Great Lakes Coastal Wetland Communities: Vulnerabilities to Climate Change and Response to Adaptation Strategies

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EXECUTIVE SUMMARY

Great Lakes coastal wetlands are located along the dynamic land-water interface where wetland ecosystems are continually responding to changing water levels, and are reliant on this hydrologic variability to maintain diversity and functioning. Human-caused climate change is expected to affect the hydrology of the Great Lakes as warming temperatures, increasing evaporation, and changing precipitation and snowcover patterns are likely to result in long-term reductions in water levels. Projected decreases in water levels could alter the current distribution and abundance of coastal wetland communities.

A collaborative research project was undertaken to assess the vulnerability of selected wetlands on Lake Ontario (Presqu'ile Bay, Hay Bay, Lynde Creek, and South Bay wetlands), Lake Erie (Long Point, Turkey Point, Dunnville, and Rondeau wetlands), and Lake St. Clair (Mitchell's Bay) to climate change. The integrated assessment utilized literature reviews, field surveys, stakeholder engagement, and modelling to explore:

- responses of Great Lakes coastal wetland communities (wetland vegetation and associated wetland-dependent birds and fishes) to historical and projected water level changes, and
- human-directed adaptations to changing water levels – infrastructure (lake regulation and dyking) and land use policy – to maintain ecosystem functions and values.

Vulnerability indices were developed to assess the current sensitivity of Great Lakes coastal wetland vegetation and wetland-dependent breeding birds to hydrologic changes, and fishes to hydrologic and thermal changes. Scores for vulnerability factors were used to categorize species into low, moderate, and high risk groups. Wetland plant species with limited drought-tolerance and modes of colonization were identified as the most vulnerable. As a result, diversity, particularly among submerged aquatic and floating leaved plants, could suffer. Plant species identified as highly vulnerable included wild rice (*Zizania palustris*) and Hill's pondweed (*Potamogeton hillii*). Least vulnerable were several invasive species such as purple loosestrife (*Lythrum salicaria*) and common reed (*Phragmites australis*). Obligate wetland breeding bird species with nesting and foraging preferences that require specific hydrologic conditions were identified as most vulnerable with the requirement for prolonged, relatively stable water levels during the breeding season being a key factor (e.g. Forster's Tern (*Sterna forsteri*), Black Tern (*Chlidonias niger*), Pied-billed Grebe (*Podilymbus podiceps*), rails, and bitterns). Many of the high risk bird species are considered at-risk species within the Great Lakes, or have declining population trends, indicating existing stresses that may be exacerbated further by climate change. High-risk lacustrine, native fishes that were most sensitive to coastal changes included cool to warmwater species with limited geographic distributions, spring and shallow-water spawning, and a preference for vegetated habitat in all life stages (e.g. pugnose minnow (*Opsopoedus emiliae)*, spotted gar (*Lepisosteus oculatus*), and muskellunge (*Esox masquinongy*)).

Models were developed to determine the response of vegetation communities, breeding bird abundance, and fish habitat suitability to water level variability. First, spatiotemporal trend analyses in a Geographic Information System (GIS) related the effects of historic water level conditions to the abundance and spatial distribution of wetland plant communities. These analyses contributed to the development of a rule-based model linking vegetation community type with average water depths and antecedent hydrologic conditions (e.g. the duration of previous flooding or dewatering). This model was applied to the wetland case study sites on Lakes Ontario and Erie, and subsequently this vegetation community information was used as input to the bird and fish models.

Bird models were developed from bird survey data on the abundance of breeding bird species in meadow marsh and treed/shrub habitats, and abundance of marsh birds breeding within emergent habitat across a range of water depths. Regression equations or fixed densities of abundance were developed for each wetland nesting guild and were used in combination with estimates of wetland habitat availability to estimate breeding bird responses. The fish habitat availability model (Habitat Alteration Assessment Tool or HAAT) used life stage preferences to estimate habitat suitability values in defined areas for fishes grouped by feeding and thermal preferences. HAAT was used to estimate suitability values for unique combinations of habitat characteristics (depth, submergent and emergent vegetation, and substrate type), and then to estimate the area of suitable habitat in a wetland for six fish guilds with three life stages (spawning, young-of-the-year, and adult).

Four climate change scenarios were selected to represent future extremes in climate ranging from the most warming and wettest conditions (warm & wet) to the most warming and driest conditions (warm & dry), and from the least warming and wettest conditions (not as warm & wet) to the least warming and driest conditions (not as warm & dry). These scenarios were developed from Global Climate Model simulations using the most up-to-date greenhouse gas emission scenarios. For the International Joint Commission Lake Ontario - St. Lawrence River Study, these scenarios were incorporated into hydrologic models to project water level responses; this project used these water level results. Projected annual average water level reductions in Lakes Ontario, Erie, St. Clair, and Michigan-Huron ranged from 15 to 118 cm for the year 2050 across all scenarios, while one scenario for Lake Ontario had an increase (6 cm). The projected 2050 lake level changes for each scenario were applied to observed water level data (base case conditions) and used as input into the vegetation community, bird, and fish models for each wetland. The biotic models were run using both a low and high water level state for initial simulations (and produced the base case for comparisons with climate change scenario outcomes). Recorded historic high and low water level periods (i.e. drought or flooding/high supply periods) were selected to coincide with years where wetland vegetation data existed for validation of the vegetation community model.

All model outputs were used to complete an integrated lower Great Lakes coastal wetland climate change impact assessment. Eight wetlands on Lakes Ontario and Erie were assessed for wetland vegetation community changes, six wetlands for bird responses, and two wetlands for fish habitat responses to extremes in historical and climate change water level scenarios. (Fish modelling was also undertaken for a wetland in Lake St. Clair).

Wetland community modelling indicated that the lower water levels projected under most climate change scenarios will have an impact on the distribution and abundance of wetland vegetation, bird, and fish communities; major shifts in all taxonomic groups are likely with long-term water level declines beginning with vegetation responses. A decrease in water levels favours the expansion of drier vegetation types particularly along the upper margins of the wetland and a reduction in open water area, including submergent vegetation, in most embayments. Hydrogeomorphology plays a significant role in wetland and habitat responses. Based on model results, protected, wetland communities within lacustrine embayment wetlands were the most capable of naturally adapting to lower water levels. Adaptations included the expansion of treed/shrub and meadow marsh vegetation into emergent vegetation areas. The expansion of emergent and submergent vegetation lakeward occurred at lower elevations and relevant water level conditions but other factors affecting growing conditions (i.e. wave and wind protection, substrate types or soils, and slope) were not incorporated. Major shifts in wetland community distribution and abundance were typical in riverine (drowned river-mouth) wetlands due to the topography of the floodplains. Expansion of emergent and submergent vegetation within this wetland type was limited to the river channel and may occur within new deltaic features should shoreline processes allow for creation of these features as water levels decline.

Great Lakes coastal wetland bird and fish communities have the ability to respond to potential changes in vegetation community redistributions although the response was not equitable across the bird and fish guilds. Potential abundance of marsh nesting obligates decreased while abundance of treed/shrub nesting species increased in all scenarios of water level decline. Responses of marsh nesting obligates and meadow marsh nesters, and nursery habitat for selected fish guilds were not consistent, and depended upon the initial high or low water level conditions and the climate change scenario. Guild-specific modelling and species-specific vulnerability assessments indicated that certain bird and fish guilds or species may be more impacted by potential changes in the timing, duration, and depth of flooding within specific vegetation communities. For fishes, the water temperature changes that would occur under a warmer climate also would determine specific responses. Over-water nesting bird species and fish species that required flooded vegetation for reproduction and nursery habitat were most vulnerable. Site-specific responses based on local physical conditions determined the wetland biotic responses which may have benefited some guilds or species while adversely affecting others. This result underscores the need for large-scale assessments to determine net changes in communities at a regional level.

Human adaptations to climate change involving coastal wetlands and lower water levels could take several forms – wetland dyking, large-scale water level regulation, and coastal land-use planning changes. Wetland modelling and stakeholder input for this project indicated that land use planning and policy actions that protect the natural processes which create wetlands and maintain their ability to adapt to varying water level conditions should be a high priority. Therefore, mechanisms are required to incorporate climate change trends and potential impacts information, such as projected changes in wetland distribution and functioning, into policy and planning at various levels of government. No examples were found on current land use planning or policy within the Great Lakes region that utilized human-directed adaptation to climate change to reduce impacts to Great Lakes coastal wetlands or any other natural coastal areas. Ten Planning Criteria and a Coastal Corridor Concept were developed as preliminary ideas proposed for the future protection of coastal areas and these concepts were discussed with stakeholders during the second year of the project. A limited development coastal corridor would help maintain the functioning of natural shoreline processes under a changing climate while also protecting property and potentially enhancing public access at low, long-term costs.

The preliminary evaluation of lake-wide water level regulation and the detailed investigation of wetland dyking as potential adaptation strategies identified benefits for certain coastal wetland biota and impacts on others. Given the costly, labour-intensive, and long-term resource requirements of water level manipulation through structural options (e.g. large-scale dams and control structures, and dyking), a broader understanding of their impacts on wetland functioning is required prior to wider promotion as a climate change adaptation strategy for coastal wetland communities. The comparative analysis of dyked areas demonstrated the potential vulnerability of current infrastructure to water level changes and highlighted the need to consider future water level scenarios in engineering designs. In surveys and break-out sessions, stakeholders generally agreed that wetland dyking as a climate change adaptation should be a secondary measure, initiated after protection of natural compensatory processes and only under special circumstances (e.g. protection of critical habitat for species-at-risk).

Investment in research to further advance the scientific understanding of Great Lakes coastal wetland vulnerabilities to climate change and to fully investigate potential adaptation strategies should include:

- verifying and validating the vulnerabilities and model predictions of the wetland species/guilds identified in this project and determining their water level thresholds, if any;
- understanding the interactive effects of substrate type, and other environmental factors and coastal processes on wetland vegetation (emergent and submergent) colonization and succession as water levels decline;
- undertaking regional fish and bird population assessments for key species that incorporate spatial and temporal considerations not addressed in this evaluation;
- collecting consistent, high quality, and high resolution field measurements of bathymetry, elevation, and substrate types in coastal areas at the land-water interface, allowing for improvements to current digital elevation models, and implementation of standardized data collection approaches across the region which would facilitate large-scale modelling, and basin level assessments;
- identifying important coastal wetlands, transitional areas, and natural processes in need of conservation and protection; and
- interacting with Great Lakes coastal zone stakeholders to maintain a dialogue on potential climate change impacts and adaptation strategies with the goal of incorporating climate change research in policy and planning.

Present-day and long-term Great Lakes conservation policy and planning initiatives need to incorporate current climate change impact assessment knowledge (e.g. changes to water levels and their associated impacts) as well as future research findings in an adaptive management approach. Particularly important is the identification of spatial and temporal thresholds affecting the adaptive capacity and biotic integrity of Great Lakes coastal wetlands.

SOMMAIRE EXECUTIF

Les milieux humides des rives des Grands Lacs, sis à l'interface dynamique entre terre et eau, sont des écosystèmes qui réagissent continuellement aux changements des niveaux d'eau et dépendent de cette variabilité hydrologique pour maintenir leur diversité et leurs processus. On pense que l'hydrologie des Grands Lacs sera affectée par les changements climatiques d'origine humaine, étant donné que le réchauffement des températures, l'accroissement de l'évaporation et la modification des régimes de précipitation et de couverture neigeuse se solderont probablement par des réductions à long terme des niveaux d'eau. Les diminutions projetées des niveaux d'eau pourraient altérer la répartition et l'abondance actuelles des communautés biologiques qu'abritent les milieux humides riverains.

Un projet de recherche coopératif a été réalisée pour évaluer la vulnérabilité à l'égard des changements climatiques de milieux humides choisis du lac Ontario (milieux humides de la baie Presqu'île, de la baie Hay, du ruisseau Lynde et de la baie South), du lac Érié (milieux humides de Long Point, de la pointe Turkey, de Dunnville et de Rondeau) et du lac Sainte-Claire baie (Mitchell's). Pour cette évaluation intégrée, on a effectué des recherches documentaires, des relevés de terrain, des consultations auprès des parties intéressées et des travaux de modélisation afin d'examiner :

- les réponses des communautés biologiques des milieux humides riverains des Grands Lacs (végétation et oiseaux et poissons associés dépendant des milieux humides) aux changements passés et projetés des niveaux d'eau, et
- les adaptations anthropiques aux changements des niveaux d'eau – infrastructure (régulation et endiguement des lacs) et politiques d'utilisation des terres – visant à maintenir les processus et les valeurs de ces écosystèmes.

Des indices de vulnérabilité ont été élaborés pour évaluer la sensibilité actuelle de la végétation des milieux humides riverains des Grands Lacs et des oiseaux nicheurs dépendant des milieux humides aux changements hydrologiques, ainsi que celle des poissons aux changements hydrologiques et thermiques. On a utilisé pour les facteurs de vulnérabilité des cotes permettant de classer les espèces en trois groupes selon le risque auquel elles se trouvent exposées : risque faible, risque modéré et risque élevé. Les espèces végétales palustres présentant une tolérance à la sécheresse et des modes de colonisation limités ont été jugées les plus vulnérables. Par conséquent, il pourrait y avoir réduction de la diversité, particulièrement pour ce qui est des plantes aquatiques submergées et des plantes feuillées flottantes. Parmi les plantes jugées hautement vulnérables, on compte le riz sauvage (*Zizania palustris*) et le potamot de Hill (*Potamogeton hillii*). Plusieurs espèces envahissantes, comme la salicaire (*Lythrum salicaria*) et le roseau commun (*Phragmites australis*), ont été jugées les moins vulnérables. Les oiseaux qui nichent obligatoirement dans les milieux humides et dont les préférences en matière de nidification et d'alimentation exigent des conditions hydrologiques spécifiques ont été jugés très vulnérables, des niveaux d'eau relativement stables sur un longue période durant la saison de reproduction étant pour eux un facteur essentiel (p. ex. Sterne de Forster [*Sterna forsteri*], Guifette noire [*Chlidonias niger*], Grèbe à bec bigarré [*Podilymbus podiceps*], râles et butors). Bon nombre d'espèces d'oiseaux exposées à un risque élevé sont considérées comme en péril dans les Grands Lacs, ou présentent des tendances démographiques à la baisse, ce qui indique l'existence de stress qui pourraient être accentués par les changements climatiques. Parmi les poissons lacustres indigènes exposés à un risque élevé, jugés très sensibles aux changements des conditions riveraines, on compte les espèces des eaux fraîches à tièdes présentant des aires de répartition limitées, frayant au printemps en eaux peu profondes et préférant les milieux végétalisés à tous les stades de leur existence (p. ex. petit-bec [*Opsopoedus emiliae*], lépisosté tacheté [*Lepisosteus oculatus*] et maskinongé [*Esox masquinongy*]).

Des modèles ont été élaborés pour déterminer l'effet de la variabilité des niveaux d'eau sur les communautés végétales, l'abondance des oiseaux nicheurs et la qualité de l'habitat pour le poisson. En premier lieu, des analyses des tendances spatio-temporelles recourant à un système d'information géographique (SIG) ont relié les effets des conditions hydrologiques passées avec l'abondance et la répartition spatiale des communautés végétales palustres. Ces analyses ont aidé à l'élaboration d'un modèle basé sur des règles reliant les types de communauté végétale avec les profondeurs moyennes de l'eau et les conditions hydrologiques antécédentes (p. ex. la durée des inondations et assèchements antérieurs). Ce modèle a été appliqué aux milieux humides

des lacs Ontario et Érié choisis pour les études de cas; par la suite, l'information sur les communautés végétales obtenue a été utilisée comme information d'entrée pour les modèles visant les oiseaux et les poissons.

Les modèles visant les oiseaux ont été élaborés à l'aide de données de relevés portant sur l'abondance des espèces d'oiseaux nichant dans les prés humides et les milieux comportant des arbres ou des arbustes, ainsi que sur l'abondance des oiseaux palustres nichant dans la végétation émergente dans un éventail de profondeurs d'eau. Des équations de régression ou des densités fixes d'oiseaux ont été établies pour chaque guilde de nicheurs palustres, et ont été utilisées en combinaison avec des estimations de la disponibilité d'habitat palustre pour estimer les réponses des oiseaux nicheurs. Le modèle de disponibilité d'habitat du poisson (Habitat Alteration Assessment Tool, ou HAAT – outil d'évaluation de l'altération de l'habitat) a utilisé les préférences des poissons à divers stades pour estimer des valeurs de qualité de l'habitat dans des zones définies pour les poissons groupés par préférences alimentaires et thermales. Le modèle HAAT a été utilisé pour estimer les valeurs de qualité de l'habitat pour des combinaisons particulières de caractéristiques du milieu (profondeur, végétation submergée et émergente et type de substrat), puis pour estimer la superficie d'habitat propice dans le milieu humide pour six guildes de poissons et trois stades (fraye, jeune de l'année et adulte).

On a choisi quatre scénarios de changement climatique représentant des conditions climatiques extrêmes pouvant survenir dans le futur : réchauffement et humidité très marqués (chaud et humide), réchauffement et sécheresse très marqués (chaud et sec), réchauffement faible et humidité très marquée (pas aussi chaud et humide) et réchauffement faible et sécheresse très marquée (pas aussi chaud et sec). Ces scénarios ont été élaborés à partir de simulations réalisées à l'aide d'un modèle de climat du globe utilisant les scénarios d'émissions de gaz à effet de serre les plus à jour. Dans l'étude de la Commission mixte internationale sur le lac Ontario et le fleuve Saint-Laurent, ces scénarios ont été utilisés dans des modèles hydrologiques pour prévoir les niveaux d'eau; le présent projet a utilisé ces résultats concernant les niveaux d'eau. Les réductions annuelles moyennes projetées des niveaux d'eau dans les lacs Ontario, Érié, Sainte-Claire et Michigan-Huron varient de 15 à 118 cm pour l'an 2050 pour l'ensemble des scénarios, à l'exception d'un scénario pour le lac Ontario, qui projette une hausse (6 cm). Les changements des niveaux des lacs projetés pour 2050 pour chaque scénario ont été appliqués aux données sur les niveaux d'eau observés (conditions du scénario de référence) et utilisés comme information d'entrée dans les modèles visant les communautés végétales, les oiseaux et les poissons pour chaque milieu humide. Les modèles biotiques ont été exécutés en utilisant des niveaux d'eau bas et élevés pour les simulations initiales (et ont produit le scénario de référence pour les comparaisons avec les résultats des scénarios de changement climatique). Des périodes passées où ont été enregistrés des niveaux d'eau élevés et bas (périodes de sécheresse ou d'inondation et de fortes eau) ont été choisies en rapport avec les années pour lesquelles existent des données sur la végétation palustre pour validation du modèle visant les communautés végétales.

Toutes les sorties de modèle ont été utilisées pour réaliser une évaluation intégrée des impacts des changements climatiques sur les milieux humides des rives des Grands Lacs inférieurs. Huit milieux humides des lacs Ontario et Érié ont été évalués pour ce qui est des changements des communautés végétales palustres, six pour ce qui est des réponses des oiseaux et deux pour ce qui est des réponses de l'habitat du poisson aux extrêmes des scénarios de niveaux d'eau pour le passé et pour les changements climatiques. (Une modélisation pour le poisson a aussi été réalisée pour un milieu humide du lac Sainte-Claire.)

La modélisation des communautés palustres a indiqué que les bas niveaux d'eau projetés dans la plupart des scénarios de changement climatique auront une incidence sur la répartition et l'abondance de la végétation, des oiseaux et des poissons des milieux humides; des changements majeurs dans tous les groupes taxinomiques sont probables par suite de baisses à long terme des niveaux d'eau, les premiers changements apparaissant dans la végétation. Une baisse des niveaux d'eau favorise l'expansion des types de végétation des milieux plus secs, particulièrement aux bordures hautes des milieux humides, ainsi qu'une réduction des zones d'eau libre et de la végétation submergée associée dans la plupart des baies. L'hydrogéomorphologie joue un rôle important dans les réponses des milieux humides et de l'habitat. Selon les résultats des modèles, les communautés palustres protégées se trouvant dans des milieux humides de baies lacustres sont les plus susceptibles de s'adapter naturellement aux bas niveaux d'eau. Les adaptations possibles comptent une expansion de la végétation arborée et arbustive et de la végétation de pré humide dans les zones de végétation émergente. L'expansion de la végétation émergente et submergée vers le lac est projetée dans les parties basses et à des niveaux d'eau propices, mais d'autres facteurs affectant les conditions de croissance (protection contre le vent et les vagues, type de substrat ou sol, et pente) n'ont pas été pris en considération. Des changements importants de la répartition et de l'abondance des communautés palustres sont habituellement projetés pour les milieux humides fluviaux (embouchures submergées de cours d'eau), en raison de la topographie des plaines inondables. Dans ce type de milieu humide, l'expansion de la végétation émergente et submergée serait limitée au chenal du cours d'eau, et pourrait peut-être aussi se produire dans les deltas à l'intérieur de nouveaux éléments du milieu pouvant être formés par les processus littoraux associés à la baisse des niveaux d'eau.

Les communautés d'oiseaux et de poissons des milieux humides des rives des Grands Lacs peuvent répondre aux changements potentiels touchant les communautés végétales, les réponses variant selon les guildes d'oiseaux et de poissons. L'abondance potentielle des oiseaux nichant obligatoirement dans les marais se trouve à diminuer tandis que celle des oiseaux nichant en milieu arboré ou arbustif se trouve à augmenter dans tous les scénarios de baisse des niveaux d'eau. Les réponses projetées des oiseaux qui nichent obligatoirement dans les marais ou qui nichent dans les prés humides ainsi que celles de l'habitat de grossissement de guildes choisies de poissons sont variables et sont fonction des conditions initiales de niveaux d'eau bas ou élevés et du scénario de changement climatique. La modélisation par guilde et les évaluations de la vulnérabilité par espèce ont montré que certaines guildes ou espèces d'oiseaux et de poissons pourraient être plus affectées par les changements potentiels des moments, des durées et des profondeurs des crues, selon les communautés végétales. Pour les poissons, les changements de la température de l'eau découlant d'un réchauffement du climat entraîneraient aussi des réponses différentes selon les espèces. Les espèces d'oiseaux nichant sur l'eau et les espèce de poissons qui ont besoin d'une végétation submergée pour la reproduction et le grossissement des jeunes se sont révélées très vulnérables. D'après les modèles, les réponses propres aux divers endroits fondées sur leurs caractéristiques physiques déterminent les réponses biotiques des milieux humides, certaines guildes ou espèces pouvant se trouver favorisées et d'autres défavorisées. Ce résultat fait ressortir la nécessité d'évaluations à grande échelle pour déterminer les changements nets des communautés à l'échelle régionale.

Les adaptations humaines aux changements climatiques touchant les milieux humides et entraînant l'abaissement des niveaux d'eau peuvent prendre plusieurs formes : endiguement des milieux humides, régulation des niveaux d'eau à grande échelle et modifications en matière de planification de l'utilisation des terres riveraines. Selon la modélisation des milieux humides et les contributions des parties intéressées dans le cadre du présent projet, on devrait accorder une grande priorité aux plans d'aménagement territorial et aux initiatives qui protègent les processus naturels entraînant la formation des milieux humides et maintenant leur capacité de s'adapter aux variations des niveaux d'eau. Par conséquent, on a besoin de mécanismes permettant la prise en compte des tendances des changements climatiques et de l'information sur leurs impacts potentiels, comme les changements projetés de la répartition et des processus des milieux humides, dans les politiques et la planification exercées par les divers ordres de gouvernement. Dans les politiques et les programmes d'aménagement territorial actuellement appliqués dans la région des Grands Lacs, on n'a trouvé aucun exemple d'adaptation aux changements climatiques dirigée par l'homme visant à en réduire les incidences sur les milieux humides riverains des Grands Lacs ou toute autre zone riveraine naturelle. Dix critères de planification et le concept de corridor riverain ont été élaborés à titre d'idées préliminaires qui ont été proposées en vue de la protection future des zones riveraines; ces idées ont été discutées avec les parties intéressés durant la deuxième année du projet. La création d'un corridor riverain où le développement serait limité aiderait à maintenir les processus littoraux naturels dans le contexte des changements climatiques tout en protégeant les propriétés et en améliorant possiblement l'accès du public, à des coûts modiques à long terme.

L'évaluation préliminaire de la régulation des niveaux d'eau à l'échelle des lacs et l'examen détaillé de l'endiguement des milieux humides comme stratégies d'adaptation potentielles ont montré que celles-ci auraient des incidences positives sur certains biotes palustres riverains et des incidences négatives sur d'autres. Comme la régulation des niveaux d'eau au moyen d'ouvrages (p. ex. grands barrages et structures de régulation, et endiguement) est coûteuse, demande beaucoup de travail et exige des ressources à long terme, on doit mieux comprendre les impacts de tels ouvrages sur les processus des milieux humides avant d'en faire une promotion étendue à titre de stratégie d'adaptation aux changements climatiques pour les communautés palustres riveraines. L'analyse comparative des zones endiguées a démontré la vulnérabilité potentielle de l'infrastructure actuelle à l'égard des changements des niveaux d'eau, et a mis en évidence la nécessité de tenir compte dans la conception des ouvrages de divers scénarios concernant les niveaux d'eau futurs. Dans le cadre des enquêtes et consultations menées auprès des parties intéressées, ces dernières ont généralement exprimé l'idée que l'endiguement des milieux humides comme adaptation aux changements climatiques ne devrait constituer qu'une mesure secondaire, venant après la protection des processus naturels de compensation et ne devant être appliquée que dans des circonstances spéciales (p. ex. protection de l'habitat essentiel ds espèces en péril).

Les activités de recherche dans lesquelles il faudrait investir pour accroître notre connaissance scientifique des vulnérabilités des milieux humides riverains des Grands Lacs à l'égard des changements climatiques et pour analyser de façon détaillée les stratégies d'adaptation potentielles devraient être les suivantes :

- vérification et validation des vulnérabilités et des prévisions des modèles relatives aux espèces et guildes palustres examinées dans le présent projet, et détermination des seuils de niveau d'eau pour ces espèces et guildes, le cas échéant;
- étude des effets interactifs du type de substrat, d'autres facteurs environnementaux et des processus littoraux sur la colonisation et la succession de la végétation palustre (émergente et submergée) quand les niveaux d'eau baissent;
- réalisation d'évaluations régionales des populations des espèces clés d'oiseaux et de poissons prenant en considération des paramètres spatiaux et temporels ignorés dans la présente évaluation;
- prise de mesures de terrain uniformes et de haute qualité et résolution en ce qui concerne la bathymétrie, l'élévation du terrain et les types de substrat dans les zones riveraines à l'interface terre-eau, permettant l'amélioration des modèles altimétriques numériques actuels et la mise en œuvre de méthodes normalisées de collecte de données pour l'ensemble de la région, ce qui facilitera la modélisation à grande échelle et les évaluations à l'échelle des bassins;
- repérage des milieux humides riverains, zones de transition et processus naturels importants devant faire l'objet de mesures de conservation et de protection; et
- interaction avec les parties intéressées de la zone littorale des Grands Lacs pour entretenir un dialogue concernant les impacts potentiels des changements climatiques et les stratégies d'adaptation, avec pour but de faire en sorte que la recherche sur les changements climatiques soit prise en considération dans les politiques et la planification.

