hierarchical systems to classify aquatic resources. Ecosystem structure and function is hierarchically controlled by multiple factors operating across time and space. These controlling factors (e.g. climate, landform and potential vegetation) have formed the underpinnings of several ecological land classification systems currently being used by resource managers in federal and state agencies (e.g. USDA Forest Service, Minnesota Department of Natural Resources). Over the past decade there has been a growing recognition that unique biotic communities, including vegetation (e.g. Nature Conservancy 1994), fish (e.g. Hughes *et al.* 1987) and macroinvertebrates (Ohio EPA 1987, Yoder 1989) exhibit considerable concordance with ecological regions defined by climate and physiographic factors. Selection of stream reference conditions within physiographically distinct regions must further consider natural differences in gradient, water quality and substrate composition in a classification scheme (Hughes *et al*. 1994). Ecological subregions or drainage units can therefore potentially discriminate distinct communities more precisely than ecoregions alone (Hughes *et al*. 1994), but the optimal scale for stratifying the landscape is currently unknown. The development of biological criteria based on regional reference conditions defined by ecoregions (*sensu* Omernik 1987) has been implemented successfully for streams in Ohio (Ohio EPA 1987, 1989) and for lakes in Minnesota (Heiskary *et al*. 1987a, 1987b).

There has been little effort, however, devoted to the development of reference conditions for coastal ecosystems and, in particular, the coastal ecosystems of the Great Lakes. The 2003 State of the Great Lakes Report (SOLEC 2003) described the status of the chemical, physical and biological integrity of the Great Lakes as 'mixed', with positive signs in the success of lake trout and bald eagle populations, and negative signs in unacceptably high levels of phosphorus, air toxins and non-native species (SOLEC 2003). The rapid pace of land-use change in coastal regions places increased pressure on local watersheds throughout the basin. These pressures are particularly acute at the coastal margins, which are biologically-rich transition zones subject to both landward and lakeward influences. For this reason, there are a number of US and Canadian initiatives to develop ecological indicators for coastal and aquatic ecosystems of the Great Lakes; these include the ongoing SOLEC process (SOLEC 2003), which focuses predominantly on the lakes themselves, and the more recent Great Lakes Environmental Indicators (GLEI) initiative (Danz *et al*. 2005), which addresses ecological integrity of coastal ecosystems of the US Great Lakes.

Reference conditions specific to coastal ecosystems are essential for developing and interpreting biocriteria or ecological indicators (Niemi *et al*. 2004). To this end, a quantitative method has been developed for selecting coastal reference areas using readily available GIS and remote sensing data. This work is part of a larger effort to characterize statistical properties of key ecosystem response variables such as fish and macroinvertebrate populations and communities under reference conditions, and to address scale and classification issues related to reference condition development. The intent was to develop and evaluate an objective method for identifying sites that have been disturbed minimally by the dominant anthropogenic stresses within specific ecoregions, based on public Geographical Information System (GIS) and remote sensing data sources. This contribution describes the method and resulting classification of reference conditions for hydrogeomorphically-defined coastal ecosystems for the US Great Lakes.

2. Study area

The study area for this project is the US Great Lakes coastline, an 8000 km stretch of coastline extending from Grand Portage, MN to Watertown, NY. This area spans numerous regional transitions in climate, physiography and, consequently, land use. To reduce variability in predictor and response variables due to ecoregional differences, the coastline was stratified by Level III ecoregions *sensu* Omernik (1987) (figure 1). The six ecoregions adjacent to the coast were analysed independently for this study: the Northern Superior Uplands (NSU), Southern Superior Uplands (SSU), the Northern Great Lakes (NGL), the South Central

Great Lakes (SCG), the Southwestern Great Lakes Moraines (SGL) and the Erie and Ontario Lake Plains (EOL; figure 1).

In addition to the ecoregional classification, coastline aquatic habitats were stratified into five hydrogeomorphic types: high-energy shoreline (HE), embayments, and three wetland categories – open-coast wetlands (OCW), river-influenced wetlands (RIW) and protected wetlands (PW). High-energy shorelines consist of reaches that lack continuous stands of emergent vegetation and are not protected from wind and wave action by embayments or other coastline features. Embayments are defined as bays whose area is $>1\,km^2$, where the mouth-to-apex distance of the embayment exceeds the distance across the mouth, and which contain fewer than two sub-embayments. Open-coast wetlands are primarily impacted by lakeward influences, such as wave action, longshore currents and ice push effects (Keough *et al.* 1999). River-influenced wetlands occur along river mouths and are predominantly influenced by upstream factors and lake waters through seiche effects. Protected wetlands occur behind a coastal barrier, such as dunes or other upland features (Keough *et al.* 1999).