Il faut que les politiques et les activités de planification actuelles et futures en matière de conservation des Grands Lacs tiennent compte des connaissances acquises sur les impacts des changements climatiques (p. ex. changements des niveaux d'eau et leurs incidences) ainsi que des résultats des recherches futures, dans l'optique d'une gestion adaptative. Il est particulièrement important de déterminer les seuils spatiaux et temporels affectant la capacité d'adaptation et l'intégrité biotique des milieux humides des rives des Grands Lacs.

1.0 INTRODUCTION

Linda Mortsch and Joel Ingram

Coastal wetlands are located in dynamic environments along the Great Lakes shoreline, and are directly influenced by fluctuating water levels both seasonally and over cycles of several years. Although they share many of the same functions and values as inland wetlands, it is the influences from large lake processes that differentiate coastal wetland hydrology and vegetation structure from inland wetlands. In Ontario alone, over 1,000 coastal wetlands have been identified (Environment Canada 2003). The most common vegetation community in the lower Great Lakes coastal wetlands are marshes that support a mixture of floating, submerged, emergent, and meadow vegetation. Other wetland types common to the Great Lakes basin include swamps which are dominated by tree and shrub species, and fens and bogs which are peat-accumulating wetlands dominated by meadow and moss species, respectively. Fluctuations in Great Lakes water levels influence coastal wetland extent and distribution, vegetation composition, and wetland-dependent birds, fishes, and other wildlife as well as determine ecological diversity and functioning (Jaworski *et al.* 1981; Keddy and Reznicek 1986; Quinlan and Mulamoottil 1987; Casanova and Brock 2000; Environment Canada 2002).

Great Lakes coastal wetlands are now more widely recognized as ecologically diverse and highly productive ecosystems that perform important functions by providing habitat, improving water quality, protecting against flooding and erosion, and allowing for recreation opportunities. A crucial benefit is that numerous regionally endangered and threatened birds, reptiles, fish, and amphibians use coastal wetlands for all or part of their life cycles. In the Great Lakes, more than two-thirds of all lake fish species spawn in coastal wetlands while many bird species rely solely on wetland habitat for nesting and rearing young. The ecological importance of wetlands has not always been appreciated. Losses of wetland area have been extensive as humans have drained, filled, and appropriated the land for development. For example, during the past two centuries, over two-thirds of the original wetland area in southern Ontario has been lost with the greatest decreases (over 90 percent (%)) in parts of southwestern Ontario and along the Lake Erie shoreline in Ohio (Snell 1987; Robb and Mitsch 1990; Environment Canada 2002). While rates of wetland loss have declined in the last few decades, many wetlands continue to be degraded by human activities through: sediment, nutrient, and contaminant loading; dredging; recreational use; and introductions of invasive species, such as purple loosestrife (*Lythrum salicaria*) and common carp (*Cyprinus carpio*), affecting habitat quality and displacing native flora and fauna (Miles *et al.* 1976; Murdoch 1981; Thompson *et al.* 1987; Herdendorf 1992; Mills *et al.* 1993; Downey *et al.* 1994; Knapton *et al.* 2000; Mitsch and Gosselink 2000).

Climate change may be an additional stress on Great Lakes wetlands already at risk from human pressures. While Great Lakes coastal wetlands are adapted to variations in water levels, climate change, through warmer air temperatures, increases in evaporation, and changes in precipitation and snowcover, could significantly alter Great Lakes hydrology over the next 50 years, relative to the past 150 years. Changes can be expected in the mean level, annual range, and seasonal cycle as well as the timing, amplitude, and duration of water levels. The most critical impact is projected lower mean water levels in the Great Lakes (Mortsch *et al.* 2000; Lofgren *et al.* 2002). Since water levels have such an influence on coastal wetlands, significant changes can be expected in wetland area and distribution, and wetland vegetation communities mediated, in part, by geomorphic form affecting the capacity to adapt. Changes in interspersion and vegetation communities influence the suitability of habitat for wetland-dependent birds and fishes.

A better understanding of Great Lakes coastal wetland community responses to future water level changes will allow for proactive wetland conservation planning. Incorporating adaptation to climate change into decision-making on future wetland securement, restoration, and management will help protect important Great Lakes coastal wetland functions and values.

1.1 RESEARCH DESIGN

The collaborative research project was undertaken by Environment Canada (Atmospheric Science and Technology Directorate, Canadian Wildlife Service), Fisheries and Oceans Canada (Great Lakes Laboratory for Fisheries and Aquatic Sciences), and the University of Waterloo (Department of Geography) to produce an integrated climate change assessment to:

- describe vulnerabilities of Great Lakes coastal wetland ecosystems (wetland vegetation and associated wetland-dependent birds and fish) to water level change as a surrogate for climate change, and
- develop and assess both management policy instruments and infrastructure adaptation strategies to maintain ecosystem function and values.

This research has built upon conceptualizations of coastal wetland response to water level change developed by Great Lakes researchers (ILERSB 1981; Jaworski *et al*. 1981; Herdendorf *et al*. 1986; Keddy and Reznicek 1986). It has utilized Geographic Information System (GIS) data sets developed during the International Joint Commission (IJC) Water Level Reference where historical air photos were interpreted to document wetland vegetation change over time (Working Committee 2, Land Use and Management 1993). Funding from the Government of Canada Great Lakes 2000 program (GL2000) allowed more air photo analysis to extend the time series for wetlands in Lakes Ontario and Erie and undertake the first assessment of Lake Huron fens. Air photo analysis and field work on understanding the relationships between Lake Ontario water level fluctuations and wetland plant, bird, and fish communities undertaken in support of the current IJC Lake Ontario-St. Lawrence River (LOSLR) Study has contributed extensively to this project (DesGranges *et al*. 2005; Doka *et al*. 2005; Wilcox *et al*. 2005). GIS-based approaches for modelling wetland vegetation response to water level change were explored by Hebb (2003). An initial assessment outlining vulnerabilities of Great Lakes coastal wetlands to water level changes (i.e. changes in the mean level and seasonal cycle) due to climate change was undertaken by Mortsch (1998).

The wetland vulnerability assessment consisted of three main components: wetland ecosystem modelling, adaptation strategies, and stakeholder engagement; these components and the research process, are outlined in Figure 1.1.

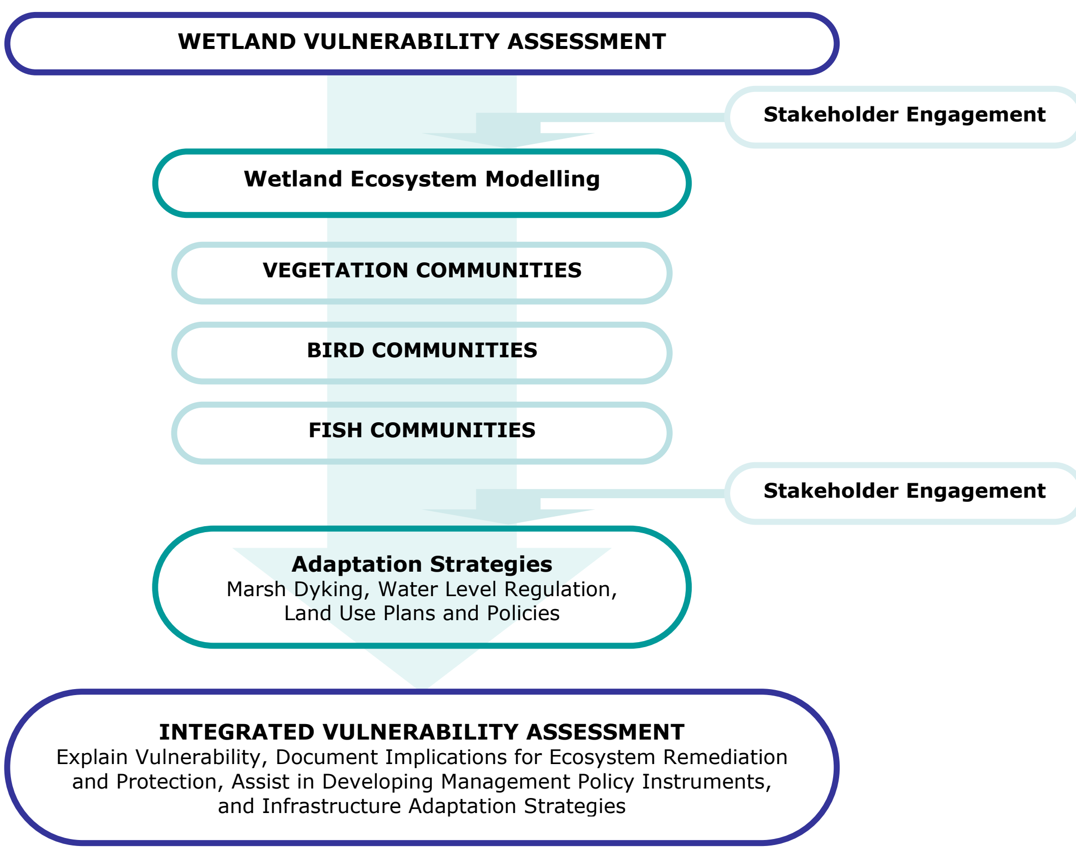

Figure 1.1 Simplified schematic of research process utilized to assess Great Lakes coastal wetland vulnerabilities to climate change and adaptation strategies

1.1.1 Wetland Ecosystem Modelling

The wetland ecosystem modelling component developed models based on historic and current responses of wetland vegetation, bird, and fish communities to water level changes, and used climate change scenarios and integration of modelling results to develop an overall assessment of future implications for Great Lakes coastal wetlands to climate change. The vegetation community component used temporal and spatial trend analysis in GIS to document historical wetland vegetation response to water levels and identify key vegetation changes at water level thresholds. A rule-based model was developed to link vegetation community occurrence with current water depth and antecedent hydrologic conditions (duration of flooding or dewatering). Potential changes in vegetation community type and patterning due to climate change water level scenarios were an important input for subsequent modelling of impacts on wetland-associated bird and fish communities using bird and fish habitat suitability models. These habitat suitability models were developed from literature reviews and field surveys of bird community response to hydrologic change, and fish assemblage response to hydrologic and thermal changes.

1.1.2 Adaptation Strategies – Development and Assessment

Three potential adaptation strategies for addressing climate change impacts in Great Lakes coastal wetlands were investigated and evaluated. They included:

- modifications to water level regulation at Moses-Sanders Dam to ameliorate the effects of water level changes due to climate change on Lake Ontario coastal wetlands;
- wetland dyking on Lakes Ontario, Erie, and St. Clair to manage water levels within wetlands to offset lower water levels in the Great Lakes; and
- land use planning and policy options as tools to protect coastal wetlands from water level changes due to climate change.

Results from the IJC LOSLR Study provided information on whether whole-lake water level regulation plans were able to preserve hydrologic attributes important to the maintenance of coastal wetland communities in Lake Ontario. The viability of wetland dyking as an adaptation strategy was assessed from two perspectives: maintaining ecological diversity, and robustness of dyke infrastructure. Comparison of survey data on wetland plant, bird, and fish communities from paired dyked and undyked wetlands in Lakes Ontario, Erie, and St. Clair was used to assess differences in community diversity. Operation of pumping infrastructures under the water level regimes from climate change scenarios was also explored. A review of existing land use policy and planning processes was undertaken to determine the potential for using these instruments as climate change adaptation tools to protect existing wetlands from increased development pressures as well as the newly created wetlands and shorelines that emerge as a result of declining water levels. Ten Planning Criteria for coastal wetland adaptation to climate change and a Coastal Corridor Concept are presented as preliminary proposals to launch discussion amongst stakeholders on climate change impacts, coastal wetlands, and land use planning.

1.1.3 Stakeholder Engagement

There are many interests and extensive expertise on Great Lakes coastal wetlands within the wetland stakeholder community. Participants for stakeholder engagement were drawn from organizations in Canada and the U.S., and represented wetland, bird, and fisheries perspectives at national, provincial, and state levels. Stakeholders were engaged twice during the project and contributed to:

- finalization of the research project scope and integration of expertise, data, and information from their sources; and
- review of preliminary results of climate change impact assessments, and discussion and assessment of potential adaptation strategies. Impacts and adaptations feedback provided by stakeholders was incorporated into the project (Appendix 1).

The project website (http://www.fes.uwaterloo.ca/research/aird/wetlands) was used as an ongoing communication tool with the Great Lakes community (Figure 1.2).

Figure 1.2 Project website (http://www.fes.uwaterloo.ca/research/aird/wetlands)

1.2 Study Sites

Numerous wetland sites were selected for vegetation analysis and modelling, bird and fish assessments and modelling as well as comparison of dyked and undyked wetlands. The location of all wetland sites are outlined in Figure 1.3, while site-specific maps are used to depict wetland vegetation and bird and fish habitat modelling results in subsequent chapters.

1.3 Report Outline

This report progresses through each of the component research activities as outlined in Figure 1.2. Chapter 2 develops the context on historical Great Lakes water level conditions, the relationship of coastal wetland vegetation to water level fluctuations, climate change projections for the Great Lakes basin, and potential vulnerabilities of coastal wetlands to climate change. In Chapter 3, a literature review develops an

Figure 1.3 Wetland study sites

understanding and assessment of Great Lakes coastal wetland plant community vulnerabilities to climate-induced hydrological change. Historical analyses of marsh and fen vegetation responses to water level conditions are documented in Chapter 4; this analysis is used, in part, to develop the wetland vegetation response model. In Chapters 5 and 6, respectively, literature reviews and models to assess marsh bird community response to water level changes and then fish community responses are described. These models are used to undertake an integrated assessment of coastal wetland vegetation, bird, and fish community response to mean water level reductions in Lakes Ontario and Erie due to climate change (Chapter 7). Potential adaptation strategies to respond to climate change including whole-lake water level regulation, wetland dyking, and land use planning are discussed in Chapter 8. A final synopsis of the report is provided in Chapter 9. Discussions from two stakeholder meetings are reported in Appendix 1, and in addition, key findings and issues arising from the stakeholder meetings are incorporated throughout the report.

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2.0 Climate Variability and Change within the Context of the Great Lakes Basin

Linda Mortsch, Elizabeth Snell, and Joel Ingram

This chapter provides an introductory review of: effects of historic climate variability on water level fluctuations in the Great Lakes; responses of Great Lakes coastal wetlands to these water level fluctuations; potential impacts of climate change in the Great Lakes basin; effects of lower water levels on Great Lakes coastal wetlands; and climate change adaptation and mitigation.

2.1 Historic Water Level Fluctuations in the Great Lakes

Lake level fluctuations integrate seasonal and long-term variability of the climate system. Over the years of instrumental measurement, annual average water levels in Lakes Ontario, Erie, and Michigan-Huron as well as Lake St. Clair have fluctuated from year-to-year (Figure 2.1); although levels have varied within a rather small range around 1.8 metres (m) from maximum to minimum level. Inter-annual water level fluctuations reflect the interaction between climate-related water losses due to evaporation from the surface of the lakes and evapotranspiration in the watershed, and water gains through precipitation on the land and lake surfaces, tributary inflows, and inflow from upstream lakes through connecting rivers. Great Lakes water levels were very high in 1973-75, 1985-86, and 1997 and very low in 1934-35 and 1964-65. Since the early 1970s, there has been a run of high water supplies (wet weather) with water levels in most years above long-term averages (Magnuson *et al.* 1997). In 1987 and 1988, levels dropped dramatically from record highs in 1986 due to high temperatures and below normal precipitation.

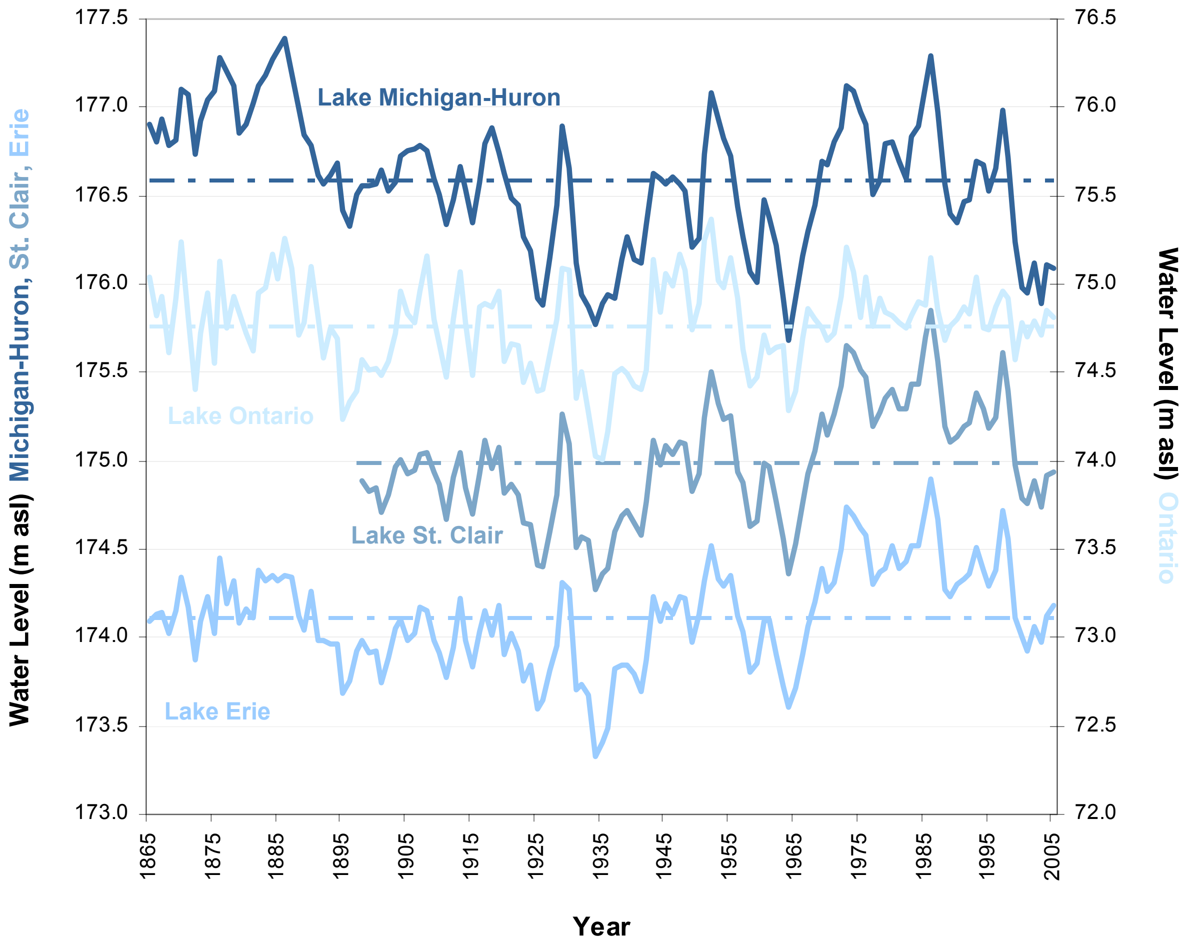


Figure 2.1 Annual average water level (solid) and long-term mean water level (dashed) for Lakes Michigan-Huron, Erie, and Ontario (1865-2005) and St. Clair (1898-2005) (Moulton pers. comm.)

Water levels dropped again from highs in 1997, in part because 1998 was the hottest year (+2.3 Celsius degrees (ºC)) and fifth driest year (-8.9%) in the Great Lakes region for the 51-year record at that time. The drought that began in 1998 and lasted until 2002 (excluding 2000) affected the water balance of the Great Lakes significantly; summer temperatures ranged from 0.9 to 1.3ºC above average while exceedingly below normal summer precipitation occurred in 2001 (-26.8%) and 2002 (-15.4%), and ranged from -1.0 to -4.3% in the other years. Lake Michigan-Huron water levels were affected the most.

Natural climatic influences on water levels predominate in all the Great Lakes although outflows from Lakes Superior and Ontario have been managed through control structures since 1921 and 1960, respectively. For example, the IJC issued Orders of Approval for regulation of water levels and flows on Lake Ontario according to a formalized regulation plan – Plan 1958D. Objectives of the Plan were to reduce extremes in water levels and maintain downstream releases to the St. Lawrence River within a prescribed range in order to provide dependable flow for hydropower, adequate depths for navigation, and protection for downstream shoreline interests. In December 2000, the IJC established the International LOSLR Study Board to conduct a five-year study to assess and evaluate the current criteria used for regulating water levels and flows on Lake Ontario and in the St. Lawrence River. The LOSLR Study considered the effects of changes in the system, including climate change, as well as effects of water level fluctuations on a wide range of interests including navigation, coastal erosion, shoreline communities, domestic and industrial water uses, commercial navigation, hydropower production, the environment, and recreational boating and tourism.

2.2 WETLAND RESPONSE TO HISTORIC WATER LEVEL CHANGES

Great Lakes coastal wetlands occupy a unique transitional position between aquatic and terrestrial environments that provides a diversity of abiotic conditions for plant germination and growth. A continuum of substrate types, shoreline gradients, and water depths occur within these wetlands (Keddy 2000; Mitsch and Gosselink 2000). Many plants have unique characteristics that enable their growth and survival in wetland environments. Some species can tolerate a range of environmental conditions, while others have a very narrow niche. Wetland plant species that possess similar tolerances generally grow at comparable elevations and moisture conditions. These wetland plants have been classified into five plant communities: 1) submergent macrophytes, 2) surface floating macrophytes, 3) emergent macrophytes, 4) wet meadow, and 5) trees and shrubs. The relationship between hydrology and these wetland vegetation communities is shown in Figure 2.2. Wetland vegetation communities expand and contract along a moisture gradient with fluctuating lake levels (Keddy and Ellis 1985).

Great Lakes coastal wetlands are continually responding to current and antecedent water levels which act as a "perturbation" on the wetland biophysical system (Jaworski *et al.* 1979; Keddy and Reznicek 1986; Quinlan and Mulamoottil 1987; Casanova and Brock 2000). Variations in water levels maintain diversity of vegetation and habitat interspersion. Also, the relative abundance of vegetation communities changes as certain plant species die back and vegetation is displaced landward or lakeward in response to water level changes. For example, during low water years, landward margins of wetlands dry and mudflats are exposed as water retreats lakeward. Emergent vegetation is displaced by sedges, grasses, and shrubs that expand into areas where the water was once too deep. Submerged aquatic vegetation is replaced by emergent vegetation as germination occurs on exposed mudflats. With the return of high water levels, vegetation communities slowly retreat landward.

Wetlands are often associated with coastal landforms such as barrier beaches, deltas, embayments, and shallow, sloped shorelines. Shoreline geomorphology integrates factors such as exposure to wind and waves, sediment deposition, slope, and landforms along the coast which in turn influence where a wetland develops, its areal extent, hydrologic character, and the structure of wetland biotic communities. In order to study hydrologic and biotic interactions within coastal wetlands, wetlands have been categorized into ecologically more definitive types based on hydrogeomorphic form (Albert *et al.* 2005) (Figure 2.3).

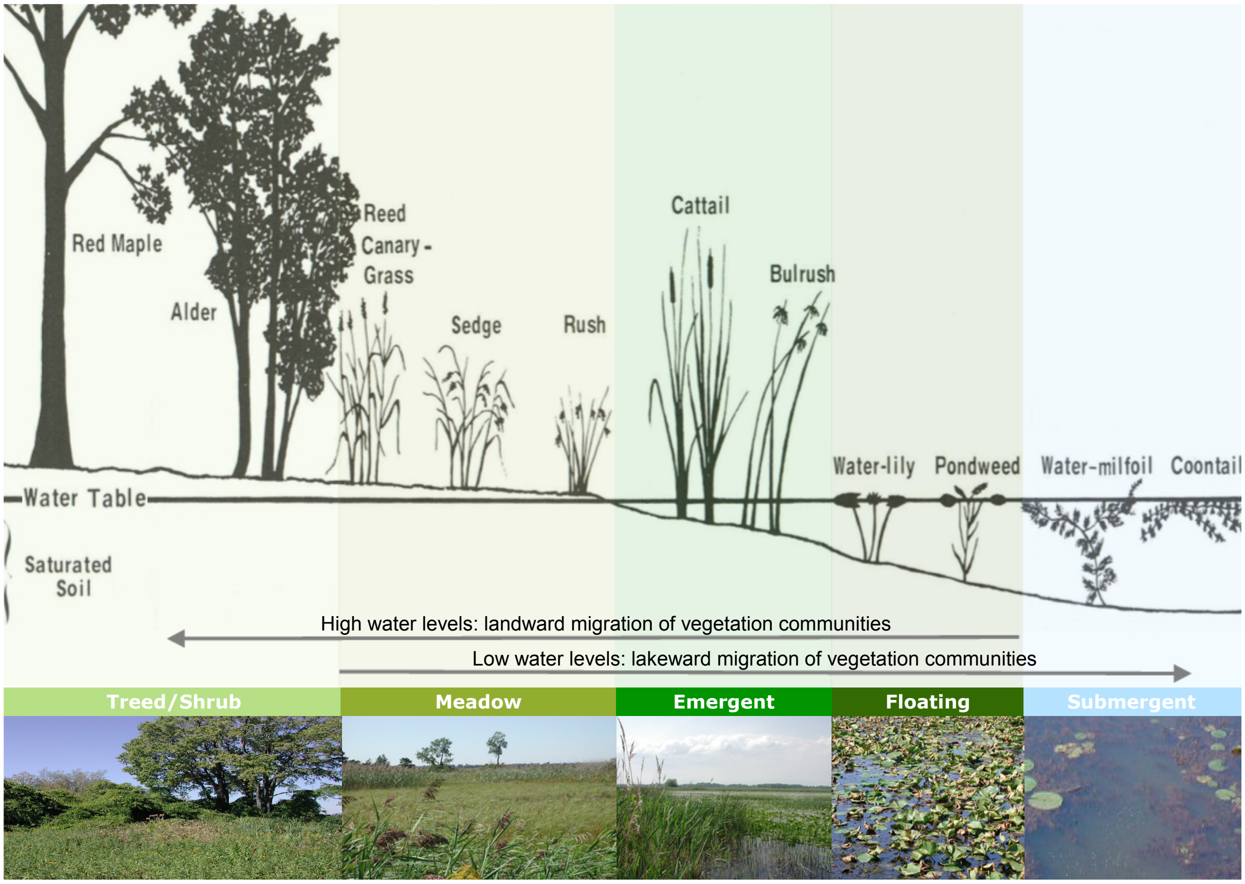


Figure 2.2 Wetland vegetation community development along water table continuum (adapted from Bolsenga and Herdendorf 1993)

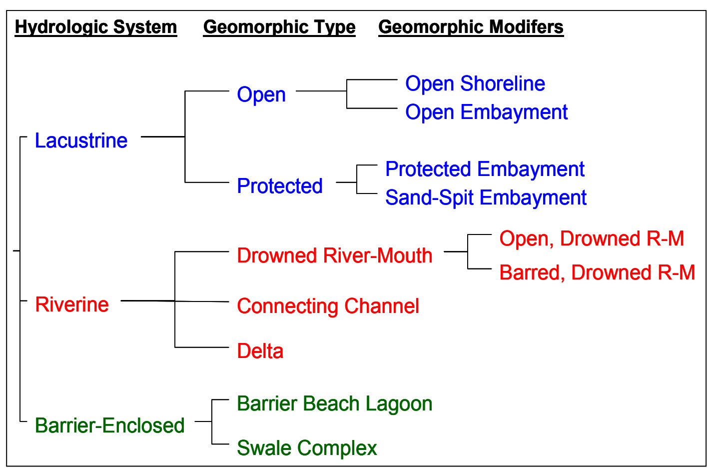


Figure 2.3 Wetland site types based on hydrogeomorphic form (adapted from Albert *et al.* 2005)

Wetland hydrogeomorphic types have unique locations in the coastal zone which influence the response to water regime changes and development of wetland vegetation communities. The hierarchical classification system (Figure 2.3) first divides wetlands into three broad hydrogeomorphic systems based upon hydrologic sources and connectivity to the lake. Lacustrine wetlands are directly controlled by Great Lakes water levels and strongly influenced by nearshore processes such as currents and ice scour. The exposure to nearshore processes influences the degree of sediment deposition and vegetation development. Riverine wetlands occur where rivers or creeks flow into or between the Great Lakes. Wetland water chemistry, water flow, and sedimentation are highly dependent on upstream watersheds. However, the wetland water levels are predominately influenced by the lake, as lake waters flood back into lower portions of the river-mouth. Extensive vegetation zones usually occur on deposited sediments associated with a shallow sloping deltaic or flood plain feature. Barrier-enclosed wetlands occur behind a barrier beach or other barrier feature that has developed due to coastal processes. These types of wetlands are highly protected from nearshore processes; typically they have a shallow sloping shoreline with significant organic sediments and vegetation community development. Water chemistry and levels are influenced by the lake, surface drainage, and groundwater. The influence of lake levels is determined by the degree of connectivity, which may change frequently as barrier beaches open and close to the lake. Within the system classification, coastal wetlands are furthered classified based upon additional geomorphic features and shoreline processes (Albert *et al.* 2005).