3. Methods

3.1 *Spatial data selection*

Several publicly available point, polygon and raster-based spatial datasets were selected to quantify anthropogenic stress (table 1). Point-source pollution data were

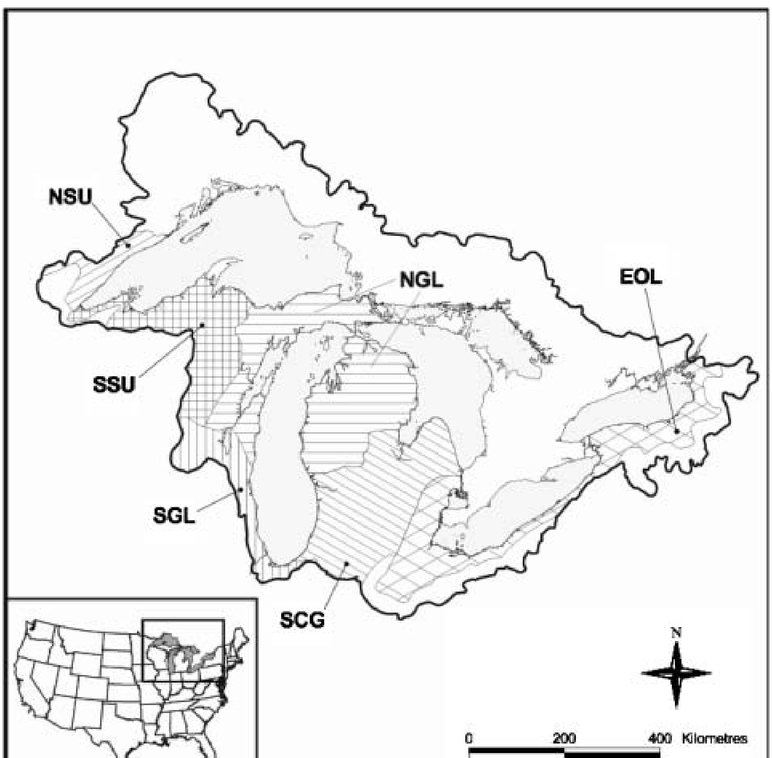

Figure 1. Level III ecoregions (Omernik 1987) that intersect the US Great Lakes coastline.

derived from three sources maintained by the US Environmental Protection Agency (EPA). The National Pollutant Discharge Elimination System (NPDES) permit database identifies point-source locations of pollutants from industrial, municipal or other facilities requiring a discharge permit. The Toxic Release Inventory (TRI) is a point-source database containing information on known toxic chemical releases and other waste management reported by certain industry groups and federal facilities (US EPA 2004). The International Joint Commission's Area Of Concern (AOC) dataset lists 26 US, 12 Canadian and five jointly managed sites that exhibit significant impairments to beneficial use and require remedial action plans.

Land use was quantified using the National Land Cover Dataset (NLCD) created by the US Geological Survey (USGS) (Vogelmann *et al.* 2001). NLCD data are derived from Landsat Thematic Mapper (TM) imagery, with 30 m pixel size. The Anderson Level II land-cover classes present in the NLCD database were summarized to separate highly-modified land-cover types (commercial/residential and agricultural) from those more characteristic of the dominant pre-settlement land cover for the region (primarily forest or wetland). Population densities (persons km^{-2}) were obtained from the Census 2000 Blockdata summary file and gridded to match the NLCD land-cover pixels and summarized on a persons km^{-2} basis. Roads were extracted from the US Census TIGER Line files (US Census Bureau 2002). Road length was summarized and road density (road length (km)/area (km^2)) calculated for watersheds and the 'moving window' (see below). The measures used here are commonly employed in ecological risk assessment; Diamond and Serveiss (2001) used a similar set of land-use and point-source data to assess relationships between watershed-scale factors and fish index of biotic integrity and mussel species diversity in south-western Virginia, USA.

Numerous data sources were used to quantify the distribution of hydrogeomorphic units within ecoregions. Base data consisted of digital raster graphics (DRG; rectified topographic maps), digital orthoquad photographs (DOQs) and digital elevation models (DEMs). To help identify coastal wetlands, a DEM was used to identify areas 0–2.5 m above the 10-year mean summer lake level (US Army Corps of Engineers data). The National Wetland Inventories from New York, Pennsylvania, Michigan and Minnesota and state-level inventories from Wisconsin and Ohio were used to identify mapped wetlands. These data were used to classify

Table 1. Anthropogenic stressor datasets and characteristics.

Dataset	Source and attributes	Summarization methods
Land use/land cover	USGS National Land Cover Dataset; 30 m pixel (Vogelmann *et al.* 2001)	Moving window, Watershed summary
Population density	US Census data	Block summaries converted to raster grids
Point-source discharge	NPDES permits (EPA); Areas of concern (EPA); Toxic Release Inventory (US EPA 2004)); Electric power plants of the US (USGS 1997); Active mines and mineral plants (USGS 1998)	Euclidian distance to point source from each coastline pixel
Road density	USGS Tiger Data	
Shoreline hardening	Medium resolution vector shoreline data (NOAA 1997)	% hardened shoreline

wetlands as open-coast, river-influenced or protected. The classification was verified by visual inspection of aerial photographs.

Embayments were identified by scanning topographic maps of the coastline, using the rules defined in the previous section. All areas that were not designated as wetlands or embayments were classified as high-energy shoreline. Because only one ecosection had enough embayments to meet the minimum sample size criterion ($n=30$), stressors for embayments were not analysed.

3.2 *Moving window analyses*

Because the high-energy shorelines did not have discrete topographically-defined drainage boundaries (as did the wetland systems), a moving window (spatially-defined moving sum or average) approach was used to summarize population, point-source and land-cover data for these coastline segments. A 33×33, 30 m pixel window (approximately 1 km^2) was centred over each coastline pixel, and the ArcGrid FocalSum command (ESRI 2002) was used to summarize stressor attributes within the window and place the summary value in the target pixel.

Agricultural land cover and residential/commercial land cover were extracted from the NLCD data as independent map layers. Pixels were coded as either agriculture or non-agriculture or as residential/commercial or non-residential/commercial. The FocalSum command was then used to sum the number of pixels coded for each land-cover type within the moving window. For point-source data, a continuous surface of distance (Euclidean) to the nearest NPDES, TRI, mine, powerplant, or AOC location was generated. While the specific effects of these diverse point sources are quite variable, it holds that locations furthest from point sources are likely reference area candidates. The shortest distance from each coastal pixel to a point source was summarized using the moving window. Similarly, the population and road densities were calculated for each window of the 30 m grid.

Stressors were summarized for each pixel using the methods described below (see Section 3.4) and pixels categorized as reference or non-reference. Individual pixels identified as potential reference sites were concatenated into polygons, and polygons containing 2 km or more of shoreline were included in the pool of candidate reference sites.

3.3 *Watershed-scale analyses*

Topographic contributing areas were calculated for each of the wetland polygons from DEMs using ArcInfo's WATERSHED command (ESRI 2000). Land cover was summarized as a percent of each contributing area. Population, road length and number of point sources were summarized as densities within each contributing area, and summary values assigned to the wetland polygons.

3.4 *Scoring anthropogenic stress and identification of reference polygons*

Potential reference condition polygons were identified based on the distributions of the stressor variables within individual ecoregions. Following the moving window or watershed analyses, each pixel or polygon was characterized by five stressor axes – agricultural land use (AG – %), urban land use (URBAN – %), distance from a point source (PSOURCE – km), road density (RDENS – km km^{-2}) and population density (POPDENS – individuals km^{-2}). Values for each stressor variable were relativized within individual ecoregions by dividing individual values by the maximum value observed for that stressor at any polygon within that classification unit (ecosection $\times$ hydrogeomorphic unit). This provided a relative score, scaled between 0 and 1, for each watershed or polygon for each stressor axis.

Each watershed or high-energy polygon was then assigned a single stressor value, defined as the *maximum relative score* (MAXREL) across any of the five axes. For example, a pixel with relativized values of AG=0.18, URBAN=0.22, POPDENS=0.12, RDENS=0.34 and PSOURCE=0.17 would be assigned a MAXREL score of 0.34. The effect of this approach was to characterize the degree of anthropogenic stress on each polygon as a function of the maximum stress across the five stressor axes, based on the distribution of that stressor within that ecoregion. This method was developed based on the assumption that, to best identify reference conditions, all stressors should be held to their lowest levels based on their cumulative distributions within the ecoregion.

To identify the least anthropogenically disturbed polygons, polygons were sorted by their MAXREL value. Those with the lowest MAXREL values are the least disturbed, and potential reference condition candidates. The application of a reference condition approach requires a sufficient sample size from which a suitable set of biological indices can be derived. A suitable minimal sample size is a function of regional variability and the desired level of detectable change (Hughes 1995). It was operationally determined that a minimum sample size of six reference sites per ecoregion was required in order to assess biological conditions. The 20th percentile of MAXREL values was the lowest boundary that permitted us to meet this criterion for the six Great Lakes ecoregions; consequently, candidate reference sites were defined as those falling within the least disturbed 20% of the distribution. This fits within the range of percentages used in other studies. Reference condition sites for warm-water wadeable stream habitat in the Huron Erie Lake plain region of Ohio were delineated by selecting the 10% of locations thought to be least disturbed in this highly agricultural and extensively modified landscape (Yoder and Rankin 1995). Others have used a cut-off of 25% for defining reference conditions (Davis and Simon 1995).

As a final step for the high-energy shorelines, candidate reference polygons were intersected with the medium resolution vector shoreline data (National Oceanic & Atmospheric Administration (NOAA) 1997), which contains information on the extent and type of shoreline hardening (artificial structures used to prevent erosion). This allowed us to inversely rank candidate reference polygons by the proportion of shoreline in each polygon that was not protected by man-made structures. Reference sites were chosen to reflect the least amount of shoreline hardening present.

4. Results

The numbers of wetlands or embayments and amount of shoreline varied widely across ecoregions (table 2). Half of all wetlands were located in the NGL, which occupies the northern lower and upper peninsulas of Michigan, as well as the Green Bay area of northwestern Wisconsin (table 2, figure 1). Open-coast wetlands were the most abundant wetland type (43%), with river-influenced and protected wetlands accounting for 30% and 27%, respectively. Embayments and open-coast wetlands were absent from the Northern Superior Uplands and Southern Great Lakes ecoregions. Protected wetlands were uncommon in the NSU and SCG ecoregions, and river-influenced wetlands were rare in the Southern Great Lakes.

Because the intent was to identify potential reference locations for sampling, only combinations of ecoregions and ecosystem types that were represented by 30 or more polygons were chosen. Further discussions focus on these well-represented types. As noted above, the distribution of the MAXREL scores were assessed and those sites in the lowest quintile (20%) of the distribution were selected as candidate reference areas. The resulting spatial distribution of these candidate reference areas along the US Great Lakes coast is presented in figure 2; sites are stratified by ecoregion and ecosystem type.