There is a close relationship between a wetland's hydrogeomorphic type and the potential impacts caused by changes in water levels (Keough *et al.* 1999). Topography and bathymetry of a wetland determine how changes in water levels affect the ability of wetland vegetation to migrate to suitable moisture and substrate conditions. If the wetland borders are steep or confined by a barrier, expansion of the wetland is unlikely during lake level fluctuations causing wetland communities to flood or dry out (Bedford *et al.* 1976; Quinlan and Mulamoottil 1987). For example, during high water levels, wetland habitat, particularly in wetlands with restricted upland borders, can be lost due to flooding (Sherman *et al.* 1996; Gottgens *et al.* 1998). During low water levels, some wetlands may advance lakeward if suitable off-shore slope, wave protection, and substrate conditions exist while other wetlands may be impeded from migrating and progress to drier habitat. For these reasons, wetland hydrogeomorphology must be considered when evaluating potential coastal wetland response and vulnerability to climate change, and adaptation strategies to maintain wetland functions and values (Mortsch 1998).

2.3 Projected Changes in the Climate of the Great Lakes Basin

Rising concentrations of greenhouse gases are projected to lead to a suite of changes in climate but the most certain outcome is an increase in global air temperature while precipitation changes are more uncertain (IPCC 2001a).

Climate change scenarios for the LOSLR Study were developed from the Global Climate Model (GCM) simulations of SRES emission scenarios in order to calculate net basin supply changes and model hydrologic and lake level effects (Croley 2003; Mortsch *et al.* 2005). The four climate change scenarios were chosen to depict: 1) the most warming and wettest conditions (warm & wet), 2) the most warming and driest conditions (warm & dry), 3) the least warming and wettest conditions (not as warm & wet), and 4) the least warming and driest conditions (not as warm & dry). The area-average temperature and precipitation changes for the Great Lakes basin are summarized in Table 2.1. These climate change scenarios were also used in this study. In the Great Lakes region, warming is projected to occur in all seasons with the greatest warming usually in winter. Annual precipitation is expected to increase, although a summer decrease in precipitation is projected in the warm & dry scenario.

Table 2.1 GCM-projected temperature and precipitation changes in the Great Lakes for 2050 relative to 1961 to 1990 baseline conditions (Mortsch *et al.* 2005)

GCM and SRES Emission Scenarios	Scenario Description	Area-Average Temperature Change (°C)					Area-Average Precipitation Change (%)				
		Annual	Winter	Spring	Summer	Autumn	Annual	Winter	Spring	Summer	Autumn
HadCM3 A1FI	Warm & Wet	+4.0	+3.9	+3.9	+4.4	+4.1	+10.3	+21.5	+19.2	+3.1	+4.7
CGCM2 A21	Warm & Dry	+3.2	+4.3	+3.3	+3.2	+2.2	+1.4	+4.4	+4.4	-1.6	+1.3
HadCM3 B22	Not as Warm & Wet	+2.8	+3.3	+2.4	+3.1	+2.6	+12.5	+20.8	+19.9	+7.7	+8.0
CGCM2 B23	Not as Warm & Dry	+2.2	+3.2	+2.6	+2.3	+1.6	+2.8	+5.3	+6.6	+0.1	+1.4

Warming affects other climate-related factors relevant to wetland ecology and hydrology in the Great Lakes. An increase in air temperature is mirrored by similar increases in water temperature; warming can be particularly significant in shallow, near-shore areas and affect dissolved oxygen content. In southern Canada, winter and spring warming may shift the proportion of winter precipitation from snowfall to rain. Also, the extent, depth, and duration of snowcover could be reduced by less snowfall, frequent thawing events, and earlier spring melt. Significant reductions in ice duration and extent on the Great Lakes are expected, and in some years the lakes may be ice-free in winter (Lofgren *et al.* 2002). A shorter ice cover season leads to more evaporation and contributes to lower lake levels as the greatest evaporative losses from the Great Lakes occur in late fall and winter when cold, dry air passes over the warmer moist lakes. Ice protects the shoreline from winter storms and is an effective barrier against wave erosion.

Extreme weather events are expected to increase with climate change (IPCC 2001a). For example, more intense precipitation events are consistent with a warmer atmosphere having a greater moisture-holding capacity and an enhanced hydrologic cycle (Trenberth 1999; Kharin and Zwiers 2000). On a global scale, the 20-year return values of daily precipitation from the Canadian Centre for Climate Modelling and Analysis (CCCma) CGCM1 GCM increased by 8% and 14% in 2040-2060 and 2080-2100, respectively (Kharin and Zwiers 2000). More intense precipitation events increase the risk of flooding and increase soil erosion causing entrainment and delivery of sediments, nutrients, and pesticides in surface waters (SWCS 2003). Although analyses of precipitation in GCMs indicate more heavy precipitation events, more dry days or days with light precipitation are also projected (Cubasch *et al.* 1995; Hennessy *et al.* 1997; Trenberth 1999). Also, a one in 80-year global temperature extreme today may occur with one in 10-year probability by 2050 (Kharin and Zwiers 2000). Droughts are also expected to increase in frequency and duration (Whetton *et al.* 1993; Francis and Hengeveld 1998).

Future water levels in the Great Lakes will reflect a critical balance between the timing and amount of precipitation, the increase in evaporation from the lakes, and higher evapotranspiration losses in the watershed leading to changes in water supply to the lakes. Projected outcomes include lower water levels and alteration in the seasonal water level cycle. Low water levels are likely to increase in frequency and duration due to reductions in net basin supplies – primarily due to higher evapotranspiration losses – to the Great Lakes (Mortsch *et al.* 2000; Lofgren *et al.* 2002; Croley 2003). For the LOSLR Study, the four climate change scenarios were applied to 50 years of daily climate data for over 1,600 climate stations in the Great Lakes basin in order to model net basin supply changes and water level responses (Croley 2003). Projected annual and seasonal water levels decrease for most scenarios (Table 2.2). In Figure 2.4, average monthly water levels are summarized for the modelled historical base case conditions and climate change conditions. Most climate change water levels fall below the base case mean water level conditions. In the two extreme climate change scenarios (warm & wet and warm & dry), average monthly water levels fall below recorded historic low extremes for extended periods.

Table 2.2 Projected changes in Lakes Ontario, Erie, St. Clair, and Michigan-Huron mean water levels (in metres) for 2050 with respect to base case (Fay and Fan pers. comm.)

SRES Emission Scenario		Annual	Winter	Spring	Summer	Autumn	Growing Season
Lake Ontario							
HadCM3 A1FI	Warm & Wet	-0.22	-0.17	-0.19	-0.28	-0.23	-0.28
CGCM2 A21	Warm & Dry	-0.37	-0.26	-0.42	-0.49	-0.30	-0.51
HadCM3 B22	Not as Warm & Wet	+0.02	+0.07	-0.07	-0.05	0.00	-0.04
CGCM2 B23	Not as Warm & Dry	-0.15	-0.08	-0.20	-0.24	-0.06	-0.27
Lake Erie							
HadCM3 A1FI	Warm & Wet	-0.67	-0.69	-0.62	-0.64	-0.73	-0.63
CGCM2 A21	Warm & Dry	-0.81	-0.79	-0.79	-0.83	-0.85	-0.81
HadCM3 B22	Not as Warm & Wet	-0.15	-0.15	-0.10	-0.13	-0.21	-0.11
CGCM2 B23	Not as Warm & Dry	-0.55	-0.55	-0.53	-0.54	-0.57	-0.53
Lake St. Clair							
HadCM3 A1FI	Warm & Wet	-0.81	-0.81	-0.77	-0.80	-0.87	-0.78
CGCM2 A21	Warm & Dry	-0.98	-0.95	-0.98	-1.01	-1.01	-1.00
HadCM3 B22	Not as Warm & Wet	-0.20	-0.21	-0.16	-0.20	-0.26	-0.18
CGCM2 B23	Not as Warm & Dry	-0.63	-0.62	-0.61	-0.64	-0.65	-0.63
Lake Michigan-Huron							
HadCM3 A1FI	Warm & Wet	-0.98	-1.00	-0.94	-0.97	-1.02	-0.95
CGCM2 A21	Warm & Dry	-1.18	-1.16	-1.16	-1.22	-1.20	-1.21
HadCM3 B22	Not as Warm & Wet	-0.29	-0.32	-0.25	-0.27	-0.32	-0.25
CGCM2 B23	Not as Warm & Dry	-0.73	-0.73	-0.70	-0.74	-0.74	-0.73

Typically, Great Lakes water levels progress through an annual cycle of highs and lows that range between 30 and 50 centimetres (cm). Levels are at their lowest in winter and rise in spring as snowmelt in the basin increases inflow to the lakes; they reach their maximum in June to September (depending on the lake). When water losses from the lake due to outflows and increased evaporation and evapotranspiration exceed incoming water supply, lake levels begin their annual decline. A changing climate could alter this seasonal progression of water levels. Warmer winters with more winter rainfall events would result in more direct runoff to the lakes contributing to higher winter water levels. While spring water levels may rise sooner due to warmer springs and an earlier melt of snowpack. Maximum levels may be diminished due to less winter snowpack and earlier initiation of evaporation and evapotranspiration. The autumn decline may occur earlier and minimum levels may be lower due to higher evaporation/evapotranspiration reducing runoff to the lakes

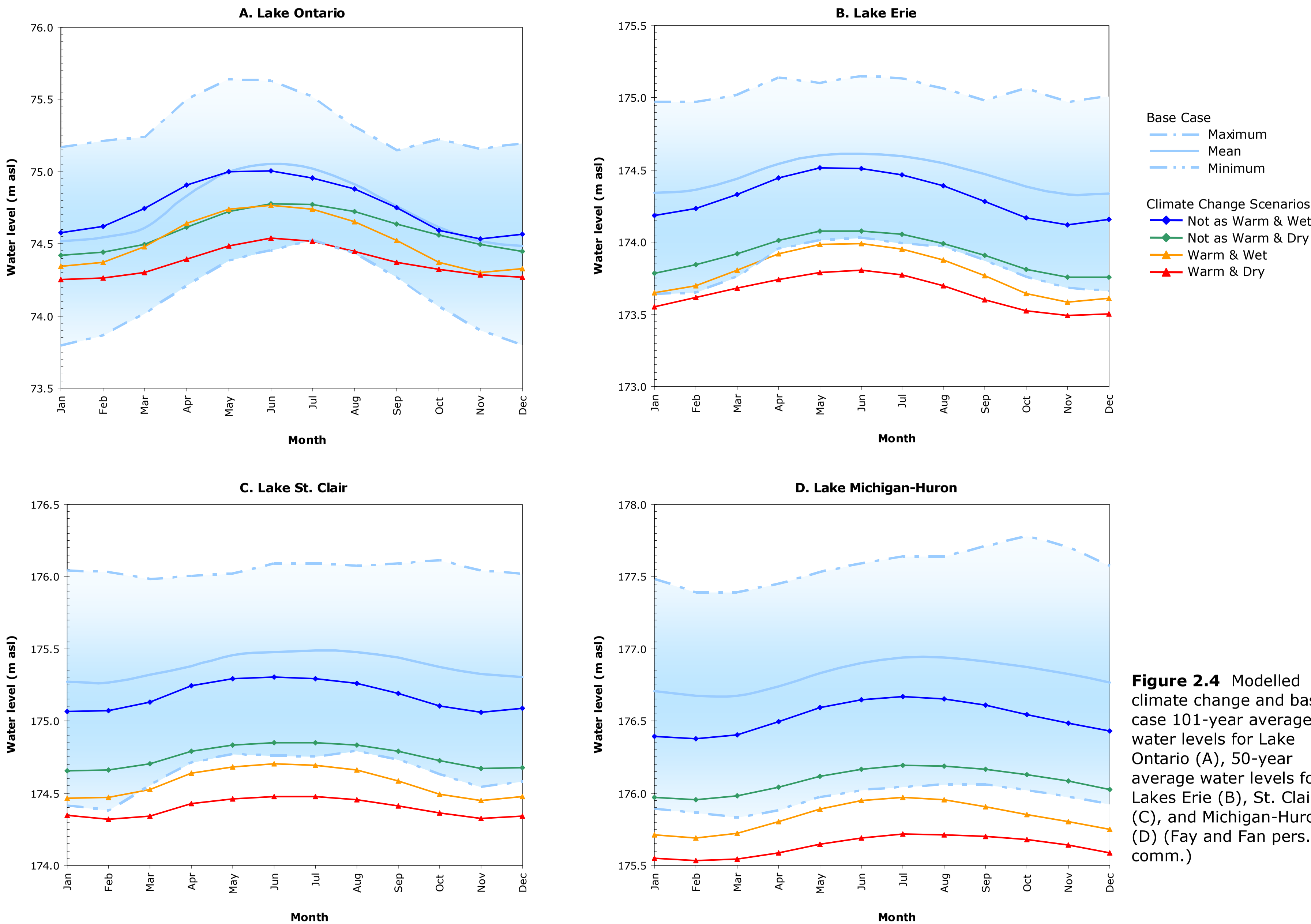


Figure 2.4 Modelled climate change and base case 101-year average water levels for Lake Ontario (A), 50-year average water levels for Lakes Erie (B), St. Clair (C), and Michigan-Huron (D) (Fay and Fan pers. comm.)

during summer and fall. In some climate change scenarios, precipitation decreases during the summer and autumn which may exacerbate low streamflow contributions to the Great Lakes during these periods. Distinct shifts in the seasonal cycle of Great Lakes water levels have been detected. In Lakes Erie and Ontario, from 1860 to 1990, the annual rise and fall of levels have advanced by approximately one month; spring levels were higher and fall levels were lower sooner (Lenters 2001). Lake Michigan-Huron also exhibited a change in the timing and range of the seasonal water level cycle since 1920 (Argyilan and Forman 2003). The modelled climate change water levels shown in Figure 2.4 do not currently exhibit this seasonal pattern.

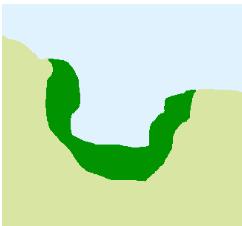
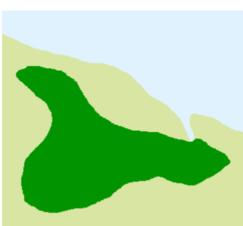
2.4 Projected Impacts of Climate Change on Great Lakes Coastal Wetlands

Climate, including temperature, precipitation, and other elements, is a key determinant of the distribution, productivity, and functioning of wetland ecosystems. Human-caused warming establishes new baseline temperature conditions which influences temperature extremes, extends the growing season length, alters phenological events, and increases potential evapotranspiration. Changes in the precipitation regime are revealed through alterations in the form of precipitation, its seasonal distribution, and precipitation extremes. Yet, the most crucial climate change impact on coastal wetlands would be changes in the hydrologic regime as it defines the critical coastal processes and moisture conditions for development of wetland soils and vegetation. This section provides a brief overview of potential impacts of climate change on Great Lakes coastal wetlands from the literature; site-specific modelling of wetland ecosystem response to climate change scenarios and discussion of these results are undertaken in Chapter 7 - Integrated Assessment.

As described in Section 2.3, climate change is expected to lower mean water levels and alter seasonal water level cycles in the Great Lakes. These changes would have numerous effects on coastal wetland functions and values, including changes in vegetation extent, composition, and diversity and as a consequence habitat quantity and quality for wetland-dependent wildlife (Mortsch and Koshida 1996; Mortsch 1998). It has been demonstrated that wetland hydrogeomorphic form exerts a fundamental control in determining how a wetland responds to water level changes. In Table 2.3, the projected impacts of climate change on three hydrogeomorphic site types – lacustrine, riverine, and barrier-enclosed – are summarized from the literature. Climate change effects on wetland ecosystems will be determined, in part, by how rapidly climate changes and resultant effects on water levels. If changes occur slowly, wetland ecosystems may have more opportunity to adapt but if changes are rapid the adaptive capacity of many wetland species may be exceeded. The degree of water level change will also determine the potential impact on wetland ecosystems. All wetlands by their very nature are able to adapt to some degree of water level change, but long-term significant decreases in water levels (e.g. 50 cm or more) could result in significant changes to the current distribution and abundance of wetlands. Many wetlands are already critically stressed by urbanization, agricultural runoff, and fragmentation and may have little resilience to respond to the new pressures of climate change (Easterling *et al.* 2004).

Of the five wetland vegetation types, marshes are expected to be most adaptable to water level changes due to their requirement for and inherent ability to respond to water level fluctuations. Swamps, due to their location in the highest and driest part of the wetland profile and domination by slow-growing trees, are vulnerable to drying due to water level decreases. The effects on fens, reliant on lake level changes as well as regional groundwater influence, are not as direct and obvious. Coastal wetlands that have developed in the irregular topography of the Precambrian Shield are less likely to have suitable sites and substrates for downslope migration (Mortsch 1998). Coastal wetlands that persist through water level changes will become increasingly important as habitat for wildlife, protection for coastal property against more extreme storms, and sinks for nutrients and sediments as streamflows reflect responses to high precipitation events as well as decreased flows and higher pollutant loads.

Table 2.3 Summary of impacts of climate change on Great Lakes coastal wetland hydrogeomorphic site types

Wetland Site Type (study site examples)	**Major Characteristics**	**Main Impacts of Climate Change**
Lacustrine (e.g. Long Point, Turkey Point, Presqu'ile, South Bay) 	• open to and most affected by Great Lakes, including water level fluctuations, nearshore currents, seiches, and ice scour • wetlands in open and protected bays • varying degrees of organic sediment and vegetation development • bathymetry, gentle to steep slope, dependent on degree of protection from lake effects and geology (ice scour and seiches)	• potential for more exposure to extreme winter storms and less ice protection • aquatic, submergent and emergent vegetation may migrate lakeward with lower levels if suitable sediment, slope, seed banks exist • drier vegetation communities (sedges, grasses and shrubs) expand in current wetland • warmer temperatures may result in vegetation community shifting over decades and centuries, starting with changes in species composition and dominance, if seed access (e.g. corridor, birds) • cumulative stresses may encourage spread of invasive species • loss and contamination from increased demands for dredging • mud flats exposed • less interspersion
Riverine (e.g. Dunnville, Lynde Creek, Hay Bay, Lake St. Clair wetlands) 	• occur near the mouth of tributaries to and connecting channels of the Great Lakes • water quality, inflow and sediment loading are strongly influenced by runoff from the watershed but also affected by the lake • often protected from waves • types include: open to the lake, along connecting channels, behind barrier bars and in delta • steep river bank and river channel, with flat flood plain	• more variable river flooding regimes affect wetland which can lessened influence of lake levels • more sedimentation from more extreme precipitation events causing more erosion upstream; vegetation covered with sediments and fish and wildlife habitat adversely affected • lower flows may increase pollutant concentrations • warmer water temperatures decrease dissolved oxygen • may be able to migrate toward river-mouth as levels decline but dependent on sediment, slope and seed bank • warmer temperatures may result in vegetation community shift over decades and centuries, starting with changes in species composition and dominance • cumulative stresses may encourage spread of invasive species
Barrier-Enclosed 	• occur behind a barrier beach formed by coastal processes • gradual slope but barrier beach is an obstruction to downslope vegetation movement once a particular water level threshold has been reached • generally protected from waves but may be lake-connected during high water periods (or extreme storms) • varying connectivity to lake and influence by lake water levels • includes barrier beach and swale complexes between relic beach ridges with decreasing lake level influence as move landward • more prevalent in lower lakes where more coastal sediments are available	• unable to shift lakeward with lower lake levels so gradual drying of wetland; dominated by meadow, shrub and tree communities with associated shift in diversity, productivity and habitat value • drying may increase risk of fire • shifting coastal processes may alter barrier or re-form a lakeward one • warmer temperatures may result in vegetation community shift over decades and centuries, starting with changes in species composition and dominance, if seed access (e.g. corridor, birds) • warmer water temperatures decrease dissolved oxygen • cumulative stresses may encourage spread of invasive species • wetland area decreases

(Mortsch and Koshida 1996; Mortsch 1998; IPCC 2001b; Hebb 2003; Kling *et al.* 2003; Wilcox 2004; Albert *et al.* 2005)

2.5 Climate Change Adaptation and Mitigation

To date, much of the focus for dealing with human-caused climate change has been mitigation – reducing emissions and increasing sinks of greenhouse gases – to prevent or slow climate change. Doubling of carbon dioxide (CO_2) in the atmosphere is a distinct possibility by 2100 and a tripling or quadrupling by that time may occur depending upon various scenarios of economic development, population growth, and associated greenhouse gas emissions (IPCC 2001b). Impacts of human-caused climate change are likely and adaptation – responding to the impacts of climate change – will need to be undertaken.

Adaptation is defined as "… any adjustment that takes place in natural or human systems in response to actual or expected impacts of climate change, aimed at moderating harm or exploiting beneficial opportunities" (IPCC 2001b). In natural systems, adaptation is autonomous as these systems respond automatically to the stresses or opportunities of changing environmental cues and conditions; humans and human created systems can also respond in a similar manner to economic, social, and environmental cues. However, planned adaptation can also be undertaken by individuals, institutions, governments, and businesses where adaptation is undertaken with an awareness that climate is changing, or is about to change, and purposeful action is needed to return to, maintain, or achieve a desired state. To further that end, climate change information is explicitly considered and acted upon in the management and policy making process. In Chapter 8, three potential planned adaptation strategies are considered. They include technology-based strategies to manage water levels for the benefit of coastal wetlands (e.g. dyking and water level regulation) as well as behaviour management through land use policy as a means to protect wetlands from the impacts of changing water level regimes and human responses.

Although Chapter 8 explores potential adaptations to deal with climate change impacts, it does not explore the capacity to undertake these adaptations and the barriers to successful adaptation. For example, the rate of climate change affects the ability to adapt. Slow, gradual changes allow for impacts to evolve and the formulation of plans and assembly of resources to adapt. Extreme, rapid change challenges the capability of natural and human systems to respond.

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3.0 Vulnerability of Wetland Plant Communities in Great Lakes Coastal Wetlands to Climate-Induced Hydrological Change

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3.1 Assessment of the Hydrological Vulnerability of Wetland Plant Communities in Coastal Wetlands on the Lower Great Lakes

Many ecological functions provided by wetlands are driven by the diversity, distribution, and abundance of aquatic plants. These plants facilitate nutrient cycling, trap sediment to improve water clarity, provide high levels of primary productivity as the foundation for complex food webs, and provide habitat for many invertebrates, fish, amphibians, reptiles, birds, and mammals (Environment Canada 2002). At least 450 species of vascular plants regularly occur in Great Lakes coastal wetlands (Keddy and Reznicek 1986) and 15 plants have been identified as species at risk by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) (COSEWIC 2003). Great Lakes coastal wetland ecosystems rely on the highly dynamic environment at the boundary of land and water which makes them vulnerable to many of the projected hydrologic effects of climate change. This chapter reviews the hydrological vulnerability of selected wetland plant species/communities in coastal wetlands on the lower Great Lakes based on a number of environmental preferences, life history traits, and population parameters. A hydrological vulnerability index is used to compare the vulnerability of coastal wetland plants to climate-induced hydrologic change.

3.1.1 Landscape-scale Processes

Large-scale coastal and fluvial processes affect the distribution of Great Lakes wetlands by influencing sediment erosion and accretion. The resulting shoreline characteristics, landscape topography, bathymetry, and distribution of sediment types ultimately determine coastal wetland distribution and extent. Coastal wetlands typically develop in protected areas with low wave energy, sediment accumulation, and high nutrient supply. Once a wetland is established, these landscape geomorphic features, together with climate, determine the hydrological regime of the wetland, and ultimately, the vegetation that becomes established. Thus, geomorphology affects wetland vulnerability to climate-induced hydrological change.

Most barrier beach wetlands have shallow bathymetric profiles and limited water input from surrounding watersheds (see Chapter 2 for more information). As such, climate-induced hydrological alterations are likely to affect vegetation throughout the entire wetland. Although Quinlan and Mulamoottil (1987) found that the overall extent of three barrier beach wetlands on Lake Ontario did not change with naturally fluctuating lake levels between 1927 and 1983, the structure of plant communities within the marshes was affected significantly. Sedges and grasses dominated the marsh during extended periods of low lake levels while increased interspersion and colonization of emergent plants occurred during periods of high water levels. Furthermore, barrier beaches physically prevent any expansion of wetland vegetation lakeward. In many cases, the lakeward sides of barriers are exposed to high wave energy resulting in mineral substrates (coarse sand and cobble) and steep bathymetry that are also unsuitable to wetland vegetation (Mortsch 1998).

Great Lakes lacustrine wetlands typically have varying degrees of protection from lake processes. Although these wetlands do not have barriers impeding the migration of vegetation communities, offshore slope, wave exposure, or unsuitable substrate may affect plant colonization. For example, in some areas of Lake Erie and Lake St. Clair, the shoreline gradient is very shallow (i.e. elevation changes less than one metre over one kilometre) with fine substrates, which provide conditions that facilitate wetland migration depending on how rapidly water levels rise or fall. In other regions, lacustrine wetland expansion is limited by steep offshore slopes. Submerged aquatic plants may be limited in the extent of migration as they are unlikely to survive in deep water (i.e. 8-12 m) (Meyer *et al.* 1943; Schmid 1965; Sheldon and Boylen 1977; Anderson 1978; Lyon *et al.* 1986). In Precambrian Shield areas of Lake Huron and Lake Superior, wetlands are located on isolated pockets of sediment deposited on bedrock with steep or irregular gradients offshore that are not suitable for wetland development (Mortsch 1998).

The response of riverine wetlands to hydrological impacts of climate change may be more complex than other geomorphic types due to the influence of both coastal (wave action and currents) and riverine (flooding, currents, and ice flow) processes (Crowder *et al.* 1996; Keough *et al.* 1999). These wetlands often occupy flooded river valleys; they have a partial or complete connection with the lake and are also significantly affected by the upstream flow from the watershed draining into the wetland. Increased evapotranspiration due to climate change, will likely lead to reductions in watershed runoff. Less upstream inflow combined with lower lake levels, may encourage vegetation in open drowned river-mouth wetlands to expand towards the river channel and the exposed lake shoreline. Consequently, wetland area may increase but changes in vegetation structure are also likely with less submerged aquatic vegetation and more meadow and shrub species colonizing floodplain areas.

Many wetlands in barred drowned river-mouths may not be able to colonize new areas on the lakeward side of the barrier due to inhospitable substrate and slope. These wetlands may also experience increased periods of disconnection from the lake due to lower water levels on both sides of the river-mouth bars. Increased intensity and frequency of short-term flooding may result in high sediment deposition in the wetlands from surrounding watersheds and elevated turbidity. As a result, seed germination and plant growth may be affected significantly (see Section 3.1.4). Sediment-heavy riverine systems may result in new deltaic wetlands that extend into the lake as lower lake levels facilitate the deposition of sediment into a larger floodplain.

The response of wetlands and plant communities to a changing climate may also be influenced by the rate of hydrological change and anthropogenic stressors. A slow rate of water level change would allow migration and expansion of vegetation and wildlife species, while a sudden water level change could completely overwhelm the adaptive capacity of these communities or result in a significant lag as wetlands gradually establish in new geomorphically suitable areas (Crowder *et al.* 1996; Mortsch 1998). In addition, shoreline hardening is becoming more pronounced across the Great Lakes basin as the population grows, spurring increased shoreline residential development and recreational facilities. Dredging to maintain shipping and recreational boating channels also impacts shoreline processes, while riverine wetland adaptation may be impacted by dam obstructions. These anthropogenic barriers restrict the ability of wetland vegetation to adapt to changing water levels.

Although physical factors (geomorphology) determine the distribution and abundance of the vegetation community during extreme events, such as prolonged low or high water levels (Kadlec 1962; Harris and Marshall 1963; Spence 1982), vegetative plasticity allows some wetland communities to tolerate slow and cyclical environmental change (Crowder *et al.* 1996). For example, during natural low water cycles, wetland communities migrate towards the new shoreline (where bathymetry permits) to maintain their hydrological requirements. Previously submerged wetland areas become shallow standing water and mudflats that allow emergent wetland plant seeds to germinate and expand into these areas. Further, many elevated wetland areas dry allowing facultative wetland plants, such as willow (*Salix* spp.), to expand and encroach into these areas (Keddy and Reznicek 1982, 1986; Bauder 2000). When water levels rise, previously dry wetland areas become flooded resulting in die-off of many emergent and facultative wetland plants. Submerged aquatic plants advance landward into these newly flooded areas while emergent and facultative wetland plants retreat into moist and drier sites. As a result, wetland plant diversity tends to be high in wetlands with fluctuating water levels because plant communities are periodically shifting between dominant species at different elevations (Harris and Marshall 1963; van der Valk and Davis 1978; Keough *et al.* 1999; Mitsch and Gosselink 2000). This ability of wetland communities to migrate is known as vegetative plasticity.

The concepts behind vegetative plasticity have contributed to the development of an index of plant adaptability to habitat change. Oldham *et al.* (1995) ranked native plants in southern Ontario using a "Coefficient of Conservatism" (CC) based on tolerance of various habitat conditions. Since this index examines the range of tolerable growing habitat conditions for most wetland plants, it also provides an assessment of wetland plant tolerance to disturbance. For example, wild rice (*Zizania palustris*) has a high CC because it is an annual plant requiring shallow, organic substrates to grow and is easily uprooted with disturbance. Other wetland plants with specific growing habitat requirements, such as submerged aquatic plants, also tend to have moderate to high CC. In contrast, common reed (*Phragmites australis*) has a low CC due to its ability to grow in both dry and wet habitats, spread very quickly from rhizomes, and quickly exploit disturbed soils (Kiviat 1987; Newmaster *et al.* 1997; Amsberry *et al.* 2000). Wetland plant species that are sensitive to habitat change (i.e. high CC), such as wild rice, and those plants requiring specific habitats to grow, such as submerged aquatic plants, are more vulnerable to hydrological changes due to climate change than low CC plant species, such as common reed.

3.1.2 Plant Structural and Morphological Adaptations

Although a diversity of environmental conditions ranging from fully submerged to moist substrates exist within a wetland, most wetland plants only occur in specific habitat conditions because of growing requirements and/or oxygen and water depth tolerances. Many species have evolved morphological and physiological adaptations that allow them to exploit and dominate specific wetland conditions but limit growth and survival outside of "optimal" growing conditions. For example, many submerged aquatic plants, such as wild celery (*Vallisneria americana*), have leaves with relatively large voluminous cells for increased buoyancy and have no water conserving adaptations, such as stomata or cuticle (Mackie 2001). These characteristics allow submerged aquatic plants to proliferate in aquatic environments. However, optimal wetland conditions for most wetland plants seldom exist for prolonged periods of time because of fluctuating water levels. To survive through such unfavourable conditions, some wetland plants must either reproduce quickly and add seeds or propagules to the existing seed banks, or enter a period of dormancy. Without structural support, stomata, and a cuticle, submerged aquatic plants cannot survive in terrestrial environments, and must enter dormancy if flooded wetland areas dry up (Harris and Marshall 1963; Kelsall and Leopold 2002). All submerged aquatic plants are wetland obligates (i.e. species that only grow only in wetlands) (Oldham *et al.* 1995).

Wetland obligate emergent species, such as cattail (*Typha* spp.), have structural support and water conserving adaptations. These species have developed aerenchyma (air-filled spongy tissue) in roots and rhizomes to facilitate oxygen transfer during flooded anaerobic conditions (Coops *et al.* 1996; Keddy 2000; Mitsch and Gosselink 2000; Mackie 2001). Cattail and bulrush (*Scirpus* spp.) can also temporarily persist for one or two years in water too deep for survival by rapid stem elongation in conjunction with hypertrophy of lenticels (openings in stems and roots that permit gas exchange between internal tissues and the atmosphere) (Lieffers and Shay 1981; Grace 1989; Waters and Shay 1990; Batterson *et al.* 1991). Long, porous, emergent stems facilitate oxygen transfer between roots and the atmosphere and the release of some toxic compounds (Kozlowski 1984). Consequently, these plants can survive in a wide range of hydrological conditions (Squires and van der Valk 1992). Similarly, broad-leaved arrowhead (*Sagittaria latifolia*) tends to grow in shallow standing water but can tolerate submersed conditions by developing leaves that are thin and ribbon-like (Newmaster *et al.* 1997). Prolonged high water levels, however, affect wetland plant communities by reducing wetland plant diversity. High water levels, lasting three or more years eliminate emergent plants in deep water (Kadlec 1962; Harris and Marshall 1963; Dabbs 1971; Millar 1973; Squires and van der Valk 1992; Casanova and Brock 2000).

Other plants are facultative wetland species, such as common reed and willow, which grow in wetland and upland habitats (Oldham *et al.* 1995). These plants are often dominant in areas where they occur because of adaptations such as high silica content and woody structure, which create structural support and reduce dehydration in transitional wetland/terrestrial habitats (Kiviat 1987). Common reed survives in anoxic habitats by transferring nutrients and oxygen through aerenchyma in rhizomes and stems (Kozlowski 1984; Marks *et al.* 1994; Keough *et al.* 1999; Amsberry *et al.* 2000). As a result, common reed grows in both flooded and dry habitats (Kiviat 1987; Haworth-Brockman 1987; Marks *et al.* 1994; Kelsall and Leopold 2002). These adaptations allow facultative wetland plants to out-compete neighbouring wetland obligate plants within transitional habitats by shading, crowding, and inhibiting seed germination (Haslam 1971; Jones and Lehman 1987; Brown 1998; Rice *et al.* 2000).

A plant species' hydrological vulnerability to climate change is determined by the rate and extent of coastal wetland change as well as the dependency of a specific wetland plant for a certain environmental niche. Specifically, the distribution and abundance of submerged aquatic (e.g. wild celery), floating leaved (e.g. yellow pond lily (*Nuphar variegatum*)), and some water dependent emergent plants (e.g. bulrush) may decline while other emergent (e.g. cattail) and facultative wetland plants (e.g. common reed) may expand into moist, or dry, wetland habitat. Wetland plant communities may change from being dominated by wetland obligate plants to facultative wetland plants.

3.1.3 Propagation: Seed Production, Tubers, and Turions

Although morphological and physiological adaptations may temporarily enhance survival during a drought, long-term plant survival is determined by the maintenance of seeds or vegetative propagules such as tubers or turions (a turion is a dense cluster of overwintering leaves) in seed banks or reservoirs (van der Valk 1981; Newmaster *et al.* 1997; Brock and Rogers 1998). Due to differences in energy requirements and desiccation rates, plant dormancy state (e.g. tubers, turions, or seeds) may affect a plant species' hydrological vulnerability to climate change. By entering dormancy as a tuber or seed, wetland plants can survive through years of unfavourable conditions such as a drought. Some tubers and seeds can remain viable in substrates for 20-25 years (O'Neill 1972; Weinhold and van der Valk 1989; Squires and van der Valk 1992; Brock *et al.* 2003). Tubers, however, require more time to develop and may be less resistant to desiccation than seeds. For example, some wetland plants can germinate, grow, and set seed in the first 8-12 weeks of the growing season (Brock and Rogers 1998; Brock *et al.* 2003), while most tubers are not produced until late summer (i.e. in 12-16 weeks) (Newmaster *et al.* 1997). In addition, tubers cannot survive as long as seeds in wetland substrates because of higher desiccation and decomposition rates (Weinhold and van der Valk 1989; Brock and Rogers 1998).

Seed production and viability in wetland plants is species specific. For example, some wetland plants, such as common reed, propagate very little by seed (Haslam 1971; van der Valk 1981) whereas other plants, such as purple loosestrife and cattail, produce enormous quantities of seed that, generally, persist in seed banks for years (van der Valk 1981; Thompson *et al.* 1987; Weinhold and van der Valk 1989). Because seed production is an adaptation that allows plants to escape unfavourable habitat conditions such as low water levels, drought, or winter-kill (Raven *et al.* 1987), the ability of a wetland plant to produce viable seeds affects its hydrological vulnerability to climate change. Van der Valk (1981) concluded that wetland plants that produce vast quantities of viable seed are essentially impossible to eliminate from a wetland because of persistence in seed banks. However, wetland plants that reproduce from propagules may be eliminated from a wetland because colonization ability depends on vegetative spread from surrounding plants. Furthermore, some submerged aquatic plants, particularly perennials, such as coontail (*Ceratophyllum demersum*) and common waterweed (*Elodea canadensis*), depend more on vegetative propagules than seeds for colonization (Capers 2003). If low water levels due to climate change result in the die-off of some wetland plants, plant reproduction from propagules may be greatly diminished. These plants may be more vulnerable to hydrological changes than seed producing wetland plants, such as cattail and northern water milfoil (*Myriophyllum sibiricum*), and spore producing algae (e.g. muskgrass (*Chara vulgaris*)).

3.1.4 Germination and Growth

In addition to a wetland's geomorphic type, fine scale abiotic features affect wetland plant communities. Variation in wetland micro-topography, including changes in slope and substrate (e.g. hummocks), can lead to highly variable soil moisture regimes, even within a small wetland. Consequently, high vegetative diversity may occur as different species have different germination and growth requirements related to hydrological regime. Conditions such as water depth and flood duration (Meyer *et al.* 1943; Sheldon and Boylen 1977; Spence and Dale 1978; Lieffers and Shay 1981; Stanley and Shaw 1986; Hudon 1997), sediment particle size (Hutchinson 1975; Keddy and Constabel 1986; Knapton and Petrie 1999), water quality (e.g. oxygen levels) (Anderson 1978; Wilson and Keddy 1985; Keddy 2000; Mitsch and Gosselink 2000), and turbidity (Duarte *et al.* 1986; Chambers and Kalff 1987; Crowder and Painter 1991; Chow-Fraser *et al.* 1998; Knapton and Petrie 1999) influence seed germination and/or plant growth and survival.

Seed germination may be more important than seed production because without sufficient germination rates seed-producing wetland plant communities cannot be maintained. Water is the most important factor for germination (Raven *et al.* 1987; Kellogg *et al.* 2003). Most wetland plant seeds have species-specific water germinating requirements (Keddy and Ellis 1985). For example, all submerged aquatic and floating leaved plants require seed submersion to germinate (Kelsall and Leopold 2002). Germination requirements of emergent vegetation are variable; some plants require water above the seed surface (e.g. wild rice and giant burreed (*Sparganium eurycarpum*)), or water at or below the seed surface (e.g. cattail and common reed), while others can germinate in all three conditions (e.g. purple loosestrife and pickerel-weed (*Pontederia cordata*)) (Kelsall and Leopold 2002). Low water levels, or drawdowns, typically result in high germination rates for most seeds (Kadlec 1962; Harris and Marshall 1963; van der Valk 1981). However, if the duration of low water levels is extended because of climate change, many wetland plants that require seed submersion, such as wild celery and wild rice, may be replaced by more dry tolerant wetland plant species, such as cattail and pickerel-weed.

Water germination requirements are also affected by seed and soil particle size interactions. The presence of fine sediments within wetland substrates may enhance seed germination of some wetland plants, particularly those with small (e.g. cattail) or flat seeds (e.g. broadleaf arrowhead) (Keddy and Constabel 1986). High surface area to volume ratio may benefit these wetland plants by increasing seed surface contact with available water in fine sediments. The number of intense precipitation events is expected to increase due to climate change, and the resulting runoff and flash-flooding may increase sedimentation rates causing changes in wetland substrate characteristics and wetland water depth. Consequently, the distribution and abundance of some wetland plants that have specific substrate preferences, such as wild celery and common waterweed for organic sediment (Spence 1982; Knapton and Petrie 1999) or muskgrass and sago pondweed (*Potamogeton pectinatus*) for sandy sediment, may change (Hutchinson 1975; Knapton and Petrie 1999).

Despite projected changes in Great Lakes water levels due to climate change, a land-water interface will always occur to allow seed germination. However, current plant distributions will be significantly affected by climate-induced changes in water depth. There are three dominant growth and survival scenarios related to water levels that are possible for wetland plants including: 1) above the sediment surface, 2) at the sediment surface, and 3) below the sediment surface (Kelsall and Leopold 2002). Wetland plants requiring water above the sediment surface are perhaps the most vulnerable to hydrological changes, due to climate change, because of projected increases in the frequency and duration of low water levels (Mortsch 1998; Kling *et al.* 2003).

Many wetland plants require standing water to grow and survive. As previously discussed, submerged aquatic plants lack structural support and have no stomata or cuticle (Mackie 2001). These wetland plants require water above the sediment surface to grow and persist (Harris and Marshall 1963; Anderson 1978; Kelsall and Leopold 2002). Similarly, most emergent wetland plants require water above the sediment surface to grow and survive because of high water loss from reduced stomata. In addition to the occurrence of standing water, water depth also affects wetland plant growth and survival (Meyer *et al.* 1943; Sheldon and Boylen 1977; Lieffers and Shay 1981; Hudon 1997). For example, common waterweed does not grow in water depths less than 0.5 m (Stanley and Shaw 1986) whereas muskgrass may thrive at these depths (Schmid 1965; Hutchinson 1975; Knapton and Petrie 1999). Many emergent wetland plants also achieve maximum growth and survival when water is above the plant surface (Kelsall and Leopold 2002), but have species-specific tolerances to maximum water depth (Harris and Marshall 1963; Spence 1982). Hardstem (*Scirpus acutus*) and softstem bulrush (*Scirpus validus*) require standing water to grow and survive but hardstem bulrush tends to grow in deeper water than softstem bulrush (Dabbs 1971). In contrast to wetland obligates, many facultative wetland plants, such as sedges, only grow in wetland habitats where water is below the sediment surface because of intolerance to anoxic conditions (Kelsall and Leopold 2002).

Flood duration may be more important than flooding occurrence for emergent and facultative wetland plant growth and survival (van der Valk 1981; Kozlowski 1984; Squires and van der Valk 1992; Casanova and Brock 2000). For example, flood duration determines the extent to which a wetland plant must tolerate anoxic conditions. When organic wetland soils become flooded, oxygen is rapidly depleted due to microbial respiration and slower diffusion of molecular oxygen through water than through air (Keddy 2000; Mitsch and Gosselink 2000). Many wetland obligate plants have evolved adaptations for anoxic conditions (Kozlowski 1984; Mitsch and Gosselink 2000) and thus persist in these environments. Conversely, many facultative wetland plants such as sedges (*Carex* spp.) cannot tolerate anoxic conditions because of changes in the uptake of macronutrients (Nitrogen, Phosphorus, and Potassium) and production of growth hormones (Reid and Crozier 1971). Further, the accumulation of some organic compounds, such as ethanol and ethylene, has been known to inhibit growth and survival (Barclay and Crawford 1982; Kozlowski 1984; McKee *et al.* 1989; Keddy 2000; Mitsch and Gosselink 2000). Under a changing climate, flood duration in current wetland areas is likely to be reduced and may become dominated by flash flood events following storm events and less frequent but heavy precipitation. This shorter flood duration, combined with less standing water, could allow facultative wetland plants to survive through brief anoxic conditions and out-compete wetland obligate plants.

An increase in intense precipitation events, due to climate change, with associated higher erosion and entrainment of sediments may affect wetland plant communities by increasing turbidity leading to changes in water clarity. This change may reduce the diversity of submerged aquatic plants. Generally, submerged aquatic plants require water with low turbidity to grow and survive (Duarte *et al.* 1986; Chambers and Kalff 1987; Chow-Fraser *et al.* 1998); however, a plant's actual turbidity tolerance is affected by its growth form. Some submerged aquatic plants, such as Richardson's pondweed (*Potamogeton richardsonii*), produce tall shoots with leaves concentrated near the water surface (Newmaster *et al.* 1997) while other wetland plants, such as fern pondweed (*Potamogeton robbinsii*), produce bottom-dwelling shoots and leaves (Chambers and Kalff 1987). Plants growing closer to the water surface tolerate light-limiting turbid waters better than bottom dwelling plants. In addition, more suspended sediments are likely to settle out of the water column onto plant leaves and stems in the mid and lower water column than on plants growing closer to the water surface affecting the growth and survival of these plants.

3.1.5 Plant Life Span

The ability of a plant to adapt to changes in hydrology is also affected by its life span (i.e. whether it is a perennial or annual plant). Generally, perennial plants live for more than two years while annual plants complete their life cycle in one year (Raven *et al.* 1987; Newmaster *et al.* 1997). Perennial plants can be further divided into long-lived perennials, such as trees and shrubs, and short-lived perennials, such as herbaceous vegetation. Both plant types have traits that make them vulnerable to climate change. For example, plant expansion and contraction may be more extensive in annuals than perennials in response to inter-annual water level fluctuations because of shorter lifespan and quicker ability to produce seeds and grow (Kadlec 1962; van der Valk 1981; Squires and van der Valk 1992). Since perennials can typically survive a wider range of environmental conditions and reproduce in more than one growing season, they may be less vulnerable to long-term hydrological changes than annual plants which germinate, grow, and set seed only during years with favourable conditions (Keddy and Reznicek 1986). Long-lived perennials may be the least vulnerable to hydrological changes because of morphological and physiological adaptations. Generally, long-lived perennials are hardier than short-lived perennials because of their woody structure and deep roots (Raven *et al.* 1987).

3.1.6 Drought Tolerance and Vegetative Spread Rate

Periods of drought are likely to increase in frequency and intensity due to climate change. Wetland plants have different drought tolerances. Some plants survive drought by entering dormancy as seeds or tubers, while others use morphological adaptations, such as the formation of aerenchyma, hypertrophy of lenticels, adventitious and deep running roots, and long rhizomes, to decrease their vulnerability to changes in water availability (Kozlowski 1984; Keough *et al.* 1999). Adventitious roots may allow some wetland plants to survive during low water levels by regenerating root systems (Clemens *et al.* 1978; Kozlowski 1984), or allowing quick relocation into favourable growing environments. Common reed often survives droughts by transporting nutrients and water along lateral rhizomes that often exceed 10 m in length and roots that often penetrate wetland substrates up to one metre in depth (Kiviat 1987; Grace 1989; Marks *et al.* 1994; Amsberry *et al.* 2000). Further, vegetative reproduction allows some wetland plants to quickly expand during favourable growing conditions. This expansion allows these plants to out-compete neighbouring, slower-reproducing wetland plants. Highly-adapted wetland plants are more likely to survive during prolonged low water levels associated with climate change.

3.1.7 Plant Species at Risk or Species with Low Populations

Wetland plants identified as species at risk (SAR) and those species with low populations may be more vulnerable to hydrological changes associated with climate change than wetland plants with moderate or high populations because of a greater risk of extinction. Environmental stochasticity, or a chance event, such as drought or flooding, may have a greater impact on small populations rather than large populations because fewer individuals exist to buffer, or compensate, any change in population size (Caughley and Gunn 1996). In addition, small populations contribute less to source and sink populations. Generally, the viability of a sink population (not self-sustaining) is maintained by emigrating individuals from a source population (growing population) (Ricklefs 1990). SAR species have low populations and likely have a reduced ability to recolonize a wetland area from seeds or propagules due to fewer individuals. Their hydrological vulnerability to climate change is higher than more abundant wetland plant species.

3.1.8 Summary

Climate-induced hydrological changes may affect the distribution, abundance, and composition of wetland plant communities in Great Lakes coastal wetlands. The extent of these impacts will be influenced by the geomorphology of the wetland and the vegetation plasticity of the associated plant communities. Although wetland plants and communities have evolved to compensate for water level fluctuations through vegetative plasticity and vegetation succession, longer and more frequent periods of low water levels are likely to reduce the diversity of wetland plant communities through the expansion of some emergent (e.g. cattail) and facultative wetland plants (e.g. common reed) in all wetland types (Harris and Marshall 1963; van der Valk and Davis 1978; Keddy and Reznicek 1986). An increased frequency of low water levels on the Great Lakes, in conjunction with high sedimentation from flooding due to intense precipitation events, may also limit many wetland obligates, such as submerged aquatic plants, to remaining deep water sections (or "refugia") (Kling *et al.* 2003). The location and extent of these refugia also depend on wetland bathymetry and soil distribution and these areas may not be available in some wetlands if water levels become too shallow. As a result, submerged aquatic plant communities may shift from deep water submerged aquatic plant species (e.g. common waterweed, coontail, wild celery), to shallow submergent plants (e.g. muskgrass and Richardson's pondweed), surface floating plants (e.g. yellow pond lily), and emergent plants (e.g. cattail, common reed, wild rice) (Harris and Marshall 1963; Newmaster *et al.* 1997).

Due to low water levels, the current distribution of coastal wetlands may change, with the upper boundaries of wetlands succeeding to an upland community and emergent plants, such as cattail and common reed, moving into open water habitat and out-competing many submerged aquatic and floating leaved plants (Harris and Marshall 1963; Millar 1973; Poiani and Johnson 1991). Wetland obligate plant diversity, particularly among submerged aquatic and floating leaved plants may decline as facultative wetland plants expand (Mitsch and Gosselink 2000; Wilcox and Meeker 2003).

3.2 Hydrological Vulnerability Index

An assessment of the hydrological vulnerability of key wetland plants was undertaken using criteria that incorporated reproductive and survival requirements for these wetland plant species (Table 3.1). Wetland plants representative of wetland communities found in the lower Great Lakes were selected using data obtained for the Environmental/Wetland Technical Working Group of the IJC LOSLR Study (Wilcox *et al.* 2005). The wetland plant communities were categorized based on their affinity to specific elevations, which represent historically unique hydrologic conditions. The communities included: Open Emergent Marsh (OEM), Emergent Marsh (EMM), Meadow Marsh (MM), Meadow Marsh Transitional (with shrubs) (MMT), and Treed Swamp (TRS). Hydrological vulnerability indices were developed for the ten most abundant wetland plants (more or less depending on total number of species recorded) in each community (as determined from percent cover) as well as wetland plants identified as SAR.

Three wetland plant environmental preferences, that are directly related to plant hydrological requirements, were used to assess the hydrological vulnerability of wetland plants to climate change and included wetness index (i.e. marsh dependency), germination requirements (i.e. seed flooding), and growth and survival requirements (i.e. length of plant inundation). Selected life history traits, such as drought tolerance, life history (annual vs. perennial), and vegetative spread rate that may allow some plants to adapt to projected hydrological changes were also incorporated in the vulnerability assessment. Each plant species was assigned a Coefficient of Conservatism to assess the tolerance of these wetland plants to habitat change (Oldham *et al.* 1995).

Table 3.1 Codes and hydrological vulnerability scores for selected habitat requirements, life history traits, and population parameters used to assess the hydrological vulnerability of selected wetland plants in Great Lakes coastal wetlands

Environmental Preferences and Traits	Hydrological Vulnerability Score
Wetness Index	
OBL = Wetland Obligate; occurs almost always in wetlands >99%	20
FACW+ = Facultative Wetland; usually occurs in wetlands 67-99%	18
FACW = Facultative Wetland; usually occurs in wetlands 67-99%	16
FACW- = Facultative Wetland; usually occurs in wetlands 67-99%	14
FAC+ = Facultative; equally likely to occur in wetland or upland 34-66%	12
FAC = Facultative; equally likely to occur in wetland or upland 34-66%	10
FAC- = Facultative; equally likely to occur in wetland or upland 34-66%	8
FACU = Facultative Upland; occasionally occurs in wetlands 1-33%	6
"+" denotes species with a greater estimated probability of occurring in wetlands	
"-" denotes species with a lower estimated probability of occurring in wetlands	
Germination Requirements	
Type 2 = Water above sediment surface	9
Type 1 = Water at or below sediment surface	7
Type 1 and 2	3
Growth and Survival	
Type A = Water above sediment surface	9
Type D = Water at sediment surface	7
Type B = Water below sediment surface	4
Combinations	2
Drought Tolerance	
None	16
Low	12
Medium	8
High	4
Life History	
ANN = Annual	9
PER = Herbaceous Perennial	6
PERV = Long lived Perennial	3
Vegetative Spread Rate	
None	12
Slow	9
Moderate	6
Rapid	3
Extremely aggressive	0
Coefficient of Conservatism	
-3 = invasive plant	-3
3-30 = growing tolerance of native plant (3 = plants found in a wide variety of growing environments; 30 = plants with a high degree of fidelity to a narrow growing environment)	3 - 30

Maximum possible score (105) and minimum score (15)

Hydrological vulnerability indices were calculated for each selected wetland plant by summing vulnerability scores from a series of environmental preferences, life history traits, and Coefficients of Conservatism. Species' environmental preferences and life history traits were determined from data provided by the *Manual of Vascular Plants of Northeastern United States and Adjacent Canada* (Gleason and Cronquist 1991), *Wetland Plants of Ontario* (Newmaster *et al.* 1997), and *The PLANTS Database* (USDA, NRCS 2002). If data from these references did not categorize a plant into a ranking, species were assigned a rank based on expert opinion. Each environmental preference, life history trait, and Coefficient of Conservatism was subdivided and weighted in relation to the degree of projected hydrological change (Table 3.1).

3.2.1 Wetness Index

This habitat requirement had a high weighting (i.e. score out of 20) because hydrological changes, due to climate, will affect coastal wetland habitat. Wetland plants were grouped according to wetland dependency and were categorized into 1 of 8 rankings, i.e. from Wetland Obligate Plants to Facultative Upland Plants by following Oldham *et al.* (1995) (Table 3.1). Wetland Obligate Plants were defined as those plants found almost exclusively in wetlands (>99% occurrence). Vulnerability of these plants was ranked highest because of their dependency on coastal wetland habitat while Facultative Upland Plants were ranked lowest because of their low relationship to lake water levels.