4.1 *Wetlands*

As expected, there was a high degree of variability in most stressor variables. Agricultural land use within wetland watersheds typically ranged from a few percent to 70% or greater (figure 3(*a*)). There were differences among ecoregions, however, with the EOL and the SGC having the highest median values, ranging from 41% to 71% agriculture; whereas the other ecoregions had medians of 15% or less (table 3). The distribution of urban land was highly skewed toward lower values, with 83% of all watersheds having less than 5% urban land cover. A small number of watersheds had significant urban areas; 4% of the wetland watersheds had 20% or more of their area in urban development (figure 3(*b*)).

Road density (RDENS) is a commonly used environmental indicator (Mrazik 1999, SOLEC 2003). Across ecoregions, median road densities within watersheds varied within a narrow range from 1.03 km km^{-2} to 1.86 km km^{-2} (table 3). Variability within ecoregions was high however, with 5% of the sites exceeding 4.0 km km^{-2} (figure 3(*c*)). Population density was also quite variable across the Great Lakes coast – median population values ranged from a relatively sparse 33 persons km^{-2} in the SSU to 450 persons km^{-2} in the EOL (table 3). Typical city populations in the Great Lakes range from 400–500 persons km^{-2} (Detroit, Cleveland) up to 1200 persons km^{-2} (Chicago; Yaks 2000). The numbers of point sources within a watershed were generally low, less than 0.5 km^{-2}. River-influenced wetlands of the EOL stood out, however, with several sites exceeding 10 point sources km^{-2} (table 3).

Across all wetland sites (open-coast, protected and river-influenced), AG and RDENS were the dominant stress factors; typically 77% or more of the sites within an ecoregion were affected primarily by one of these two factors. The dominant

Table 2. Amount of shoreline reach or number of wetlands or embayments by hydrogeomorphic type and ecosection.

Ecosection	High-energy shoreline (km)	Embayment	River-influenced wetland	Protected wetland	Open-coast wetland
EOL	1613	18	77	45	38
NGL	2687	34	53	95	188
NSU	389	0	16	2	0
SCG	592	2	12	6	33
SGL	520	0	2	10	0
SSU	920	10	39	29	27

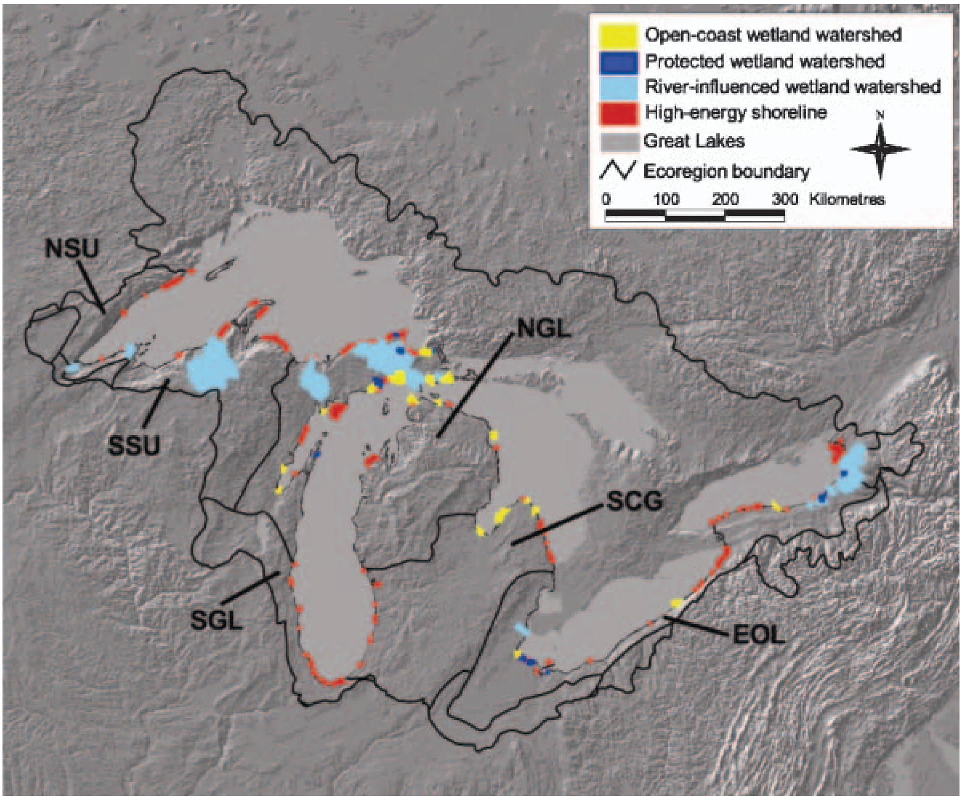

Figure 2. Reference sites for open-coast, river-influenced and protected wetlands and high-energy shorelines for ecoregions of the US Great Lakes coast.

stressor variables were clearly segregated by ecoregion, however, with AG being the dominant factor in all wetland types of the EOL and RDENS dominating wetlands of the NGL ecoregion (figure 4).

Reference conditions in all ecoregions were typically determined by multiple stressors. In the NGL open-coast wetlands, for example, four different stressors (URBAN, RDENS, AG and POPDENS) had the lowest MAXREL values in the lowest 20% of the distribution (figure 5; table 4). Open-coast and protected wetlands in the EOL ecoregion were defined in terms of RDENS, AG and URBAN. The only ecosystem defined in terms of a single stressor was RIW; in the EOL; AG was the defining variable, whereas RDENS was the dominant stressor in the NGL. The density of anthropogenic point sources never occurred as the defining factor in wetland reference areas, although this variable did have maxima further down in the distribution (figure 5). Distance to point source was consistently an important stress factor for high-energy shorelines, as was road density (table 4).

As expected, the median and ranges of the five stressor variables were lower for those sites selected as reference areas. The MAXREL approach, i.e. setting the score for a site based on the highest-ranked stressor, relegated all other stressor variables to lower percentiles of their distribution (table 3). As was observed with the full datasets, there were strong differences among ecoregions with respect to the distributions of stressor variables. Reference wetlands of the EOL, for example, had a much broader range and higher median values for agricultural land use compared

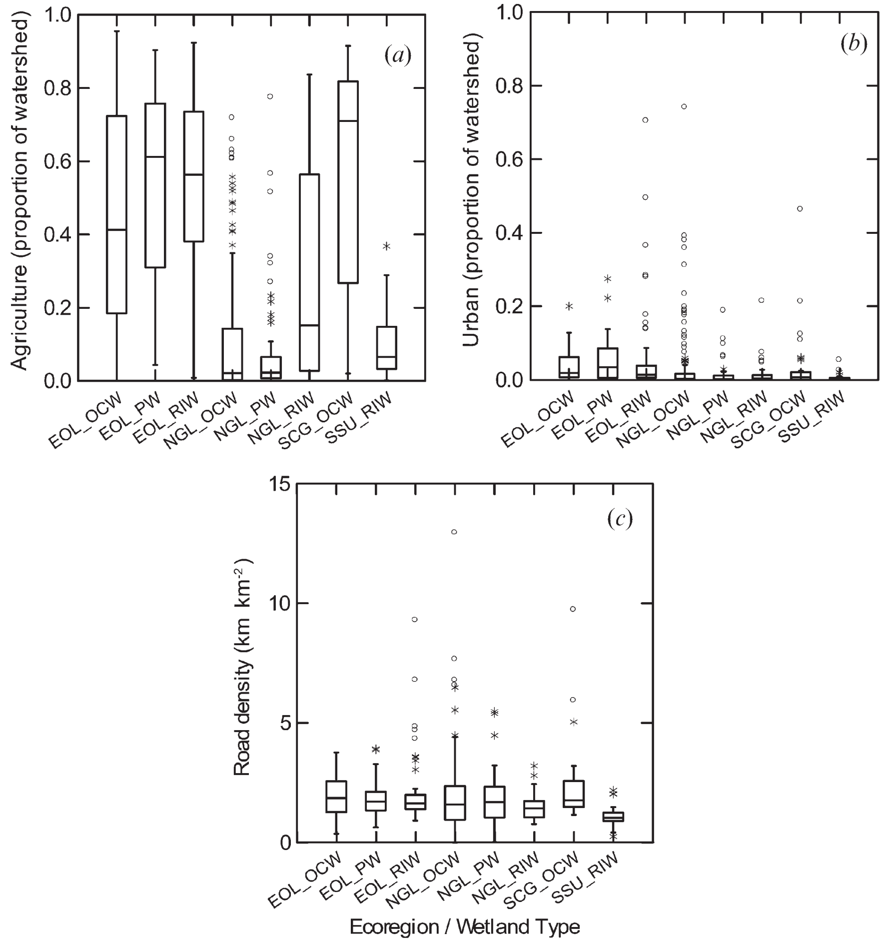

Figure 3. Box plots of (*a*) agricultural and (*b*) urban land use and (*c*) road density across ecosystems and ecosections of the US Great Lakes coastline. Centre line shows the median, box displays the range between the 25th and 75th percentile and 'whiskers' show the range of values within 1.5 × the interquartile range. Ecoregions: EOL, Erie-Ontario Lakeplain; NGL, Northern Great Lakes; SCG, south-central Great Lakes; SSU, south Superior uplands. Wetland types: OCW, open-coast wetland; PW, protected wetlands; RIW, river-influenced wetland.

with wetlands of the NGL (figure 6(*a*)). Similarly, reference sites of river-influenced and open-coast wetlands in the EOL and SCG, respectively, allowed higher population densities in their reference sites than any of the other subsections (figure 6(*b*)). These differences reflect the overall higher levels of disturbance in the south-eastern portion of the Great Lakes basin, compared with the west and north.

4.2 *High-energy shorelines*

High-energy shorelines were assessed somewhat differently than wetlands, in that the stressor data were in the 1 km^2 window surrounding each coastline pixel quantified vs. the generally larger areas that were assessed by delineating contributing watersheds around wetlands. These smaller areas showed a different pattern of stressors compared with wetland ecosystems. In general, shorelines

Table 3. Median and range of anthropogenic stressor variables for wetlands and high energy shorlines across six ecoregions.