3.2.2 Germination Requirements

Germination requirements (seeds) were categorized into one of three groups following Kelsall and Leopold (2002). Plant seeds that germinate with water above the surface sediment (Type 2 plants) were considered most vulnerable to hydrological changes because of projected lower water levels. Type 1 plants requiring water at or below sediment surface also had relatively high vulnerability because of their dependence on hydrology or moisture. Overall, this habitat requirement was not considered an important factor for vulnerability because many plants can also propagate asexually.

3.2.3 Growth and Survival Requirements

Growth and survival requirements were ranked into four groups. Those plant species requiring water levels above or at the plant surface (Type A and Type D) were considered most vulnerable to hydrological changes. Plants capable of growing and/or surviving in a combination of water regimes were considered more adaptive to water level changes, and therefore, were assigned the lowest vulnerability.

3.2.4 Drought Tolerance

Overall, climate change scenarios project an increase in air temperatures, higher rates of evapotranspiration, and more dry conditions (Mortsch 1998; Kling *et al.* 2003). Those plants capable of surviving periods of drought will likely be affected less by hydrological changes.

3.2.5 Life History

Plant life cycle was also examined in relation to altered hydrology. Perennial plants were considered less vulnerable to hydrological change due to stored resources and longer reproduction (i.e. reproduce in more than one growing season). To further differentiate the impact of hydrological changes on perennials, long-lived perennials (e.g. deciduous trees) were separated from short-lived perennials (e.g. herbaceous vegetation). Annuals were ranked most vulnerable to hydrological changes due to their relatively short growing life.

3.2.6 Vegetative Spread Rate

The rate of asexual reproduction may also affect a plant's vulnerability to hydrological changes. Plants capable of rapid vegetative spread may continue to grow in inundated habitats where seeds cannot persist. Similarly, rapid asexual reproduction may allow some plants to colonize habitats faster than those plants incapable of vegetative spread. Therefore, plants only spreading from seed were considered most vulnerable to hydrological changes.

3.2.7 Coefficient of Conservatism

Coefficient of Conservatism for each plant species was obtained from Oldham *et al.* (1995). These scores were based on a plant's conformity to a range of parameters and susceptibility to disturbance. Plants growing in a wide variety of communities and tolerant of disturbance received the lowest vulnerability score. Overall, this coefficient was considered the most important factor relative to other traits and requirements because projected hydrological changes may result in altered coastal wetland habitats and affect wetland plants with narrow ecological niches the most.

From these species-specific scores, an overall hydrological vulnerability index for each plant was calculated and plotted (Table 3.2; Figure 3.1). Risk categories were assigned based on defined plateaus and inflection points. Plants with scores between 80 and 105 were identified as "High Risk" species vulnerable to hydrologic alterations. "Moderate Risk" species scored between 65 and 79 and "Low Risk" species ranged between 0 and 64.

Table 3.2 Wetland plant habitat requirements, life history traits, population parameters, and hydrologic vulnerability index scores of selected wetland plants in Great Lakes coastal wetlands (see Table 3.1 for codes and scoring)

Common Name	Scientific Name	Wetness Index		Germination Requirements		Growth and Survival		Drought Tolerance		Life History		Vegetative Spread Rate		Coefficient of Conservatism		Hydrological
		Code	Rank	Code	Rank	Code	Rank	Code	Rank	Code	Rank	Code	Rank	Code	Rank	Vulnerability
Wild Rice	*Zizania palustris*	OBL	20	Type 2	9	Type A	9	None	16	ANN	9	None	12	9	27	**102**
Hill's Pondweed	*Potamogeton hillii*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Slow	9	8	24	**93**
Horsetail Spike-Rush	*Eleocharis equisetoides*	OBL	20	Type 1 or 2	3	Type A	9	None	16	PER	6	Slow	9	10	30	**93**
Crimsoneyed Rose Mallow	*Hibiscus moscheutos*	OBL	20	Type 1	7	Type D/B	2	None	16	ANN	9	None	12	9	27	**93**
Lesser Panicled Sedge	*Carex diandra*	OBL	20	Type 1	7	Type D	7	None	16	PER	6	None	12	7	21	**89**
Northern Water Milfoil	*Myriophyllum sibiricum*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Slow	9	6	18	**87**
Fern Pondweed	*Potamogeton robbinsii*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Moderate	6	7	21	**87**
Pickerelweed	*Pontederia cordata*	OBL	20	Type 1 or 2	3	Type A	9	None	16	PER	6	None	12	7	21	**87**
Slender Naiad	*Najas flexilis*	OBL	20	Type 2	9	Type A	9	None	16	ANN	9	Slow	9	5	15	**87**
Wire Sedge	*Carex lasiocarpa*	OBL	20	Type 2	9	Type A	9	Low	12	PER	6	Moderate	6	8	24	**86**
Roundstem False Foxglove	*Agalinis gattingeri*	FACW	16	Type 1	7	Type B	4	Medium	8	ANN	9	None	12	10	30	**86**
Sago Pondweed	*Potamogeton pectinatus*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	None	12	4	12	**84**
White Water Lily	*Nymphaea odorata*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Slow	9	5	15	**84**
Water Celery	*Vallisneria americana*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Moderate	6	6	18	**84**
Yellow Pond Lily	*Nuphar variegatum*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Slow	9	4	12	**81**
Common Bladderwort	*Utricularia vulgaris*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Slow	9	4	12	**81**
Richardson's Pondweed	*Potamogeton richardsonii*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Moderate	6	5	15	**81**
Flat-Stemmed Pondweed	*Potamogeton zosteriformis*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Moderate	6	5	15	**81**
Slender Pondweed	*Potamogeton pusillus*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Moderate	6	5	15	**81**
Star Duckweed	*Lemna trisulca*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Moderate	6	4	12	**78**
Leafy Pondweed	*Potamogeton foliosus*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Moderate	6	4	12	**78**
Tussock Sedge	*Carex stricta*	OBL	20	Type 1	7	Type A	9	Low	12	PER	6	None	12	4	12	**78**
Floating-Leaved Pondweed	*Potamogeton natans*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Rapid	3	5	15	**78**
Large-Leaved Pondweed	*Potamogeton amplifolius*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Rapid	3	5	15	**78**
Swamp Candles	*Lysimachia terrestris*	OBL	20	Type 1	7	Type A	9	Low	12	PER	6	Moderate	6	6	18	**78**
Black Ash	*Fraxinus nigra*	FACW+	18	Type 1	7	Type B	4	Low	12	PERV	3	None	12	7	21	**77**
Bebb's Sedge	*Carex bebbii*	OBL	20	Type 1	7	Type D	7	None	16	PER	6	None	12	3	9	**77**
Curly White Water Crowfoot	*Ranunculus longirostris*	OBL	20	Type 2	9	Type A	9	Low	12	PER	6	Moderate	6	5	15	**77**
Stiff Arrowhead	*Sagittaria rigida*	OBL	20	Type 1 or 2	3	Type A/D	2	None	16	PER	6	None	12	6	18	**77**
Beaked Sedge	*Carex utriculata*	OBL	20	Type 1	7	Type D/B	2	Low	12	PER	6	Slow	9	7	21	**77**
Hardstem Bulrush	*Scirpus acutus*	OBL	20	Type 2	9	Type A	9	Medium	8	PER	6	Moderate	6	6	18	**76**
Water Plantain	*Alisma plantago-aquatica*	OBL	20	Type 1 or 2	3	Type A	9	None	16	PER	6	None	12	3	9	**75**
Broad-Leaved Arrowhead	*Sagittaria latifolia*	OBL	20	Type 1	7	Type A/D	2	None	16	PER	6	None	12	4	12	**75**
Coontail	*Ceratophyllum demersum*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Rapid	3	4	12	**75**
Water Smartweed	*Polygonum amphibium*	OBL	20	Type 1	7	Type A/B/D	2	None	16	PER	6	Slow	9	5	15	**75**
American Bluehearts	*Buchnera americana*	FAC-	8	Type 1	7	Type B	4	Medium	8	PER	6	None	12	10	30	**75**
Common Burreed	*Sparganium emersum*	OBL	20	Type 2	9	Type A/D	2	None	16	PER	6	Moderate	6	5	15	**74**
River Bulrush	*Scirpus fluviatilis*	OBL	20	Type 1	7	Type B/D	2	Low	12	PER	6	Moderate	6	7	21	**74**
Common Buttonbush	*Cephalanthus occidentalis*	OBL	20	Type 1	7	Type B/D	2	Medium	8	PERV	3	None	12	7	21	**73**
Lakebank Sedge	*Carex lacustris*	OBL	20	Type 1 or 2	3	Type D	7	None	16	PER	6	Moderate	6	5	15	**73**
Tall Manna Grass	*Glyceria grandis*	OBL	20	Type 1	7	Type B	4	Low	12	PER	6	Slow	9	5	15	**73**
American Water Willow	*Justica americana*	OBL	20	Type 1 or 2	3	Type A/D	2	Low	12	PER	6	Rapid	3	9	27	**73**
Large-Fruited Burreed	*Sparganium eurycarpum*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Rapid	3	3	9	**72**
Fowl Meadow Grass	*Poa palustris*	FACW+	18	Type 1	7	Type D/B	2	Low	12	PER	6	None	12	5	15	**72**
Softstem Bulrush	*Scirpus validus*	OBL	20	Type 1 or 2	3	Type A	9	None	16	PER	6	Rapid	3	5	15	**72**
Jewelweed	*Impatiens capensis*	FACW	16	Type 1	7	Type B	4	Low	12	ANN	9	None	12	4	12	**72**
Speckled Alder	*Alnus incana*	OBL	20	Type 1	7	Type B/D	2	Low	12	PERV	3	Slow	9	6	18	**71**
Woolgrass Bulrush	*Scirpus cyperinus*	OBL	20	Type 1	7	Type B/D	2	Low	12	PER	6	None	12	4	12	**71**
Common Three-Square Bulr	*Scirpus pungens*	OBL	20	Type 1	7	Type A/D	2	Low	12	PER	6	Moderate	6	6	18	**71**
Cyperus-Like Sedge	*Carex pseudo-cyperus*	OBL	20	Type 1	7	Type B/D	2	Low	12	PER	6	Moderate	6	6	18	**71**
Broad-Leaved Cattail	*Typha latifolia*	OBL	20	Type 1	7	Type A	9	None	16	PER	6	Rapid	3	3	9	**70**
Silver Maple	*Acer saccharinum*	FACW	16	Type 1	7	Type B	4	Low	12	PERV	3	None	12	5	15	**69**
Withe Rod	*Viburnum cassinoides*	FACW	16	Type 1	7	Type B/D	2	Medium	8	PERV	3	None	12	7	21	**69**
Lesser Duckweed	*Lemna minor*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Rapid	3	2	6	**69**
Tickle Grass	*Agrostis scabra*	FAC	10	Type 1	7	Type B	4	Low	12	PER	6	None	12	6	18	**69**
Fox Sedge	*Carex vulpinoidea*	OBL	20	Type 1	7	Type D/B	2	Low	12	PER	6	None	12	3	9	**68**
Greater Duckweed	*Spirodela polyrhiza*	OBL	20	Type 2	9	Type B/D	2	None	16	PER	6	Rapid	3	4	12	**68**
Common Waterweed	*Elodea canadensis*	OBL	20	Type 2	9	Type B/D	2	None	16	PER	6	Rapid	3	4	12	**68**
Canada Bluejoint	*Calamagrostis canadensis*	OBL	20	Type 1	7	Type A/B	2	Low	12	PER	6	Slow	9	4	12	**68**
Curly Pondweed	*Potamogeton crispus*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Slow	9	-1	-3	**66**
Narrow-Leaved Cattail	*Typha angustifolia*	OBL	20	Type 1	7	Type A	9	Low	12	PER	6	Rapid	3	3	9	**66**
Narrow-Leaved Meadowswee	*Spiraea alba*	FACW+	18	Type 1	7	Type B	4	Low	12	PERV	3	None	12	3	9	**65**
Steeple-Bush	*Spiraea tomentosa*	FACW	16	Type 1	7	Type D	7	Medium	8	PERV	3	Slow	9	5	15	**65**
Common Elderberry	*Sambucus canadensis*	FACW-	14	Type 1	7	Type B	4	Medium	8	PERV	3	None	12	5	15	**63**
Cattail Hybrid	*Typha glauca*	OBL	20	Type 1	7	Type A	9	Low	12	PER	6	Extremely aggressive	0	3	9	**63**
Nannyberry	*Viburnum lentago*	FAC+	12	Type 1	7	Type B	4	Low	12	PERV	3	None	12	4	12	**62**
Fringed Loosestrife	*Lysimachia ciliata*	FACW	16	Type 1	7	Type B/D	2	Low	12	PER	6	Moderate	6	4	12	**61**
Sandbar Willow	*Salix exigua*	OBL	20	Type 1	7	Type D	7	Medium	8	PERV	3	Moderate	6	3	9	**60**
Red Maple	*Acer rubrum*	FAC	10	Type 1	7	Type B	4	Low	12	PERV	3	None	12	4	12	**60**
Eurasian Water Milfoil	*Myriophyllum spicatum*	OBL	20	Type 2	9	Type A	9	None	16	PER	6	Rapid	3	-1	-3	**60**
Eastern Woodland Sedge	*Carex blanda*	FAC	10	Type 1	7	Type B	4	Low	12	PER	6	None	12	3	9	**60**
Green Ash	*Fraxinus pennsylvanica*	FACW	16	Type 1	7	Type B	4	Medium	8	PERV	3	None	12	3	9	**59**
Dwarf Raspberry	*Rubus pubescens*	FACW+	18	Type 1	7	Type B	4	Medium	8	PERV	3	Moderate	6	4	12	**58**
Slender Willow	*Salix petiolaris*	FACW+	18	Type 1	7	Type B/D	2	Medium	8	PERV	3	Slow	9	3	9	**56**
Canada Anemone	*Anemone canadensis*	FACW	16	Type 1	7	Type B	4	Medium	8	PER	6	Moderate	6	3	9	**56**
Common Frogbit	*Hydrocharis morsus-ranae*	OBL	20	Type 2	9	Type A/D	2	None	16	PER	6	Rapid	3	-1	-3	**53**
Reed Canary Grass	*Phalaris arundinacea*	FACW+	18	Type 1	7	Type D	7	Low	12	PER	6	Rapid	3	0	0	**53**
Red-Osier Dogwood	*Cornus stolonifera*	FACW	16	Type 1	7	Type B/D	2	Medium	8	PERV	3	Slow	9	2	6	**51**
Flowering Rush	*Butomus umbellatus*	OBL	20	Type 1 or 2	3	Type B/D	2	None	16	PER	6	Moderate	6	-1	-3	**50**
Common Reed	*Phragmites australis*	FACW+	18	Type 1	7	Type D	7	Low	12	PER	6	Extremely aggressive	0	0	0	**50**
Gray Dogwood	*Cornus racemosa*	FACW-	14	Type 1	7	Type B/D	2	Medium	8	PERV	3	Slow	9	2	6	**49**
Glossy Buckthorn	*Frangula alnus*	FAC+	12	Type 1	7	Type B	4	Medium	8	PERV	3	None	12	-1	-3	**43**
Boxelder	*Acer negundo*	FACW-	14	Type 1	7	Type B	4	High	4	PERV	3	Slow	9	0	0	**41**
Purple Loosestrife	*Lythrum salicaria*	OBL	20	Type 1 or 2	3	Type B/D	2	Medium	8	PER	6	Rapid	3	-1	-3	**39**
Crack Willow	*Salix fragilis*	FAC+	12	Type 1	7	Type B/D	2	Medium	8	PERV	3	Slow	9	-1	-3	**38**
Wild Red Raspberry	*Rubus idaeus*	FACW-	14	Type 1	7	Type B	4	High	4	PERV	3	Rapid	3	0	0	**35**
Multiflora Rose	*Rosa multiflora*	FACU	6	Type 1	7	Type B	4	High	4	PERV	3	Slow	9	-1	-3	**30**

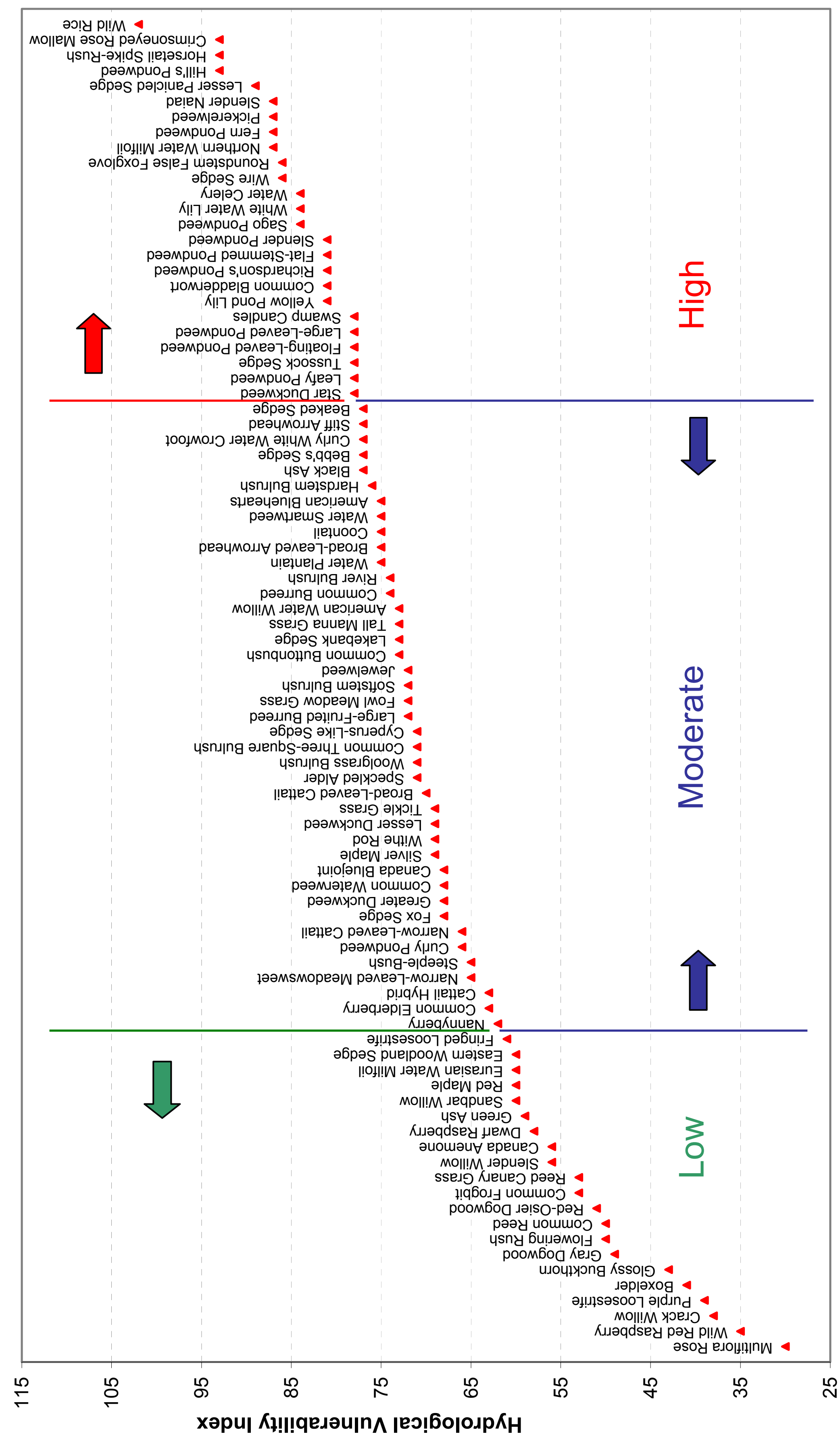

Figure 3.1 Hydrological vulnerability indices for selected wetland plant species growing in coastal wetlands of the lower Great Lakes

3.2.8 Summary

Obligate wetland plant species, with limited drought tolerance and modes of colonization were identified as most vulnerable to changing hydrology due to climate change. Although wetland plant communities depend on a certain degree of natural hydrologic variability to maintain plant diversity (Keddy 2000), too much variability creates a hydrologic environment that only supports species that can colonize and mature quickly (Wilcox and Meeker 1991). For example, many submerged aquatic species such as lilies, wild rice, and pickerelweed require clear shallow standing water throughout the growing season for growth and reproduction. The potential for fewer, but more intense precipitation events due to climate change increases turbidity and variability in growing season water depths, and reduces the hydrologically suitable area for these species.

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4.0 COASTAL WETLAND VEGETATION COMMUNITY RESPONSE TO CLIMATE CHANGE

Andrea Hebb, Elizabeth Snell, and Geniene Sabila

Water level fluctuations are an important determinant of wetland vegetation dynamics in coastal wetlands along the Great Lakes. In order to understand wetland vegetation response to water level fluctuations, a spatiotemporal analysis was completed. A wetland response model was also developed to simulate historical wetland vegetation change and predict future responses under climate change. This chapter discusses methods that were implemented to assess the effects of water level fluctuations on wetland vegetation communities along the shores of Lakes Huron, Erie, and Ontario and reports on the results of the spatiotemporal analysis. The last section of the chapter describes the development and assessment of the wetland vegetation response model, which is later applied to simulate future wetland change under climate change scenarios in Chapter 7 (Figure 4.1).

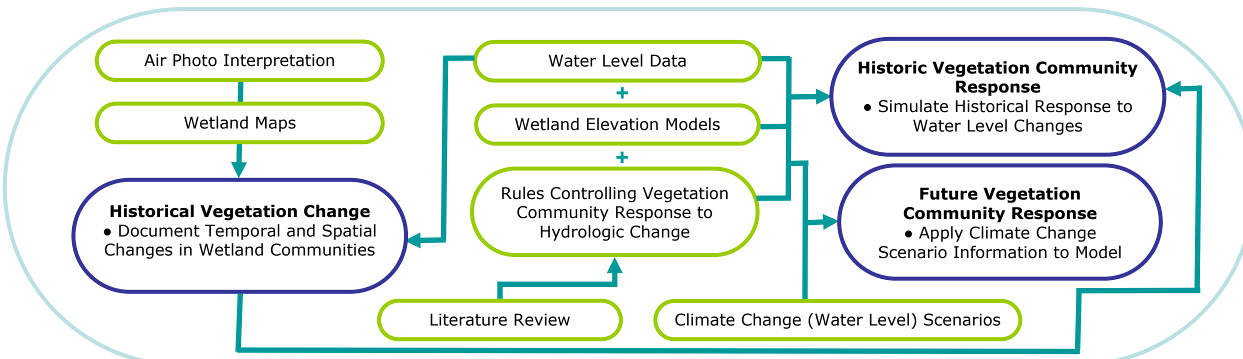

Figure 4.1 Flow diagram of the approach used to evaluate the vulnerability of wetland vegetation communities and potential response to climate-induced hydrological change

4.1 WETLAND VEGETATION DATABASE DEVELOPMENT

This section outlines the development of the wetland vegetation database from collection and interpretation of aerial photographs, to creation of the spatial dataset, and finally, to the statistical and geographical techniques used to quantify wetland vegetation response over time and in relation to water level fluctuations.

4.1.1 Air Photo Interpretation and Spatial Dataset Development

The wetland vegetation database was developed by interpreting study site aerial photographs obtained from the Ontario Ministry of Natural Resources (OMNR), several Conservation Authorities, and the National Air Photo Library, Natural Resources Canada (NRCan). Historical air photo interpretation provided a long time series of historical land resource information. Air photos representing a range of Great Lakes water levels from 1927 to 2001 were interpreted and analyzed for vegetation composition using a standardized vegetation classification and key (Appendix 4.1). Low and high water level periods were represented by data from the 1930's and 1960's and the 1970's and 1980's, respectively.

To adequately assess historical trends within wetlands, the date, scale, and quality of the aerial photography should remain consistent between years. Photographs taken prior to 1978 were typically captured at a lower resolution than later years, but the level of detail was still acceptable for estimating and analyzing general vegetation trends over time (Snell and Cecile Environmental Research 2001). Although air photo were generally taken during the growing season, they varied by a few months across the available years. This variation was acceptable for estimating and analyzing general vegetation trends using a simplified wetland classification scheme (Section 4.1.2). A summary of the historical wetland vegetation data is provided in Appendix 4.2.

Most of the interpreted wetland maps for Lakes Huron and Erie were scanned as tiff images, converted to raster grids, then digitally vectorized into a GIS using the semi-automatic tracing feature of *ArcTools*, an interactive button and menu interface in the *ARC/INFO Workstation* of *ArcGIS 8.3* (ESRI 2003). The interpreted wetland map for Presqu'ile 1931 was also scan vectorized. Select years for Rondeau, Turkey Point, and Presqu'ile were manually digitized. The 1999 data for Long Point and Turkey Point were interpreted and manually digitized by Bird Studies Canada. The remaining wetland sites on Lake Ontario (Hay Bay, Lynde Creek, and South Bay - all years) and Presqu'ile 1999 were interpreted and digitized using on-screen digitizing techniques in *ARC/INFO* of *ArcGIS 8.1* (ESRI 2001). All wetland coverages were edited and polygon topology was added; unique codes representing the various wetland vegetation communities and land use categories were assigned to each polygon in the coverages. The coverages were registered to real world coordinates in UTM, NAD83.

4.1.2 Vegetation Classification

Although the initial vegetation interpretations occurred at a much more detailed class level (Appendix 4.1), a simplified wetland vegetation and land use classification scheme was used to maximize interpretation consistency and comparability for modelling (Table 4.1). The simplified wetland vegetation classes represented dominant communities documented within the study area and classes with similar responses to water level fluctuations, based on literature, field surveys, and recommendations from wetland specialists. All other vegetation and land use categories, including dyked/inland/disconnected wetlands, wooded upland, and all non-wetland areas such as built-up, residential, industrial park, and agricultural land, were masked out of the analysis. The simplified coverages were converted into grids with a 10-m cell size for analysis.

Table 4.1 Simplified wetland vegetation classes (from wettest to driest)

Wetland Class	Description	Lake
Water (W)	Lake and open water (submergents are possible), rivers, ponds	All
Emergent/Floating Mixed (EF)	Flat emergents, *Lemna*, *Nuphar, Zizania*, *Nelumbo*	All
Emergent (E)	Includes a mixture of flat/wet emergents and tall dense dry emergents such as *Typha*, *Pontederia*, *Scirpus*, *Juncus*, *Cephalanthus*, Phragmites, grass/sedge hummocks, trees and shrubs in water, wet meadow	All
Fen (F)	Wet fen, water evident; fen, can include small shrubs	Huron
Sparse Fen with Sand (Fx)	Sparse fen with sand showing	Huron
Fen with Scattered Trees (Ft)	Fen with scattered trees or shrub covered; includes *Thuja*, *Larix*, *Juniper*	Huron
Alvar (A)	Dry alvar or alvar-like; rock with some low vegetation; moist alvar or alvar-like meadow	Huron
Meadow Marsh (M)	Includes *Bidens*, *Poaceae,* and *Cyperaceae*, poplar seedlings, grass/sedge without hummocks	All
Treed/Shrub (T)	Includes trees, shrubs, scattered trees, swamp (moist soil), and possibly upland forest habitats (in the modelling exercise)	All
Exposed Substrate (Ex)	Flood deposited sediment, sand, rock	All

4.1.3 Spatiotemporal Analysis

A basic premise in landscape ecology is that elements within a landscape are strongly influenced by ecological processes. The ability to quantify landscape structure, or spatial relationships among distinct elements in the landscape, is a prerequisite for studying landscape function and change. *FRAGSTATS* (McGarigal *et al.* 2002) is a recognized tool to quantitatively analyze relationships between landscape elements. This analysis considered the proportion and spatial distribution of ten wetland classes to determine trends in wetland vegetation area and relationships between vegetation and land cover distributions to water level fluctuations. A variety of metrics, both at the class and landscape level, were computed to measure changes in wetland composition and configuration over time, and relate these changes to historic water level fluctuations. In addition to characterizing historical wetland response to water level changes, the spatiotemporal analysis results were used to assess the wetland vegetation response model developed in Section 4.3.