All sites

		EOL				NGL				SCG		SSU		NSU	SGL
		OCW	PW	RIW	HE	OCW	PW	RIW	HE	OCW	HE	RIW	HE	HE	HE
Agriculture (%)	Median	41	61	56	7	2	2	15	1	71	4	7	0	1	0
	Range	0–96	4–90	1–92	0–90	0–72	0–77	0–84	0–67	2–91	0–56	0–37	0–36	0–18	0–47
Urban (%)	Median	2	3	1	14	0	0	0	0	1	7	0	0	2	34
	Range	0–20	0–28	0–70	0–95	0–74	0–19	0–21	0–90	0–46	0–77	0–5	0–91	0–88	0–96
Population	Median	178	373	449	117	83	77	108	14	192	0	33	3	11	113
(people/km^{-2})	Range	0–2244	0–4257	58–16867	0–8599	0–13342	0–3015	1–2563	0–2855	0–7907	0–3267	0–819	0–2668	0–3256	0–29396
Road density	Median	1.86	1.71	1.64	2.56	1.59	1.69	1.44	1.92	1.77	2.76	1.03	0.88	3.00	2.02
(km km^{-2})	Range	0.37–3.76	0.63–3.93	0.91–9.29	0.00–16.9	0.00–12.95	0.01–5.48	0.77–3.21	0–12.93	1.16–9.73	0–11.58	0.25–2.18	0–11.04	0.00–13.4	0.00–11.75
NPDES permit density*	Median	0.0	0.0	1.9	1.8	0.0	0.0	0.1	8.6	0.0	2.7	0.0	10.5	4.3	1.0
(# permits km^{-2})	Range	0.0–0.4	0.0–0.4	0.0–24.0	0.0–21.9	0.0–1.2	0.0–0.1	0.0–11.3	0.0–52.5	0.0–0.2	0.03–17.8	0.0–5.7	0.0–42.3	0.03–24.2	0.0–6.7

Reference sites

		EOL				NGL				SCG		SSU		NSU	SGL
		OCW	PW	RIW	HE	OCW	PW	RIW	HE	OCW	HE	RIW	HE	HE	HE
Agriculture (%)	Median	17	29	28	29	0	3	1	1	9	14	5	1	1	0
	Range	0–34	5–41	15–38	1–66	0–3	0–11	0–6	0–34	4–37	1–32	0–9	0–5	0–2	0–11
Urban (%)	Median	6	0	1	4	0	0	0	0	1	5	0	0	0	29
	Range	0–9	0–9	0–14	0–49	0–2	0–0	0–1	0–3	0–11	1–20	0–1	0–3	0–4	2–49
Population	Median	9	166	298	15	15	11	21	1	161	1	14	23	3	80
(people/km^{-2})	Range	0–611	0–345	142–3241	1–275	0–196	0–317	1–39	0–11	0–2870	0–68	0–54	0–5	1–8	0–2462
Road density	Median	0.50	0.78	1.49	2.39	0.45	1.03	0.92	1.23	2.71	3.55	0.72	2.84	2.69	2.84
(km km^{-2})	Range	0.00–1.71	0.63–2.00	0.93–3.43	0.00–6.10	0.00–0.78	0.58–1.34	0.77–0.99	0.00–3.11	2.57–3.20	0.00–5.52	0.25–0.90	0.00–2.73	0.00–5.39	0.00–5.33
NPDES permit density*	Median	0.0	0.0	0.8	5.5	0.0	0.0	0.0	24.7	0.0	7.2	0.0	22.5	12.2	3.0
(# permits km^{-2})	Range	0.0–0.0	0.0–0.0	0.0–13.6	4.6–16.4	0.0–0.0	0.0–0.0	0.0–0.2	18.0–52.0	0.0–0.0	5.3–16.3	0.0–0.2	17.8–39.1	10.5–18.5	2.2–6.0

*High energy is expressed as distance to nearest point source (km)

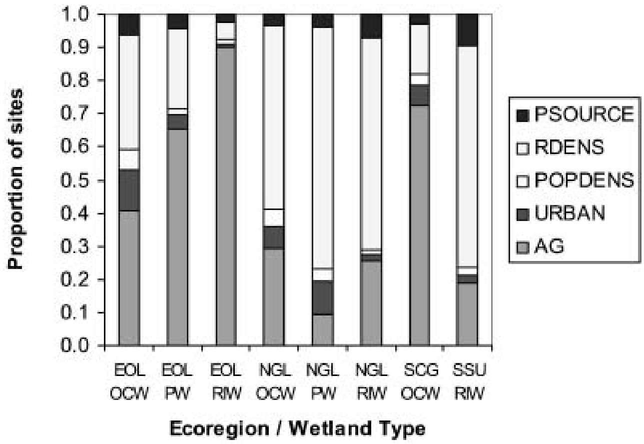

Figure 4. Proportion of wetland sites influenced by their defining stressor variables for ecosystems and ecosections of the US Great Lakes coastline. Ecoregions: EOL, Erie-Ontario lakeplain; NGL, northern Great Lakes; SCG, south central Great Lakes; SSU, south Superior uplands. Wetland types: OCW, open-coast wetland; PW, protected wetlands; RIW, river-influenced wetland.

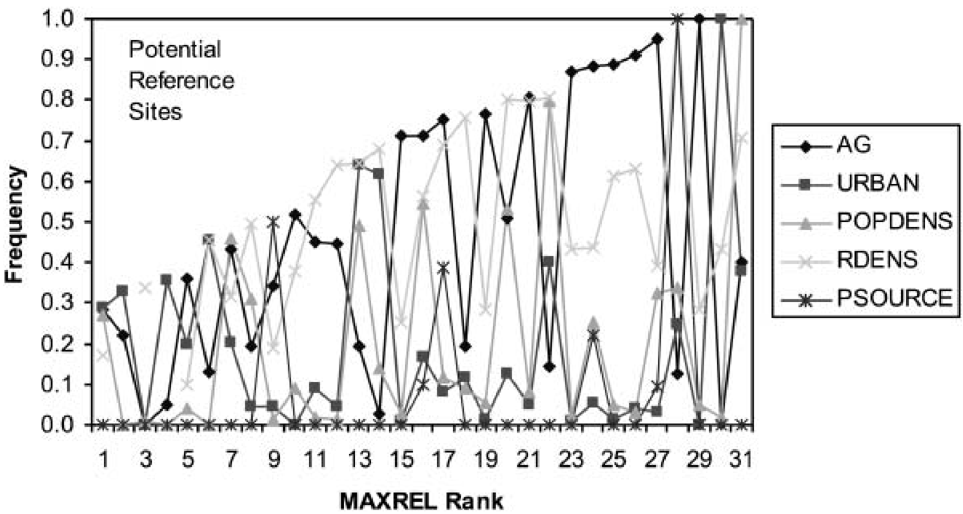

Figure 5. Ranking of EOL open-coast wetland sites by the MAXREL stress indicator; sites to the left of vertical bar are candidate reference sites.

supported less agriculture and more urban land use than was observed for wetlands (table 3). Distance to point sources and road density were the most common stressors impacting high-energy shorelines (table 3). In the NGL and EOL ecoregions, the median and range of road densities were higher along high-energy shorelines than in wetlands, probably due to the smaller polygon size combined with the historic development of major and minor transportation corridors roads along the Great Lakes coastline.

Under reference conditions, there were no strong differences between high-energy shorelines and wetlands with respect to land use or population density. As was

Table 4. Distribution of reference sites by ecosystem type and ecoregion and defining stressor.

		Stressor (identified by MAXREL)				
Ecoregion	Ecosystem	Population density	% Urban	% Ag	Road dens.	Point source
Northern Great Lakes	Open-coast wetland	6	3	3	18	0
	Protected wetland	1	0	0	9	0
	River-inf. wetland	0	0	0	11	0
	High-energy shoreline	1	1	1	10	20
Erie and Ontario Lake Plains	Open-coast wetland	0	3	1	2	0
	Protected wetland	0	1	4	4	0
	River-inf. wetland	0	0	15	0	0
	High-energy shoreline	1	1	5	5	24
South-central Great Lakes	Open-coast wetland	0	0	3	3	0
	High-energy shoreline	1	1	4	6	3
SW Great Lakes Moraines	High-energy shoreline	1	4	1	5	3
S Superior Uplands	River-inf. wetland	0	0	2	6	0
	High-energy shoreline	2	1	1	8	4
N Superior Uplands	High-energy shoreline	1	1	2	2	1

observed in the full dataset, however, road densities were higher along high-energy shorelines than in the wetland types (table 3) and RDENS was frequently a dominant stressor. Point sources were quantified differently for high-energy shorelines (as distance to nearest point source) and are not directly comparable to wetlands.

5. Discussion

A key objective of ecoregional classifications is to partition the landscape into relatively homogeneous classes based on large-scale factors such as climate, regional physiography and soils (Bailey 1987, Niemi *et al.* 2004). These, in turn, exert significant influence over factors important to the health of aquatic ecosystems, such as stream habitat and biota (Richards *et al.* 1996), water chemistry (Johnson *et al.* 1997) and vegetative community composition and functional processes (Host *et al.* 1987, 1988). Clearly, there are strong interactions between physiography and human activities on the landscape; Richards *et al.* (1996), working in streams of central Michigan, detected significant ‘shared variation’ between geology and land use when accounting for variation in stream habitat variables. Thus, to be useful in bioassessment and developing biocriteria, reference conditions must be developed in the context of ecoregional classifications (Barbour *et al.* 1996, Reynoldson *et al.* 1997). Ecological classifications are hierarchical, however, and the optimal scale for identifying and operationally using reference conditions is unknown. The key scale issue in identifying reference conditions is selection of an appropriate classification unit that sufficiently reduces the variability related to physiographic or climatic effects while not being overly specific (i.e. requiring many unique sets of reference conditions). The strong differences observed in anthropogenic stressors among the six ecoregions of the Great Lakes imply that the Omernik’s Level III classification scale provides a meaningful initial stratification for identifying reference conditions and developing bioassessment protocols.