Landscape is defined as an area containing a mosaic of patches; in this study, the landscape is the extent of each wetland study site. Class refers to a group of patches with similar characteristics (i.e. patches classified as the same wetland class), and patch is defined as the basic element of the landscape defined by a particular phenomenon, or in this case a single unit of contiguous cells defined by the simplified wetland classification (McGarigal *et al.* 2001). Patch level metrics were not computed in this analysis because they did not provide any interpretive value in analyzing trends of wetland vegetation community abundance and distribution. The metrics were grouped into six descriptors of landscape structure and composition: 1) area, density, and edge; 2) shape; 3) isolation and proximity; 4) contrast; 5) contagion and interspersion; and 6) diversity. A description of the metrics used in the analysis is provided in Appendix 4.3 and input weighting schemes used for the analysis are provided in Appendix 4.4. Correlation coefficients were calculated to determine the relationship between the metrics and mean lake levels for each year; coefficients ranging from -1.0 and -0.7 (negative relationship) or from 0.7 to 1.0 (positive relationship) indicated strong relationships between the metrics and water levels.

In addition to the temporal trend analysis in *FRAGSTATS*, spatial trends were documented through visual observations of changes in wetland community distribution between years of historical data.

4.2 Results of the Spatiotemporal Trend Analysis of Historical Wetland Change

The results of the spatiotemporal trend analysis indicate several key changes in spatial distribution and pattern of wetland communities within the study sites. Discussion of the results focuses on wetland vegetation response to low water levels to provide an understanding of wetland vulnerability to future low water levels projected under climate change scenarios. Analysis results are discussed by lake.

4.2.1 Lake Ontario (regulated water levels, marshes)

The four study sites on Lake Ontario were classified as marsh dominant wetlands. The most notable changes in wetland composition and configuration were documented in response to water level fluctuations prior to the beginning of water level regulation in 1959. Landscape metrics for the drowned river-mouth (riverine) wetlands, Hay Bay (Figure 4.2; Table 4.2) and Lynde Creek (Figure 4.3; Table 4.3), exhibited the greatest correlation with water levels. Metrics for Presqu'ile (Figure 4.4; Table 4.4) and South Bay (Figure 4.5; Table 4.5), protected embayment wetlands, exhibited less correlation but generally showed similar trends. Metrics at the class level were also well correlated with water level fluctuations at Hay Bay; trends at the other Lake Ontario wetlands were not as evident and varied with wetland community (Appendix 4.5).

There were several key changes in wetland composition and configuration in response to declining water levels. First and foremost, as water levels declined, vegetation in the wetlands tended towards drier communities. Areas of open water and emergent/floating mixed vegetation were replaced by the lakeward migration of drier vegetation communities resulting in an overall increase in total vegetated wetland area. There were notable increases in emergent and meadow marsh, and to a lesser extent, treed/shrub vegetation in the Lake Ontario wetlands. Furthermore, substrate in the Lynde Creek delta was exposed in shallow areas during the low water period in 1959.

Generally, the wetlands were less fragmented and complex during drier years. The shapes of patches in the wetlands and vegetation communities were simpler as drier wetland communities expanded, forming larger, solid, and continuous patches of vegetation. At Presqu'ile, however, patches of meadow marsh were actually more fragmented and complex in shape, likely due to the complex arrangement of fingers and sand spits in the wetland and the distribution of meadow marsh vegetation along these fingers.

There was less interspersion in the wetlands as patches of similar wetland vegetation communities were more aggregated in distribution and the amount of edge contrast decreased. Within the wetlands, neighbourhoods were occupied by more patches of the same wetland community or located closer to or beside similar wetland communities. There was also less diversity noted in the Lake Ontario riverine wetlands as water levels declined. The proportion, distribution, and abundance of area among different wetland communities were more uneven, indicating that the wetland area was dominated by a fewer number of wetland communities. For example, at Hay Bay, emergent vegetation was the predominant wetland community during low water years. The findings of the spatiotemporal analysis for the Lake Ontario wetlands are summarized in Table 4.6.

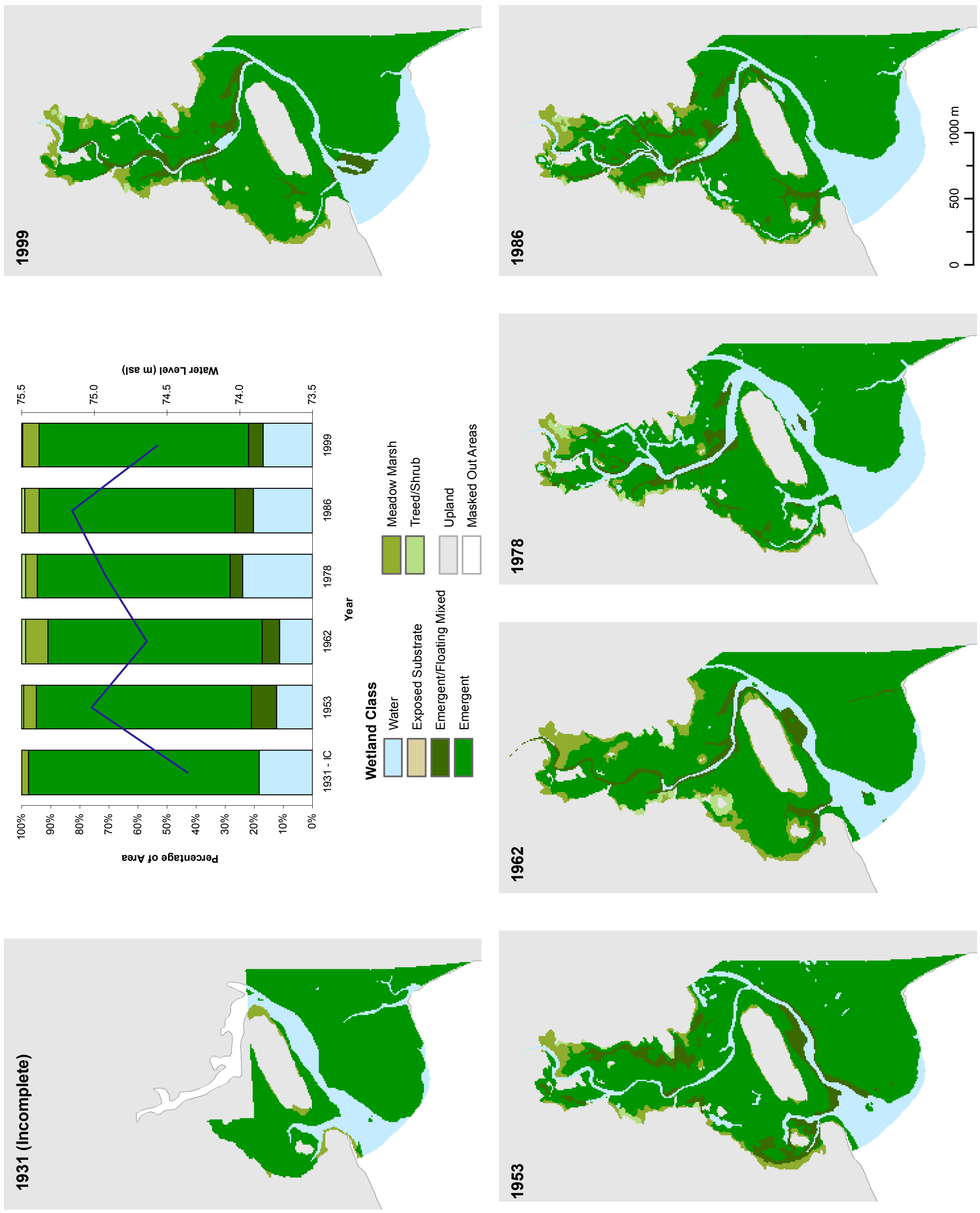

Figure 4.2 Historical distribution of wetland vegetation at Hay Bay, 1931-1999

Table 4.2 Landscape metrics for Hay Bay, 1931-1999

Landscape Metric	Code	Year and Water Level (m asl)						Correl.	Trend w/
		1931^{x}	1953	1962	1978	1986	1999	Coeff.*	↓ Water
		74.36	*75.02*	*74.64*	*74.92*	*75.15*	*74.57*		
AREA, DENSITY, EDGE									
Total Area^ (ha)	TA	138.07	235.49	238.63	204.34	213.97	223.67	-0.38	Increases
Number of Patches (#)	NP	28	118	74	160	176	92	**0.85**	Decreases
Patch Density (#/100 ha)	PD	16.59	43.85	27.50	59.46	65.41	34.19	**0.85**	Decreases
Patch Area - Mean (m)	AREA_MN	6.03	2.28	3.64	1.68	1.53	2.92	**-0.85**	Increases
Patch Area - Area-Weighted Mean (m)	AREA_AM	67.16	70.03	50.84	54.15	49.18	50.79	0.33	Decreases
Patch Area - Median (m)	AREA_MD	0.09	0.08	0.17	0.06	0.04	0.08	-0.66	Increases
Patch Area - Range (m)	AREA_RA	95.60	96.30	92.61	82.81	79.86	78.39	0.05	Decreases
Patch Area - Standard Deviation (m)	AREA_SD	19.20	12.43	13.10	9.39	8.54	11.83	-0.65	Increases
Patch Area - Coefficient of Variation (%)	AREA_CV	318.48	545.04	360.31	558.53	558.27	404.52	**0.91**	Decreases
Radius of Gyration - Mean (m)	GYRATE_MN	64.20	40.38	61.93	31.14	31.75	48.47	**-0.80**	Increases
Radius of Gyration - Area-Weighted Mean (m)	GYRATE_AM	372.92	418.39	346.93	398.13	364.81	353.76	0.58	Decreases
Radius of Gyration - Median (m)	GYRATE_MD	12.88	11.40	23.93	10.51	9.13	15.23	**-0.77**	Increases
Radius of Gyration - Range (m)	GYRATE_RA	422.14	639.08	477.63	657.86	577.61	434.82	**0.80**	Decreases
Radius of Gyration - Standard Deviation (m)	GYRATE_SD	115.56	91.24	104.53	74.66	70.49	91.00	**-0.72**	Increases
Radius of Gyration - Coefficient of Variation (%)	GYRATE_CV	180.00	225.99	168.79	239.75	222.03	187.76	**0.80**	Decreases
Total Edge Length (m)	TE	11080	39130	32890	41720	45630	32380	**0.95**	Decreases
Edge Density (m/ha)	ED	65.65	145.42	122.23	155.04	169.57	120.33	**0.95**	Decreases
SHAPE									
Landscape Shape Index	LSI	4.26	8.97	8.02	9.36	9.95	7.94	**0.95**	Decreases
Perimeter-Area Ratio - Mean	PARA_MN	2377.23	2388.38	1948.48	2566.06	2541.08	2339.74	0.67	Decreases
Perimeter-Area Ratio - Area-Weighted Mean	PARA_AM	196.95	364.64	318.26	383.89	412.95	314.47	**0.95**	Decreases
Perimeter-Area Ratio - Median	PARA_MD	2533.33	2342.86	1750.00	2666.67	2619.05	2165.61	**0.76**	Decreases
Perimeter-Area Ratio - Range	PARA_RA	3910.47	3910.92	3919.46	3904.37	3912.11	3913.01	-0.47	Increases
Perimeter-Area Ratio - Standard Deviation	PARA_SD	1406.78	1231.94	1197.58	1169.37	1154.09	1274.92	-0.65	Increases
Perimeter-Area Ratio - Coefficient of Variation (%)	PARA_CV	59.18	51.58	61.46	45.57	45.42	54.49	**-0.78**	Increases
Shape Index - Mean	SHAPE_MN	1.70	1.63	1.91	1.56	1.58	1.77	**-0.82**	Increases
Shape Index - Area Weighted Mean	SHAPE_AM	2.66	4.58	3.21	4.42	4.14	3.38	**0.84**	Decreases
Shape Index - Median	SHAPE_MD	1.27	1.32	1.62	1.28	1.25	1.41	**-0.79**	Increases
Shape Index - Range	SHAPE_RA	3.43	6.88	6.91	6.13	5.34	4.80	0.11	Decreases
Shape Index - Standard Deviation	SHAPE_SD	0.94	1.04	1.14	0.90	0.94	0.98	-0.43	Increases
Shape Index - Coefficient of Variation (%)	SHAPE_CV	55.60	64.09	59.52	57.27	59.32	55.30	0.54	Decreases
Related Circumscribing Circle - Mean	CIRCLE_MN	0.604	0.614	0.669	0.602	0.610	0.638	**-0.77**	Increases
Related Circumscribing Circle - Area-Weighted Mean	CIRCLE_AM	0.663	0.749	0.747	0.769	0.770	0.742	**0.75**	Decreases
Related Circumscribing Circle - Median	CIRCLE_MD	0.618	0.647	0.702	0.620	0.618	0.682	**-0.87**	Increases
Related Circumscribing Circle - Range	CIRCLE_RA	0.573	0.660	0.714	0.649	0.647	0.592	0.08	Decreases
Related Circumscribing Circle - Standard Deviation	CIRCLE_SD	0.206	0.188	0.178	0.190	0.191	0.200	-0.02	Increases
Related Circumscribing Circle - Coefficient of Variation (%)	CIRCLE_CV	34.12	30.53	26.58	31.51	31.36	31.27	0.46	Decreases
Contiguity Index - Mean	CONTIG_MN	0.384	0.378	0.483	0.333	0.336	0.391	-0.69	Increases
Contiguity Index - Area-Weighted Mean	CONTIG_AM	0.945	0.898	0.911	0.893	0.885	0.912	**-0.95**	Increases
Contiguity Index - Median	CONTIG_MD	0.289	0.367	0.524	0.298	0.278	0.412	**-0.79**	Increases
Contiguity Index - Range	CONTIG_RA	0.974	0.974	0.976	0.972	0.974	0.975	-0.42	Increases
Contiguity Index - Standard Deviation	CONTIG_SD	0.343	0.293	0.293	0.278	0.275	0.307	**-0.79**	Increases
Contiguity Index - Coefficient of Variation (%)	CONTIG_CV	89.21	77.53	60.69	83.43	81.89	78.51	0.56	Decreases
ISOLATION AND PROXIMITY									
Proximity Index - Mean	PROX_MN	139.23	82.46	185.04	70.41	104.32	119.26	-0.62	Increases
Proximity Index - Area-Weighted Mean	PROX_AM	175.30	813.05	940.37	178.37	635.80	927.10	-0.48	Increases
Proximity Index - Median	PROX_MD	0.53	0.61	3.24	1.05	0.94	1.23	-0.61	Increases
Proximity Index - Range	PROX_RA	1196.73	1998.20	2965.81	1246.60	1796.33	1784.79	-0.41	Increases
Proximity Index - Standard Deviation	PROX_SD	308.40	285.42	461.48	202.60	295.28	351.52	-0.63	Increases
Proximity Index - Coefficient of Variation (%)	PROX_CV	221.51	346.14	249.40	287.73	283.07	294.75	0.42	Decreases
Similarity Index - Mean	SIMI_MN	4258.73	6124.73	3162.08	4102.21	3721.61	4261.40	0.34	Decreases
Similarity Index - Area-Weighted Mean	SIMI_AM	4141.19	7106.66	5975.88	7936.29	7527.61	7094.24	0.58	Decreases
Similarity Index - Median	SIMI_MD	3147.46	7425.74	3187.15	4025.88	4276.20	4910.40	0.34	Decreases
Similarity Index - Range	SIMI_RA	8629.61	17491.31	16118.66	15443.60	15938.42	16107.51	0.17	Decreases
Similarity Index - Standard Deviation	SIMI_SD	2187.58	3061.05	2799.73	2902.17	2420.78	2798.20	-0.26	Increases
Similarity Index - Coefficient of Variation (%)	SIMI_CV	51.37	49.98	88.54	70.75	65.05	65.66	-0.54	Increases
Euclidian Nearest Neighbour Distance - Mean (m)	ENN_MN	73.89	48.99	47.01	45.38	45.11	46.28	-0.08	Increases
Euclidian Nearest Neighbour Distance - Area-Weighted Mean (m)	ENN_AM	52.34	29.35	28.07	29.73	26.43	26.68	0.16	Decreases
Euclidian Nearest Neighbour Distance - Median (m)	ENN_MD	30.00	29.14	22.36	30.00	28.28	22.36	**0.87**	Decreases
Euclidian Nearest Neighbour Distance - Range (m)	ENN_RA	347.64	296.23	241.73	297.65	213.45	860.00	-0.66	Increases
Euclidian Nearest Neighbour Distance - Standard Deviation (m)	ENN_SD	85.80	45.57	50.66	41.86	41.18	91.96	**-0.76**	Increases
Euclidian Nearest Neighbour Distance - Coefficient of Variation (%)	ENN_CV	116.12	93.02	107.76	92.25	91.28	198.71	**-0.75**	Increases
CONTRAST									
Contrast-Weighted Edge Density (m/ha)	CWED	14.39	24.14	20.09	27.67	27.95	19.88	**0.89**	Decreases
Total Edge Contrast Index (%)	TECI	10.96	11.01	10.25	12.09	11.48	10.24	**0.75**	Decreases
Edge Contrast Index - Mean (%)	ECON_MN	13.54	13.57	12.12	14.65	13.64	11.93	**0.78**	Decreases
Edge Contrast Index - Area-Weighted Mean (%)	ECON_AM	13.49	12.48	12.28	14.27	13.76	12.77	0.51	Decreases
Edge Contrast Index - Median (%)	ECON_MD	10.33	11.21	10.00	10.65	11.90	10.00	**0.97**	Decreases
Edge Contrast Index - Range (%)	ECON_RA	14.02	36.15	29.50	60.00	35.00	40.00	0.12	Decreases
Edge Contrast Index - Standard Deviation (%)	ECON_SD	4.56	6.35	6.06	7.51	5.69	5.54	0.23	Decreases
Edge Contrast Index - Coefficient of Variation (%)	ECON_CV	33.68	46.82	50.01	51.27	41.70	46.39	-0.49	Increases
CONTAGION AND INTERSPERSION									
Percentage of Like Adjacencies (%)	PLADJ	95.08	90.88	92.04	90.40	89.68	92.14	**-0.95**	Increases
Contagion (%)	CONTAG	66.92	63.37	63.77	60.87	59.10	64.42	**-0.80**	Increases
Aggregation Index (%)	AI	96.14	91.85	93.05	91.37	90.65	93.11	**-0.95**	Increases
DIVERSITY									
Shannon's Diversity Index	SHDI	0.577	0.866	0.897	0.931	0.965	0.878	0.55	Decreases
Simpson's Diversity Index	SIDI	0.333	0.429	0.435	0.496	0.501	0.450	0.52	Decreases
Modified Simpson's Diversity Index	MSIDI	0.406	0.561	0.571	0.686	0.695	0.597	0.53	Decreases
Shannon's Evenness Index	SHEI	0.525	0.538	0.557	0.579	0.600	0.546	0.55	Decreases
Simpson's Evenness Index	SIEI	0.500	0.537	0.544	0.620	0.626	0.562	0.52	Decreases
Modified Simpson's Evenness Index	MSIEI	0.369	0.348	0.355	0.426	0.432	0.371	0.53	Decreases

^{x} 1931 was exluded from the trend analysis since the coverage was incomplete
* Correlation Coefficient between metric value and water level for each year, significant correlations (i.e. values ≥ |0.7|) are bolded
^ Total vegetated wetland area (excludes open water)

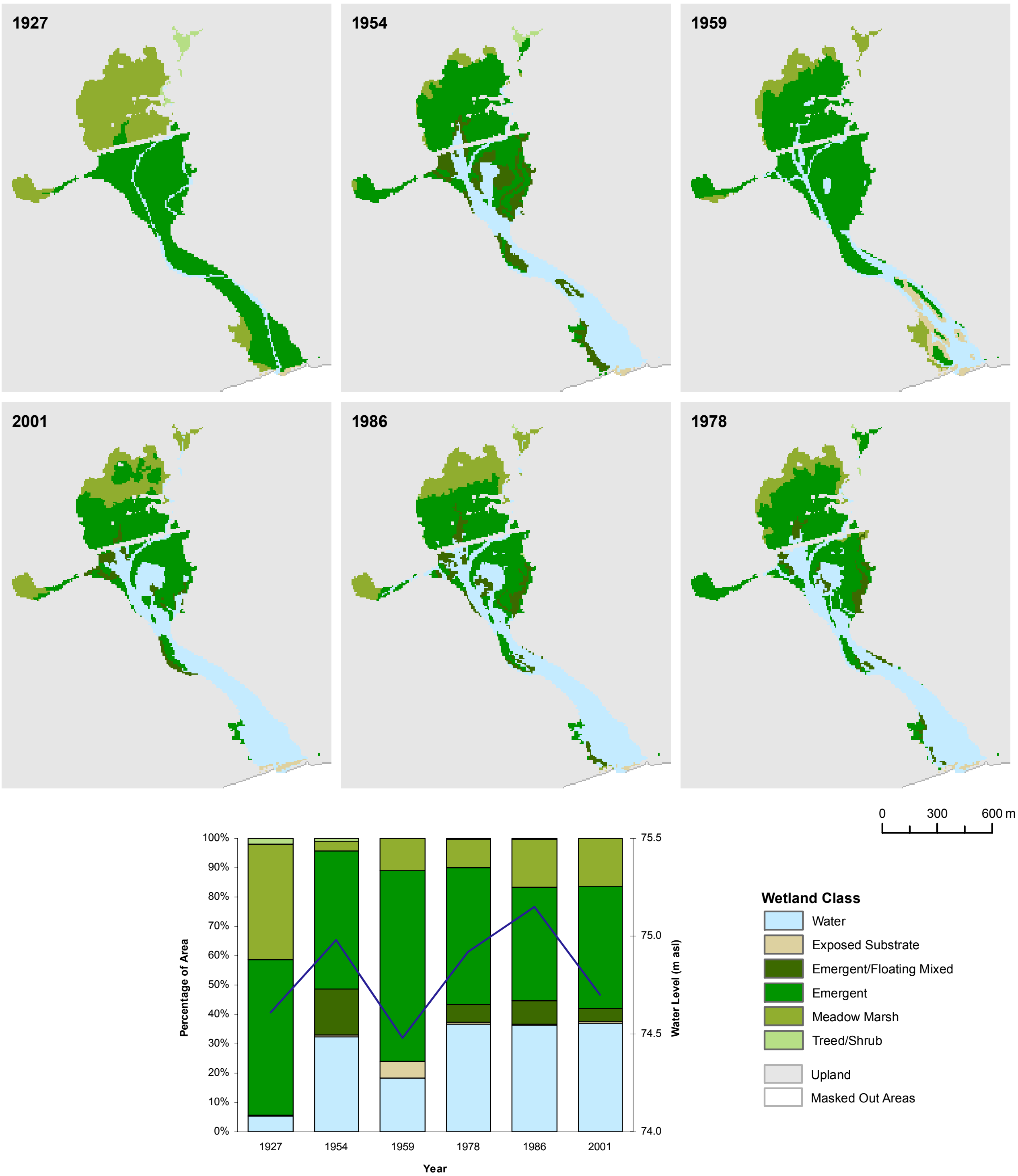

Figure 4.3 Historical distribution of wetland vegetation at Lynde Creek, 1927-2001