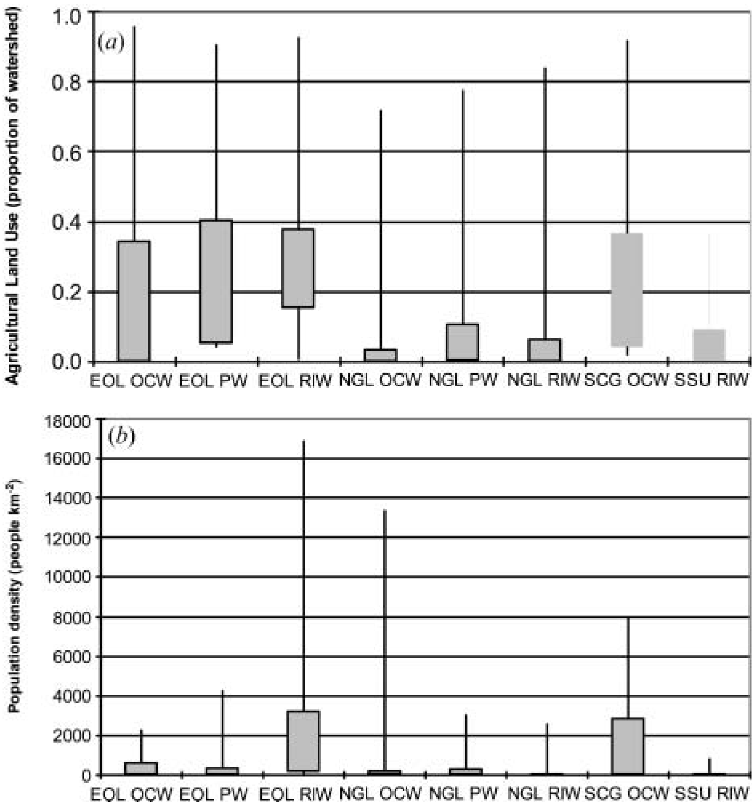

Figure 6. Range of (*a*) agricultural land use and (*b*) population density for reference sites and the full dataset. Vertical lines show range for full dataset, rectangles depict range for reference sites.

Within the EOL and NGL ecoregions, wetland types showed a high degree of overlap in the ranges of stressor variables, both in the full datasets and in the reference set. Median values were also quite similar among wetland types, suggesting that at the scale of contributing watersheds, land use, population and point-source distributions are relatively homogeneous. However, the mechanisms by which stresses are delivered differ among wetland types, with river and inland land use affecting river-influenced wetlands by definition, open waters of the lake predominantly affecting open-coast wetlands and protected wetlands occupying a more intermediate position. Given these regulatory differences, it appears reasonable to assess these ecosystems independently, and subsequently determine from the behaviour of response variables if these systems might ultimately be combined for bioassessment purposes.

The approach used here differs from other reference area work in the sequence of analyses and in the use of independent datasets. Bioassessment of the Oregon coast and Lower Columbia River, for example, used regional professionals in fisheries, hydrology geology and entomology to identify candidate sites (Mrazik 1999). This was followed by an analysis of road density, used as an index of human disturbance, to numerically rank sites. Candidate sites in the least road-developed areas were then further classified based on region, elevation and stream size. This was followed by a field evaluation of candidate reference sites and ultimately the collection of response-variable data at sites that met reference criteria (Mrazik 1999). The work here is similar in the use of a hierarchical stratification of sites based first on ecoregion and secondarily on physical factors (elevation and size in Oregon, hydrogeomorphic types in the Great Lakes). It differs in that remote sensing and GIS data were used in place of expert opinion, and in the use of multiple stressor factors rather than a hierarchy of stress variables.

Barbour *et al.* (1996) found that aggregated subregions proved to be a better discriminator of macroinvertebrate-based metrics in Florida wetlands than individual subregions or chemically-defined stream types. In this case, reference sites were selected by the investigators as 'minimally disturbed streams with small catchments'. To account for the natural variability among reference sites, they chose the lower quartile of each metric's distribution as a threshold: sites with metric scores above this threshold were considered to be unimpaired. They recommended, however, that a broader, multi-state approach to expanding the reference site database would strengthen the site classification process. The assessment of the 8000 km US Great Lakes coastline gives the advantage of sampling across a large range of climatic and physiographic conditions, as well as the broad gradient of environmental stresses imposed. Identifying reference sites for each ecoregion resulted in a well-dispersed selection of sites (figure 2), which accounts for constraints imposed by climate and regional physiography, along with the concomitant differences in land use and human settlement across a broad ecological gradient.

The ability to use remotely sensed and other GIS databases in an *a priori* fashion provides an objective means for reference area selection. Although many groups use professional opinion to select reference conditions, Fore (2003) found that 73% of hand-picked reference sites in the mid-Atlantic region did not meet independently-established criteria for reference condition. One reason for the poor performance of the professional judgement is that the influence of landscape-scale stressors, such as the presence of row-crop agriculture within the watershed, may not be apparent in the field evaluation of individual sites. The use of remotely sensed and other spatial data provides a means of quantifying anthropogenic stressors that operate at broader spatial scales. A final field evaluation of the biological characteristics of reference sites, however, is a critical step to identify local disturbances below the resolution of spatial datasets.

The sites identified in this analysis will be useful in characterizing statistical properties of fish, macroinvertebrate, water quality and other ecosystem response variables collected from reference sites, and in comparison with metric data from impaired coastal ecosystems collected as part of the Great Lakes Environmental Indicators project (Danz *et al.* in press). This larger analysis will allow us to address questions of scale and ecological thresholds in developing effective biomonitoring and bioassessment protocols for large geographical regions.

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