Table 4.3 Landscape metrics for Lynde Creek, 1927-2001

Landscape Metric	Code	Year and Water Level (m asl)						Correl. Coeff.*	Trend w/ ↓ Water
		1927	1954	1959	1978	1986	2001		
		74.61	***74.98***	***74.48***	***74.92***	***75.15***	***74.70***		
AREA, DENSITY, EDGE									
Total Area^ (ha)	TA	35.49	53.3	38.01	45.97	35.64	35.78	0.37	Decreases
Number of Patches (#)	NP	31	73	55	85	86	64	**0.80**	Decreases
Patch Density (#/100 ha)	PD	55.13	129.82	97.81	151.16	152.94	113.82	**0.80**	Decreases
Patch Area - Mean (m)	AREA_MN	1.81	0.77	1.02	0.66	0.65	0.88	-0.64	Increases
Patch Area - Area-Weighted Mean (m)	AREA_AM	21.06	10.92	9.54	11.41	10.80	10.58	-0.28	Increases
Patch Area - Median (m)	AREA_MD	0.04	0.09	0.09	0.05	0.04	0.09	-0.36	Increases
Patch Area - Range (m)	AREA_RA	28.70	17.40	15.16	19.63	19.39	19.72	-0.14	Increases
Patch Area - Standard Deviation (m)	AREA_SD	5.91	2.80	2.95	2.67	2.58	2.92	-0.47	Increases
Patch Area - Coefficient of Variation (%)	AREA_CV	325.77	362.93	288.64	403.02	393.92	332.23	**0.91**	Decreases
Radius of Gyration - Mean (m)	GYRATE_MN	49.13	29.86	34.56	25.08	24.72	30.76	-0.68	Increases
Radius of Gyration - Area-Weighted Mean (m)	GYRATE_AM	289.99	207.23	151.12	228.90	215.58	216.02	0.11	Decreases
Radius of Gyration - Median (m)	GYRATE_MD	11.09	15.00	12.79	10.05	10.59	14.27	-0.20	Increases
Radius of Gyration - Range (m)	GYRATE_RA	419.97	421.14	243.60	441.74	424.12	421.62	0.65	Decreases
Radius of Gyration - Standard Deviation (m)	GYRATE_SD	89.09	54.98	49.48	53.69	50.74	57.66	-0.36	Increases
Radius of Gyration - Coefficient of Variation (%)	GYRATE_CV	181.32	184.13	143.14	214.08	205.28	187.48	**0.78**	Decreases
Total Edge Length (m)	TE	5960	11240	10770	11290	11700	9620	0.56	Decreases
Edge Density (m/ha)	ED	105.99	199.89	191.53	200.78	208.07	171.08	0.56	Decreases
SHAPE									
Landscape Shape Index	LSI	6.18	7.94	7.78	7.96	8.09	7.40	0.56	Decreases
Perimeter-Area Ratio - Mean	PARA_MN	2502.25	2288.61	2306.16	2659.30	2633.14	2308.94	0.51	Decreases
Perimeter-Area Ratio - Area-Weighted Mean	PARA_AM	435.71	623.51	606.79	625.29	639.87	565.89	0.56	Decreases
Perimeter-Area Ratio - Median	PARA_MD	2500.00	1857.14	2000.00	2666.67	2666.67	2291.67	0.35	Decreases
Perimeter-Area Ratio - Range	PARA_RA	3790.60	3763.57	3749.51	3721.31	3685.64	3717.74	-0.58	Increases
Perimeter-Area Ratio - Standard Deviation	PARA_SD	1288.60	1242.06	1254.75	1152.28	1198.67	1216.79	-0.60	Increases
Perimeter-Area Ratio - Coefficient of Variation (%)	PARA_CV	51.50	54.27	54.41	43.33	45.52	52.70	-0.60	Increases
Shape Index - Mean	SHAPE_MN	1.83	1.55	1.69	1.47	1.50	1.54	**-0.72**	Increases
Shape Index - Area Weighted Mean	SHAPE_AM	3.33	3.00	2.96	3.44	3.40	3.00	0.48	Decreases
Shape Index - Median	SHAPE_MD	1.33	1.33	1.33	1.29	1.27	1.33	**-0.75**	Increases
Shape Index - Range	SHAPE_RA	4.63	3.63	4.53	4.10	4.16	3.04	-0.26	Increases
Shape Index - Standard Deviation	SHAPE_SD	1.15	0.72	0.96	0.64	0.69	0.63	-0.65	Increases
Shape Index - Coefficient of Variation (%)	SHAPE_CV	62.90	46.61	56.51	43.79	45.82	40.83	-0.60	Increases
Related Circumscribing Circle - Mean	CIRCLE_MN	0.631	0.602	0.628	0.590	0.584	0.613	**-0.93**	Increases
Related Circumscribing Circle - Area-Weighted Mean	CIRCLE_AM	0.712	0.701	0.713	0.737	0.721	0.728	0.11	Decreases
Related Circumscribing Circle - Median	CIRCLE_MD	0.682	0.576	0.624	0.608	0.565	0.631	**-0.80**	Increases
Related Circumscribing Circle - Range	CIRCLE_RA	0.615	0.556	0.554	0.540	0.552	0.562	-0.44	Increases
Related Circumscribing Circle - Standard Deviation	CIRCLE_SD	0.207	0.171	0.188	0.184	0.182	0.176	-0.50	Increases
Related Circumscribing Circle - Coefficient of Variation (%)	CIRCLE_CV	32.80	28.34	29.98	31.21	31.18	28.71	-0.07	Increases
Contiguity Index - Mean	CONTIG_MN	0.355	0.403	0.399	0.306	0.323	0.394	-0.50	Increases
Contiguity Index - Area-Weighted Mean	CONTIG_AM	0.880	0.827	0.834	0.827	0.825	0.843	-0.56	Increases
Contiguity Index - Median	CONTIG_MD	0.250	0.444	0.417	0.278	0.278	0.403	-0.23	Increases
Contiguity Index - Range	CONTIG_RA	0.940	0.933	0.929	0.921	0.911	0.919	-0.58	Increases
Contiguity Index - Standard Deviation	CONTIG_SD	0.311	0.294	0.303	0.275	0.283	0.294	**-0.75**	Increases
Contiguity Index - Coefficient of Variation (%)	CONTIG_CV	87.77	72.97	75.96	90.12	87.58	74.75	0.30	Decreases
ISOLATION AND PROXIMITY									
Proximity Index - Mean	PROX_MN	45.20	43.79	59.26	55.72	38.30	53.83	-0.68	Increases
Proximity Index - Area-Weighted Mean	PROX_AM	11.23	35.52	153.06	37.67	28.01	32.07	-0.55	Increases
Proximity Index - Median	PROX_MD	0.25	1.90	3.60	1.24	1.68	3.35	-0.33	Increases
Proximity Index - Range	PROX_RA	358.88	414.50	468.81	491.20	485.00	394.66	0.41	Decreases
Proximity Index - Standard Deviation	PROX_SD	99.55	90.51	121.13	118.38	91.73	108.72	-0.61	Increases
Proximity Index - Coefficient of Variation (%)	PROX_CV	220.25	206.68	204.39	212.45	239.50	201.98	0.59	Decreases
Similarity Index - Mean	SIMI_MN	1072.42	805.15	623.15	931.84	907.40	820.33	0.29	Decreases
Similarity Index - Area-Weighted Mean	SIMI_AM	626.44	1118.59	876.12	1083.37	1148.70	1086.38	**0.73**	Decreases
Similarity Index - Median	SIMI_MD	563.15	453.80	564.56	805.62	833.96	689.58	0.46	Decreases
Similarity Index - Range	SIMI_RA	2532.66	2025.82	1596.95	2371.90	2350.66	2424.60	0.34	Decreases
Similarity Index - Standard Deviation	SIMI_SD	950.70	683.22	449.64	663.92	740.06	700.55	0.19	Decreases
Similarity Index - Coefficient of Variation (%)	SIMI_CV	88.65	84.86	72.16	71.25	81.56	85.40	0.07	Decreases
Euclidian Nearest Neighbour Distance - Mean (m)	ENN_MN	56.66	40.76	37.63	53.47	54.36	40.37	0.38	Decreases
Euclidian Nearest Neighbour Distance - Area-Weighted Mean (m)	ENN_AM	38.19	28.33	28.39	25.09	36.75	27.33	0.10	Decreases
Euclidian Nearest Neighbour Distance - Median (m)	ENN_MD	28.28	22.36	22.36	22.36	25.32	22.36	-0.05	Increases
Euclidian Nearest Neighbour Distance - Range (m)	ENN_RA	293.87	412.67	170.00	1134.34	521.48	353.36	0.53	Decreases
Euclidian Nearest Neighbour Distance - Standard Deviation (m)	ENN_SD	68.64	52.33	29.20	127.72	88.68	49.40	0.58	Decreases
Euclidian Nearest Neighbour Distance - Coefficient of Variation (%)	ENN_CV	121.14	128.40	77.61	238.89	163.14	122.39	0.62	Decreases
CONTRAST									
Contrast-Weighted Edge Density (m/ha)	CWED	22.19	26.30	54.06	35.48	33.49	32.30	-0.39	Increases
Total Edge Contrast Index (%)	TECI	6.73	6.21	13.02	8.36	7.76	8.18	-0.58	Increases
Edge Contrast Index - Mean (%)	ECON_MN	9.52	7.48	15.95	11.23	10.61	10.18	-0.55	Increases
Edge Contrast Index - Area-Weighted Mean (%)	ECON_AM	8.43	8.11	14.49	11.11	10.18	11.24	-0.49	Increases
Edge Contrast Index - Median (%)	ECON_MD	7.50	6.39	10.00	10.00	10.00	10.00	-0.07	Increases
Edge Contrast Index - Range (%)	ECON_RA	26.67	30.00	41.05	31.11	40.00	40.00	-0.06	Increases
Edge Contrast Index - Standard Deviation (%)	ECON_SD	8.05	5.17	12.81	7.03	7.25	7.34	**-0.73**	Increases
Edge Contrast Index - Coefficient of Variation (%)	ECON_CV	84.58	69.22	80.32	62.56	68.37	72.08	**-0.78**	Increases
CONTAGION AND INTERSPERSION									
Percentage of Like Adjacencies (%)	PLADJ	89.11	84.41	84.83	84.37	84.00	85.85	-0.56	Increases
Contagion (%)	CONTAG	63.98	55.53	50.38	55.51	52.83	50.99	-0.07	Increases
Aggregation Index (%)	AI	91.17	86.69	86.85	86.57	86.26	88.12	-0.53	Increases
DIVERSITY									
Shannon's Diversity Index	SHDI	0.962	1.195	0.996	1.171	1.268	1.208	**0.84**	Decreases
Simpson's Diversity Index	SIDI	0.562	0.648	0.529	0.635	0.685	0.663	**0.85**	Decreases
Modified Simpson's Diversity Index	MSIDI	0.825	1.043	0.752	1.008	1.156	1.087	**0.85**	Decreases
Shannon's Evenness Index	SHEI	0.598	0.667	0.719	0.654	0.708	0.750	0.00	Decreases
Simpson's Evenness Index	SIEI	0.702	0.777	0.705	0.762	0.822	0.828	0.68	Decreases
Modified Simpson's Evenness Index	MSIEI	0.513	0.582	0.542	0.562	0.645	0.675	0.46	Decreases

* Correlation Coefficient between metric value and water level for each year, significant correlations (i.e. values ≥ |0.7|) are bolded
^ Total vegetated wetland area (excludes open water)

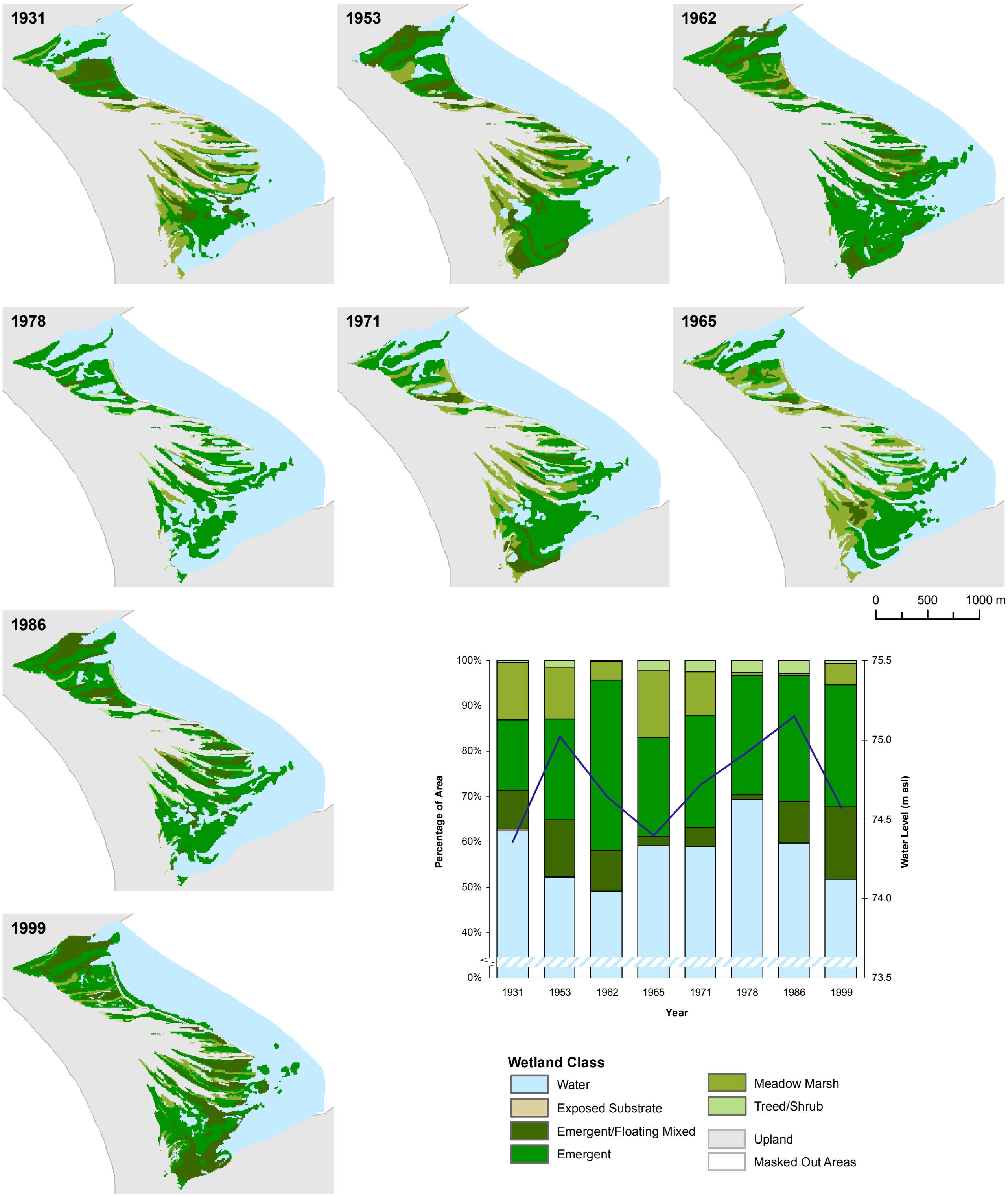

Figure 4.4 Historical distribution of wetland vegetation at Presqu'ile, 1931-1999

Table 4.4 Landscape metrics for Presqu'ile, 1931-1999

Landscape Metric	Code	Year and Water Level (m asl)								Correl.	Trend w/
		1931	1953	1962	1965	1971	1978	1986	1999	Coeff.*	↓ Water
		74.36	*75.02*	*74.64*	*74.40*	*74.72*	*74.92*	*75.15*	*74.57*		
AREA, DENSITY, EDGE											
Total Area^ (ha)	TA	103.64	131.78	140.55	112.71	113.36	84.49	111.30	133.34	-0.07	Increases
Number of Patches (#)	NP	129	153	142	148	153	135	146	214	-0.09	Increases
Patch Density (#/100 ha)	PD	46.68	55.36	51.38	53.55	55.36	48.85	52.83	77.44	-0.09	Increases
Patch Area - Mean (m)	AREA_MN	2.14	1.81	1.95	1.87	1.81	2.05	1.89	1.29	0.02	Decreases
Patch Area - Area-Weighted Mean (m)	AREA_AM	110.02	72.62	77.68	87.46	90.59	125.62	97.54	74.55	0.05	Decreases
Patch Area - Median (m)	AREA_MD	0.11	0.09	0.09	0.14	0.09	0.06	0.06	0.08	**-0.78**	Increases
Patch Area - Range (m)	AREA_RA	172.52	137.33	130.82	152.36	154.44	184.44	159.78	138.11	0.05	Decreases
Patch Area - Standard Deviation (m)	AREA_SD	15.20	11.31	12.14	12.64	12.66	15.91	13.46	9.73	0.04	Decreases
Patch Area - Coefficient of Variation (%)	AREA_CV	709.62	626.14	623.81	677.04	701.10	776.95	710.85	753.18	0.00	Increases
Radius of Gyration - Mean (m)	GYRATE_MN	44.53	41.60	41.47	42.80	40.07	36.83	37.35	34.59	-0.46	Increases
Radius of Gyration - Area-Weighted Mean (m)	GYRATE_AM	570.87	450.27	469.76	510.63	528.31	623.20	538.13	466.22	0.04	Decreases
Radius of Gyration - Median (m)	GYRATE_MD	18.65	15.31	16.88	19.02	16.29	11.48	14.45	15.05	**-0.74**	Increases
Radius of Gyration - Range (m)	GYRATE_RA	831.91	755.10	740.77	809.65	826.30	849.65	789.63	752.34	-0.13	Increases
Radius of Gyration - Standard Deviation (m)	GYRATE_SD	83.30	75.47	76.36	77.81	77.81	83.95	78.67	67.95	0.05	Decreases
Radius of Gyration - Coefficient of Variation (%)	GYRATE_CV	187.07	181.45	184.13	181.79	194.17	227.93	210.62	196.44	0.52	Decreases
Total Edge Length (m)	TE	35860	32770	38800	38920	35650	30560	35120	49670	-0.48	Increases
Edge Density (m/ha)	ED	129.76	118.58	140.40	140.83	129.00	110.58	127.08	179.73	-0.48	Increases
SHAPE											
Landscape Shape Index	LSI	10.36	9.89	10.80	10.82	10.33	9.56	10.25	12.43	-0.48	Increases
Perimeter-Area Ratio - Mean	PARA_MN	2137.00	2226.01	2040.28	2083.13	2221.83	2441.92	2361.50	2309.41	0.65	Decreases
Perimeter-Area Ratio - Area-Weighted Mean	PARA_AM	379.36	357.00	400.64	401.51	377.84	341.00	374.00	479.30	-0.48	Increases
Perimeter-Area Ratio - Median	PARA_MD	1818.18	2133.33	2000.00	1705.63	2000.00	2363.64	2366.67	2294.37	**0.77**	Decreases
Perimeter-Area Ratio - Range	PARA_RA	3865.88	3907.24	3889.17	3850.23	3864.81	3842.99	3873.96	3879.67	0.29	Decreases
Perimeter-Area Ratio - Standard Deviation	PARA_SD	1232.78	1271.63	1105.41	1195.02	1230.48	1255.79	1230.40	1080.26	0.43	Decreases
Perimeter-Area Ratio - Coefficient of Variation (%)	PARA_CV	57.69	57.13	54.18	57.37	55.38	51.43	52.10	46.78	-0.23	Increases
Shape Index - Mean	SHAPE_MN	1.74	1.62	1.71	1.72	1.66	1.58	1.62	1.70	**-0.90**	Increases
Shape Index - Area Weighted Mean	SHAPE_AM	4.03	2.77	4.21	3.93	3.71	4.66	4.13	3.99	-0.17	Increases
Shape Index - Median	SHAPE_MD	1.46	1.38	1.40	1.42	1.44	1.33	1.40	1.40	-0.63	Increases
Shape Index - Range	SHAPE_RA	4.84	3.63	7.19	4.27	3.92	4.67	7.11	6.62	0.10	Decreases
Shape Index - Standard Deviation	SHAPE_SD	0.96	0.75	0.92	0.86	0.78	0.81	0.91	0.96	-0.48	Increases
Shape Index - Coefficient of Variation (%)	SHAPE_CV	55.21	46.51	53.87	50.14	46.93	51.07	56.09	56.49	-0.18	Increases
Related Circumscribing Circle - Mean	CIRCLE_MN	0.656	0.656	0.674	0.676	0.665	0.639	0.641	0.666	-0.69	Increases
Related Circumscribing Circle - Area-Weighted Mean	CIRCLE_AM	0.776	0.762	0.761	0.785	0.791	0.779	0.774	0.782	-0.33	Increases
Related Circumscribing Circle - Median	CIRCLE_MD	0.682	0.700	0.703	0.720	0.718	0.682	0.688	0.700	-0.30	Increases
Related Circumscribing Circle - Range	CIRCLE_RA	0.630	0.588	0.589	0.641	0.646	0.570	0.590	0.614	-0.69	Increases
Related Circumscribing Circle - Standard Deviation	CIRCLE_SD	0.210	0.199	0.179	0.189	0.197	0.197	0.200	0.186	0.15	Decreases
Related Circumscribing Circle - Coefficient of Variation (%)	CIRCLE_CV	31.98	30.38	26.60	27.96	29.60	30.77	31.15	27.92	0.34	Decreases
Contiguity Index - Mean	CONTIG_MN	0.427	0.408	0.449	0.444	0.408	0.357	0.374	0.384	-0.65	Increases
Contiguity Index - Area-Weighted Mean	CONTIG_AM	0.892	0.899	0.887	0.887	0.893	0.903	0.894	0.866	0.50	Decreases
Contiguity Index - Median	CONTIG_MD	0.458	0.389	0.451	0.515	0.433	0.306	0.312	0.380	**-0.81**	Increases
Contiguity Index - Range	CONTIG_RA	0.962	0.974	0.969	0.958	0.962	0.956	0.965	0.966	0.29	Decreases
Contiguity Index - Standard Deviation	CONTIG_SD	0.300	0.310	0.272	0.288	0.294	0.295	0.296	0.260	0.40	Decreases
Contiguity Index - Coefficient of Variation (%)	CONTIG_CV	70.23	75.99	60.59	64.85	72.16	82.61	79.17	67.62	**0.73**	Decreases
ISOLATION AND PROXIMITY											
Proximity Index - Mean	PROX_MN	83.38	42.79	195.99	132.15	49.15	239.13	160.70	102.05	0.17	Decreases
Proximity Index - Area-Weighted Mean	PROX_AM	23.32	41.14	71.51	88.96	39.98	116.14	144.82	70.52	0.52	Decreases
Proximity Index - Median	PROX_MD	3.94	0.80	2.86	2.81	1.25	0.80	0.78	2.99	**-0.91**	Increases
Proximity Index - Range	PROX_RA	4313.25	1373.40	3273.55	3809.25	1930.63	4611.25	3994.86	3453.00	-0.25	Increases
Proximity Index - Standard Deviation	PROX_SD	415.77	152.62	499.17	479.99	181.77	827.37	586.58	366.49	0.13	Decreases
Proximity Index - Coefficient of Variation (%)	PROX_CV	498.64	356.66	254.69	363.21	369.82	345.99	365.01	359.13	-0.33	Increases
Similarity Index - Mean	SIMI_MN	5972.88	1969.93	4148.95	2868.44	3409.83	3783.64	3627.75	2821.77	-0.42	Increases
Similarity Index - Area-Weighted Mean	SIMI_AM	7155.29	6423.87	7826.74	7038.90	7679.07	7400.62	8763.87	9124.56	0.11	Decreases
Similarity Index - Median	SIMI_MD	2443.07	655.59	3851.52	1081.84	1139.03	1499.27	1445.44	1479.06	-0.33	Increases
Similarity Index - Range	SIMI_RA	17316.69	15668.70	17543.50	13848.74	16789.44	18133.91	18489.68	15257.53	0.50	Decreases
Similarity Index - Standard Deviation	SIMI_SD	5988.59	3206.75	3768.36	4153.97	4513.16	4974.06	4979.33	4061.99	-0.22	Increases
Similarity Index - Coefficient of Variation (%)	SIMI_CV	100.26	162.79	90.83	144.82	132.36	131.46	137.26	143.95	0.42	Decreases
Euclidian Nearest Neighbour Distance - Mean (m)	ENN_MN	43.06	50.72	46.95	42.50	45.59	48.06	45.01	41.09	0.67	Decreases
Euclidian Nearest Neighbour Distance - Area-Weighted Mean (m)	ENN_AM	26.65	36.14	23.55	24.48	34.74	24.94	25.21	24.54	0.31	Decreases
Euclidian Nearest Neighbour Distance - Median (m)	ENN_MD	22.36	30.00	31.62	28.28	30.00	28.28	30.00	28.28	0.56	Decreases
Euclidian Nearest Neighbour Distance - Range (m)	ENN_RA	616.00	592.70	370.00	230.20	234.56	347.70	266.36	812.17	-0.22	Increases
Euclidian Nearest Neighbour Distance - Standard Deviation (m)	ENN_SD	63.73	72.88	50.82	40.53	42.69	57.34	40.12	63.74	0.00	Increases
Euclidian Nearest Neighbour Distance - Coefficient of Variation (%)	ENN_CV	148.01	143.68	108.24	95.35	93.64	119.30	89.14	155.13	-0.24	Increases
CONTRAST											
Contrast-Weighted Edge Density (m/ha)	CWED	25.31	23.75	22.99	33.24	29.88	27.96	25.93	29.75	-0.39	Increases
Total Edge Contrast Index (%)	TECI	10.14	9.96	8.83	12.75	12.01	12.13	10.50	9.93	-0.08	Increases
Edge Contrast Index - Mean (%)	ECON_MN	14.28	14.73	13.47	15.45	16.05	17.94	16.36	13.55	0.50	Decreases
Edge Contrast Index - Area-Weighted Mean (%)	ECON_AM	13.24	12.08	10.27	15.72	14.95	14.95	12.09	11.31	-0.19	Increases
Edge Contrast Index - Median (%)	ECON_MD	13.33	12.60	10.79	15.00	15.86	19.32	17.88	13.33	0.45	Decreases
Edge Contrast Index - Range (%)	ECON_RA	47.86	41.50	40.00	45.00	37.50	44.35	42.50	36.67	-0.21	Increases
Edge Contrast Index - Standard Deviation (%)	ECON_SD	7.98	9.06	8.28	9.21	8.77	10.30	9.48	6.76	0.55	Decreases
Edge Contrast Index - Coefficient of Variation (%)	ECON_CV	55.87	61.49	61.44	59.62	54.64	57.42	57.96	49.90	0.27	Decreases
CONTAGION AND INTERSPERSION											
Percentage of Like Adjacencies (%)	PLADJ	90.52	91.08	89.98	89.96	90.55	91.47	90.65	88.02	0.48	Decreases
Contagion (%)	CONTAG	61.26	57.43	61.45	60.75	60.77	71.29	64.16	57.34	0.32	Decreases
Aggregation Index (%)	AI	91.54	92.15	90.98	90.97	91.58	92.37	91.62	89.02	0.48	Decreases
DIVERSITY											
Shannon's Diversity Index	SHDI	1.099	1.246	1.085	1.091	1.107	0.786	1.009	1.164	-0.23	Increases
Simpson's Diversity Index	SIDI	0.562	0.648	0.608	0.580	0.579	0.449	0.557	0.632	-0.16	Increases
Modified Simpson's Diversity Index	MSIDI	0.826	1.044	0.936	0.867	0.866	0.596	0.814	0.999	-0.12	Increases
Shannon's Evenness Index	SHEI	0.613	0.695	0.605	0.609	0.618	0.438	0.563	0.650	-0.23	Increases
Simpson's Evenness Index	SIEI	0.674	0.778	0.729	0.696	0.695	0.539	0.668	0.758	-0.16	Increases
Modified Simpson's Evenness Index	MSIEI	0.461	0.583	0.522	0.484	0.483	0.333	0.454	0.557	-0.12	Increases

* Correlation Coefficient between metric value and water level for each year, significant correlations (i.e. values ≥ |0.7|) are bolded
^ Total vegetated wetland area (excludes open water)

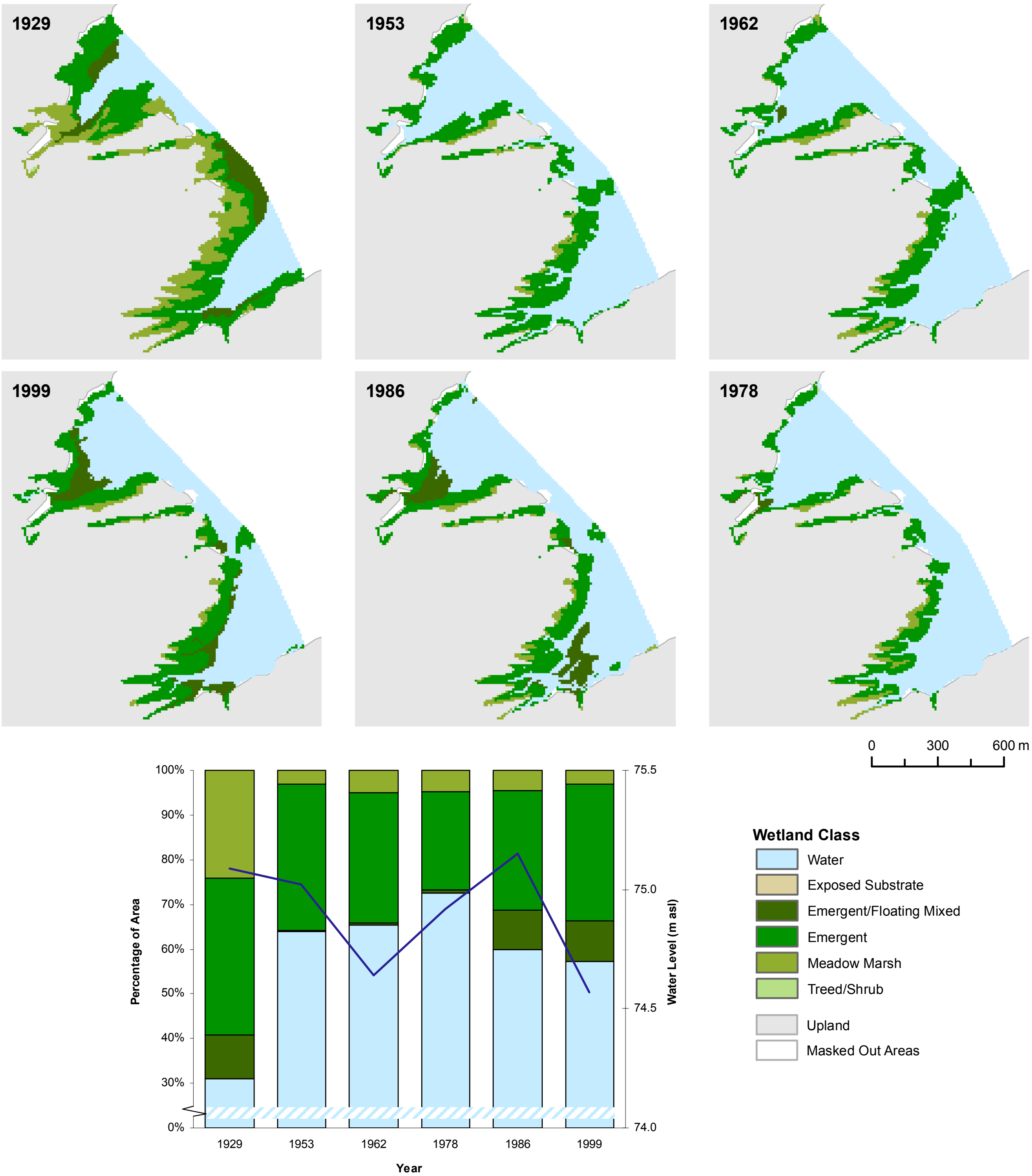

Figure 4.5 Historical distribution of wetland vegetation at South Bay, 1929-1999

Table 4.5 Landscape metrics for South Bay, 1929-1953

Landscape Metric	Code	Year and Water Level (m asl)						Correl.	Trend w/
		1929	1953	1962	1978	1986	1999	Coeff.*	↓ Water
		75.09	*75.02*	*74.64*	*74.92*	*75.15*	*74.57*		
AREA, DENSITY, EDGE									
Total Area^ (ha)	TA	38.96	20.28	19.56	15.52	22.67	24.1	0.31	Decreases
Number of Patches (#)	NP	35	50	50	52	68	52	0.08	Decreases
Patch Density (#/100 ha)	PD	62.10	88.72	88.72	92.26	120.65	92.26	0.08	Decreases
Patch Area - Mean (m)	AREA_MN	1.61	1.13	1.13	1.08	0.83	1.08	0.10	Decreases
Patch Area - Area-Weighted Mean (m)	AREA_AM	5.98	12.44	13.36	29.92	21.24	19.81	-0.13	Increases
Patch Area - Median (m)	AREA_MD	0.27	0.07	0.05	0.04	0.05	0.09	0.31	Decreases
Patch Area - Range (m)	AREA_RA	10.44	19.68	19.61	40.62	33.58	32.11	-0.14	Increases
Patch Area - Standard Deviation (m)	AREA_SD	2.65	3.57	3.71	5.59	4.11	4.50	-0.29	Increases
Patch Area - Coefficient of Variation (%)	AREA_CV	164.76	316.76	329.47	515.79	496.28	415.63	-0.10	Increases
Radius of Gyration - Mean (m)	GYRATE_MN	50.98	33.49	36.06	33.14	29.27	37.70	0.03	Decreases
Radius of Gyration - Area-Weighted Mean (m)	GYRATE_AM	135.31	169.82	191.45	377.36	325.85	317.54	-0.14	Increases
Radius of Gyration - Median (m)	GYRATE_MD	23.46	14.36	13.02	9.96	11.00	14.51	0.19	Decreases
Radius of Gyration - Range (m)	GYRATE_RA	241.66	245.37	240.81	481.92	468.29	464.67	-0.07	Increases
Radius of Gyration - Standard Deviation (m)	GYRATE_SD	54.67	47.28	57.07	70.83	63.27	72.06	-0.40	Increases
Radius of Gyration - Coefficient of Variation (%)	GYRATE_CV	107.24	141.17	158.28	213.70	216.16	191.16	-0.13	Increases
Total Edge Length (m)	TE	11470	9010	9070	7820	10340	8600	0.57	Decreases
Edge Density (m/ha)	ED	203.51	159.87	160.93	138.75	183.46	152.59	0.57	Decreases
SHAPE									
Landscape Shape Index	LSI	7.63	6.82	6.84	6.42	7.26	6.68	0.57	Decreases
Perimeter-Area Ratio - Mean	PARA_MN	1727.34	2315.85	2554.46	2565.28	2580.04	2243.86	-0.20	Increases
Perimeter-Area Ratio - Area-Weighted Mean	PARA_AM	612.49	525.20	527.32	482.97	572.39	510.65	0.57	Decreases
Perimeter-Area Ratio - Median	PARA_MD	1272.73	2142.86	2733.33	2583.33	2619.05	2000.00	-0.22	Increases
Perimeter-Area Ratio - Range	PARA_RA	3748.87	3722.70	3753.31	3793.75	3707.65	3782.69	-0.62	Increases
Perimeter-Area Ratio - Standard Deviation	PARA_SD	1220.99	1164.90	1226.97	1166.94	1188.86	1138.88	0.23	Decreases
Perimeter-Area Ratio - Coefficient of Variation (%)	PARA_CV	70.69	50.30	48.03	45.49	46.08	50.76	0.28	Decreases
Shape Index - Mean	SHAPE_MN	1.92	1.65	1.66	1.62	1.55	1.62	0.21	Decreases
Shape Index - Area Weighted Mean	SHAPE_AM	3.07	2.93	3.02	3.25	3.70	2.98	0.54	Decreases
Shape Index - Median	SHAPE_MD	1.45	1.42	1.33	1.33	1.31	1.40	0.09	Decreases
Shape Index - Range	SHAPE_RA	4.89	3.41	4.14	3.89	3.31	2.86	0.30	Decreases
Shape Index - Standard Deviation	SHAPE_SD	1.08	0.71	0.86	0.81	0.70	0.71	0.18	Decreases
Shape Index - Coefficient of Variation (%)	SHAPE_CV	56.44	42.87	51.70	49.64	45.19	43.79	0.12	Decreases
Related Circumscribing Circle - Mean	CIRCLE_MN	0.654	0.665	0.637	0.627	0.621	0.629	0.25	Decreases
Related Circumscribing Circle - Area-Weighted Mean	CIRCLE_AM	0.775	0.742	0.751	0.786	0.802	0.810	-0.06	Increases
Related Circumscribing Circle - Median	CIRCLE_MD	0.693	0.687	0.682	0.650	0.682	0.641	0.60	Decreases
Related Circumscribing Circle - Range	CIRCLE_RA	0.556	0.555	0.586	0.572	0.556	0.623	**-0.92**	Increases
Related Circumscribing Circle - Standard Deviation	CIRCLE_SD	0.176	0.166	0.196	0.197	0.185	0.188	-0.51	Increases
Related Circumscribing Circle - Coefficient of Variation (%)	CIRCLE_CV	26.92	24.97	30.81	31.43	29.87	29.83	-0.45	Increases
Contiguity Index - Mean	CONTIG_MN	0.535	0.391	0.338	0.327	0.330	0.405	0.21	Decreases
Contiguity Index - Area-Weighted Mean	CONTIG_AM	0.830	0.853	0.853	0.865	0.842	0.858	-0.57	Increases
Contiguity Index - Median	CONTIG_MD	0.620	0.406	0.285	0.289	0.272	0.439	0.16	Decreases
Contiguity Index - Range	CONTIG_RA	0.931	0.924	0.932	0.943	0.920	0.940	-0.63	Increases
Contiguity Index - Standard Deviation	CONTIG_SD	0.306	0.276	0.290	0.278	0.279	0.277	0.18	Decreases
Contiguity Index - Coefficient of Variation (%)	CONTIG_CV	57.15	70.50	85.67	84.84	84.29	68.37	-0.14	Increases
ISOLATION AND PROXIMITY									
Proximity Index - Mean	PROX_MN	35.23	40.96	44.69	69.13	57.69	32.62	0.32	Decreases
Proximity Index - Area-Weighted Mean	PROX_AM	9.84	245.07	70.40	8.83	6.49	7.97	0.11	Decreases
Proximity Index - Median	PROX_MD	0.72	3.07	1.43	1.54	1.40	3.21	-0.44	Increases
Proximity Index - Range	PROX_RA	222.00	393.90	392.40	1016.55	671.80	321.20	0.21	Decreases
Proximity Index - Standard Deviation	PROX_SD	57.09	92.59	99.11	198.98	163.16	70.73	0.28	Decreases
Proximity Index - Coefficient of Variation (%)	PROX_CV	162.05	226.05	221.77	287.84	282.82	216.84	0.13	Decreases
Similarity Index - Mean	SIMI_MN	785.27	844.01	850.94	1313.47	1430.52	1373.17	-0.08	Increases
Similarity Index - Area-Weighted Mean	SIMI_AM	1471.60	1215.56	1448.34	1502.27	2000.18	2152.86	-0.26	Increases
Similarity Index - Median	SIMI_MD	860.00	599.77	675.19	782.67	979.24	622.78	0.68	Decreases
Similarity Index - Range	SIMI_RA	2412.35	2920.40	3038.06	3901.06	3692.62	3610.54	-0.20	Increases
Similarity Index - Standard Deviation	SIMI_SD	635.38	648.83	629.04	1313.14	1262.05	1333.57	-0.12	Increases
Similarity Index - Coefficient of Variation (%)	SIMI_CV	80.91	76.87	73.92	99.97	88.22	97.12	-0.14	Increases
Euclidian Nearest Neighbour Distance - Mean (m)	ENN_MN	65.43	38.79	43.65	38.72	56.93	37.08	0.64	Decreases
Euclidian Nearest Neighbour Distance - Area-Weighted Mean (m)	ENN_AM	52.63	25.21	25.12	23.26	26.16	29.81	0.30	Decreases
Euclidian Nearest Neighbour Distance - Median (m)	ENN_MD	31.62	31.62	30.81	30.00	30.00	25.32	0.65	Decreases
Euclidian Nearest Neighbour Distance - Range (m)	ENN_RA	290.64	250.74	167.88	213.45	362.88	213.24	**0.83**	Decreases
Euclidian Nearest Neighbour Distance - Standard Deviation (m)	ENN_SD	76.36	36.69	32.23	31.85	75.12	33.54	**0.74**	Decreases
Euclidian Nearest Neighbour Distance - Coefficient of Variation (%)	ENN_CV	116.69	94.59	73.85	82.27	131.93	90.44	**0.76**	Decreases
CONTRAST									
Contrast-Weighted Edge Density (m/ha)	CWED	35.81	32.82	31.56	27.71	30.62	22.39	0.66	Decreases
Total Edge Contrast Index (%)	TECI	8.75	8.98	8.62	8.05	7.87	6.25	0.54	Decreases
Edge Contrast Index - Mean (%)	ECON_MN	10.73	11.69	10.92	11.82	10.73	9.05	0.54	Decreases
Edge Contrast Index - Area-Weighted Mean (%)	ECON_AM	11.94	13.47	12.65	11.76	11.15	9.05	0.41	Decreases
Edge Contrast Index - Median (%)	ECON_MD	10.00	10.00	10.00	10.00	10.00	9.59	0.67	Decreases
Edge Contrast Index - Range (%)	ECON_RA	22.50	30.00	20.00	40.00	25.00	20.00	0.37	Decreases
Edge Contrast Index - Standard Deviation (%)	ECON_SD	5.28	6.23	4.06	7.26	4.71	3.99	0.49	Decreases
Edge Contrast Index - Coefficient of Variation (%)	ECON_CV	49.24	53.31	37.19	61.43	43.87	44.07	0.41	Decreases
CONTAGION AND INTERSPERSION									
Percentage of Like Adjacencies (%)	PLADJ	84.69	86.87	86.82	87.93	85.69	87.23	-0.57	Increases
Contagion (%)	CONTAG	46.18	66.65	59.54	71.13	49.74	52.13	-0.12	Increases
Aggregation Index (%)	AI	86.98	88.68	88.69	89.77	87.71	89.29	-0.61	Increases
DIVERSITY									
Shannon's Diversity Index	SHDI	1.309	0.769	0.814	0.754	1.015	1.005	0.30	Decreases
Simpson's Diversity Index	SIDI	0.715	0.482	0.486	0.424	0.561	0.569	0.27	Decreases
Modified Simpson's Diversity Index	MSIDI	1.254	0.657	0.665	0.552	0.822	0.842	0.30	Decreases
Shannon's Evenness Index	SHEI	0.813	0.478	0.587	0.421	0.732	0.725	0.09	Decreases
Simpson's Evenness Index	SIEI	0.893	0.602	0.648	0.509	0.748	0.759	0.17	Decreases
Modified Simpson's Evenness Index	MSIEI	0.779	0.408	0.480	0.308	0.593	0.607	0.16	Decreases

* Correlation Coefficient between metric value and water level for each year, significant correlations (i.e. values ≥ |0.7|) are bolded
^ Total vegetated wetland area (excludes open water)

Table 4.6 Wetland vegetation response on Lake Ontario

Declining Water Levels	Rising Water Levels	Indicated By
LANDSCAPE ANALYSIS		
• Vegetated wetland area increased (Hay Bay, Presqu'ile)	• Vegetated wetland area decreased (Hay Bay, Presqu'ile)	• TA
• Wetland less fragmented, fewer patches • Patches larger in size • Patches more elongated and less compact	• Wetland more fragmented, more patches • Patches smaller in size • Patches more compact	• NP, PD • AREA_ • GYRATE_, SHAPE_
• Wetland more homogeneous	• Wetland more heterogeneous	• TE, ED
• Patches more aggregated, regular in shape • Patches more elongated and linear • Patches larger, more contiguous	• Patches more disaggregated, irregular in shape • Patches more circular • Patches smaller, fragmented, less contiguous	• LSI • CIRCLE_ • CONTIG_
• Neighbourhood occupied by more patches of the same class type (Lynde Creek) • Patches located closer to neighbours (Presqu'ile, South Bay)	• Neighbourhood occupied by fewer patches of the same class type (Lynde Creek) • Patches located further from neighbours (Presqu'ile, South Bay)	• PROX_ • ENN_
• Less edge contrast (South Bay, Hay Bay)	• More edge contrast (South Bay, Hay Bay)	• CWED
• Patches more aggregated, the proportion of like adjacencies increased (i.e. patches of similar communities more equally adjacent to each other)	• Patches more disaggregated and interspersed (except Presqu'ile)	• PLADJ, CONTAG, AI
• Wetlands less diverse, proportion and distribution of area among different patch types more uneven (riverine wetlands)	• Wetlands more diverse, distribution of patch type area more even across landscape (riverine wetlands)	• SHDI, SIDI, MSIDI, SHEI, SIEI, MSIEI
CLASS ANALYSIS		
• Open water, emergent/floating mixed decreased in area • Meadow marsh, treed/shrub increased in area	• Water, emergent/floating mixed increased in area • Meadow marsh, treed/shrub decreased in area	• CA
• Patches of open water, emergent/floating mixed, emergent, meadow marsh, treed/shrub less fragmented • Meadow marsh more fragmented at Presqu'ile	• Patches of open water, emergent/floating mixed, emergent, meadow marsh, treed/shrub more fragmented • Meadow marsh less fragmented at Presqu'ile	• NP, PD, TE, ED
• Patches of emergent, meadow marsh, treed/shrub larger and contiguous in size, and more elongated, less compact, and simpler in shape • Patches of meadow marsh more irregular, complex, convoluted, linear in shape (Presqu'ile)	• Patches of emergent, meadow marsh, treed/shrub smaller in size, and more compact and complex in shape • Patches of meadow marsh more regular and simple in shape (Presqu'ile)	• AREA_, GYRATE_, LSI, PARA_, SHAPE_, CONTIG_
• Patches of emergent/floating mixed, emergent, meadow marsh had less contrast in class edge • At Presqu'ile, patches of meadow marsh well contrasted with neighbouring patches	• Patches of emergent/floating mixed, emergent, meadow had more edge contrast with neighbours • At Presqu'ile, patches of meadow marsh less contrasted with neighbouring patches	• CWED, TECI, ECON_
• Patches of emergent/floating mixed and emergents located closer to neighbours of the same class	• Patches of emergent/floating mixed and emergents located further from neighbours of the same class	• PROX_
• Patches of emergent and meadow marsh more aggregated in distribution across the wetland	• Patches of emergent and meadow marsh more dispersed in distribution across the wetland	• PLADJ, AI
SPATIAL ANALYSIS		
• Lakeward migration of wetland communities • Succession to drier communities	• Landward migration of wetland communities • Succession to wetter communities	• Visual Observation

4.2.2 Lake Erie (unregulated water levels, marshes)

Lake Erie wetland sites were also characteristic of marsh wetlands. Unlike the Lake Ontario sites, however, lake levels were unregulated allowing wetland vegetation to respond naturally to water level fluctuations. Trends in wetland vegetation response at Dunnville (Figure 4.6; Table 4.7), the only riverine wetland study site on Lake Erie, were most strongly linked with lake level changes. The lacustrine wetlands, Long Point (Figure 4.7; Table 4.8), Rondeau (Figure 4.8; Table 4.9), and Turkey Point (Figure 4.9; Table 4.10) had less correlation but generally responded similarly to the wetland at Dunnville. At the class level, there were trends in several wetland communities, but unlike at the landscape level, these trends were less evident at Dunnville (Appendix 4.5). The findings of the spatiotemporal analysis for the Lake Erie wetlands are summarized in Table 4.11.

As water levels declined, there were a number of key changes in the composition and configuration of the Lake Erie wetlands. As water levels declined, the vegetation tended towards drier communities. The total vegetated area in these wetlands increased as drier wetland vegetation communities expanded lakeward replacing open water and emergent/floating mixed vegetation. There were notable increases in emergent and/or meadow marsh vegetation. Furthermore, substrate along shorelines was exposed during low water periods.

The wetlands were more diverse as the area among patch types became more even; this may be due to the large proportion of open water included in the wetland study sites compared to the wetlands on Lake Ontario. Although, open water dominated the Lake Erie wetlands during high water level periods, the distribution of other communities increased in the wetlands as water levels declined. Patches of emergent and meadow marsh vegetation expanded lakeward forming larger and more continuous patches of vegetation. As a result, there was less fragmentation in the wetlands. There was more interspersion in the wetlands as water levels declined. Patches of the same wetland community were located further from each other (i.e. the neighbourhood was occupied by more patches of different wetland community types). These patches were more fragmented and less contiguous in distribution but located close to or adjacent to patches of similar wetland communities.

During the spatial overlay analysis, a definite lag effect was noted in vegetation change during some years and wetlands. For example, at Long Point water levels were record low in 1964, after which water levels increased to 1968 levels. During this time, the areas of emergent/floating mixed and emergent vegetation were flooded by the rising water levels, but a significant portion of emergent vegetation converted to meadow marsh in high elevation areas that remained dry. Another example is in 1978, when water levels declined slightly from a period of near-record high water levels in 1973-74. Between 1972 and 1978, vegetation at Long Point changed to more water-tolerant communities. Areas of meadow march succeeded to emergent vegetation, emergent to emergent/floating mixed, and emergent/floating mixed to open water. Therefore, it is important to not only determine the amount of change in water levels for specific dates but also to examine variations in water level conditions between historical dates.

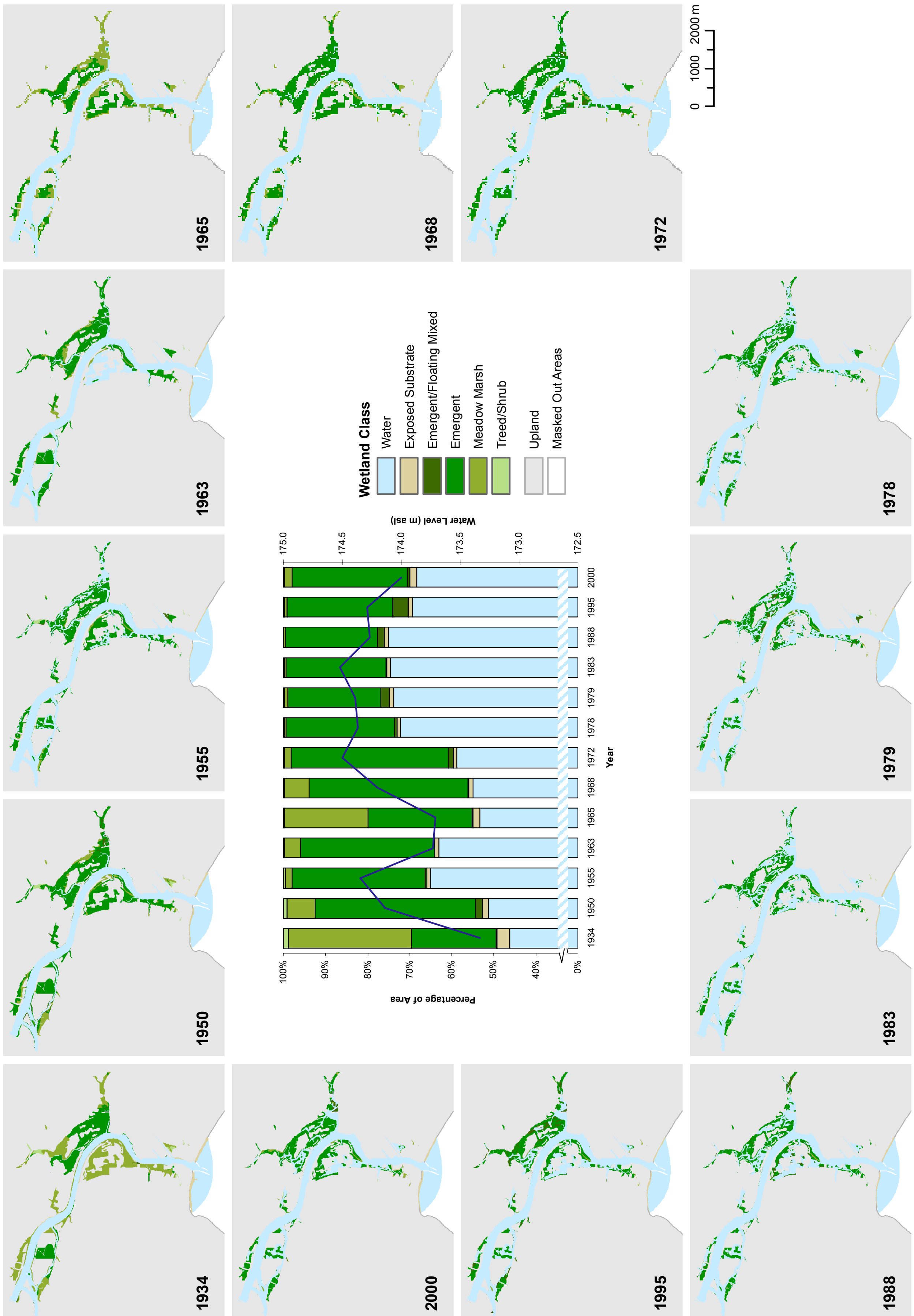

Figure 4.6 Historical distribution of wetland vegetation at Dunnville, 1934-2000

Table 4.7 Landscape metrics for Dunnville, 1934-2000

Landscape Metric	Code	Year and Water Level (m asl) 1934	1950	1955	1963	1965	1968	1972	1978	1979	1983	1988	1995	2000	Correl. Coeff.*	Trend w/ ↓ Water
		173.33	*174.14*	*174.35*	*173.73*	*173.71*	*174.20*	*174.50*	*174.37*	*174.39*	*174.52*	*174.27*	*174.29*	*174.00*		
AREA, DENSITY, EDGE																
Total Area^ (ha)	TA	146.36	256.61	232.04	166.71	176.73	222.93	214.57	196.63	133.08	124.72	121.07	119.35	151.3	0.09	Decreases
Number of Patches (#)	NP	155	313	344	223	207	231	276	373	430	347	339	314	264	**0.81**	Decreases
Patch Density (#/100 ha)	PD	32.47	65.56	72.06	46.71	43.36	48.39	57.81	78.13	90.07	72.69	71.01	65.77	55.30	**0.81**	Decreases
Patch Area - Mean (m)	AREA_MN	3.08	1.53	1.39	2.14	2.31	2.07	1.73	1.28	1.11	1.38	1.41	1.52	1.81	**-0.89**	Increases
Patch Area - Area-Weighted Mean (m)	AREA_AM	114.84	134.30	173.71	191.75	138.06	152.85	161.16	207.73	209.31	215.38	217.82	189.40	192.58	0.61	Decreases
Patch Area - Median (m)	AREA_MD	0.09	0.04	0.05	0.03	0.08	0.06	0.04	0.05	0.05	0.04	0.06	0.06	0.07	-0.60	Increases
Patch Area - Range (m)	AREA_RA	219.27	242.77	283.95	295.93	250.85	258.91	269.72	312.81	315.16	319.32	321.49	298.92	300.53	0.63	Decreases
Patch Area - Standard Deviation (m)	AREA_SD	18.55	14.23	15.46	20.15	17.69	17.65	16.61	16.26	15.20	17.16	17.46	16.90	18.57	-0.60	Increases
Patch Area - Coefficient of Variation (%)	AREA_CV	602.36	933.02	1114.32	941.10	767.21	854.16	960.06	1270.04	1369.40	1247.18	1239.64	1111.62	1027.11	**0.79**	Decreases
Radius of Gyration - Mean (m)	GYRATE_MN	59.42	34.31	32.15	36.90	49.51	40.10	34.34	28.67	27.68	29.74	31.62	34.95	36.71	**-0.88**	Increases
Radius of Gyration - Area-Weighted Mean (m)	GYRATE_AM	1138.37	1203.77	1330.58	1296.87	1231.05	1286.66	1311.08	1423.76	1411.97	1426.98	1420.06	1349.28	1369.54	**0.75**	Decreases
Radius of Gyration - Median (m)	GYRATE_MD	16.12	9.98	10.70	7.07	15.45	13.73	9.89	10.42	11.44	9.78	11.44	12.31	12.50	-0.49	Increases
Radius of Gyration - Range (m)	GYRATE_RA	2178.04	2115.82	2111.81	1940.77	2143.74	2145.67	2130.70	2078.68	2081.27	2067.85	2050.40	2069.85	2073.47	-0.11	Increases
Radius of Gyration - Standard Deviation (m)	GYRATE_SD	184.23	128.72	119.68	137.87	158.54	148.72	135.78	113.55	104.08	115.26	115.75	122.12	133.47	**-0.84**	Increases
Radius of Gyration - Coefficient of Variation (%)	GYRATE_CV	310.06	375.16	372.21	373.68	320.20	370.92	395.36	396.02	376.01	387.60	366.03	349.46	363.58	**0.81**	Decreases
Total Edge Length (m)	TE	33330	46100	55080	33860	38380	36430	43050	60710	68680	56620	53050	52200	46870	**0.73**	Decreases
Edge Density (m/ha)	ED	69.82	96.56	115.37	70.93	80.39	76.31	90.18	127.17	143.86	118.60	111.12	109.34	98.18	**0.73**	Decreases
SHAPE																
Landscape Shape Index	LSI	14.28	15.74	16.77	14.34	14.86	14.64	15.39	17.42	18.33	16.95	16.54	16.44	15.83	**0.73**	Decreases
Perimeter-Area Ratio - Mean	PARA_MN	2254.93	2640.79	2503.89	2709.75	2283.86	2387.22	2609.60	2479.74	2412.40	2527.27	2446.98	2440.64	2390.60	0.39	Decreases
Perimeter-Area Ratio - Area-Weighted Mean	PARA_AM	331.29	384.79	422.41	333.51	352.45	344.28	372.02	446.00	479.39	428.86	413.91	410.35	388.02	**0.73**	Decreases
Perimeter-Area Ratio - Median	PARA_MD	2333.33	2666.67	2500.00	3000.00	2370.37	2400.00	2666.67	2500.00	2400.00	2666.67	2500.00	2400.00	2422.22	0.09	Decreases
Perimeter-Area Ratio - Range	PARA_RA	3813.94	3809.95	3782.58	3798.81	3815.83	3812.14	3791.94	3744.77	3742.62	3751.98	3765.04	3790.39	3776.34	-0.64	Increases
Perimeter-Area Ratio - Standard Deviation	PARA_SD	1231.97	1190.76	1242.46	1274.32	1294.47	1263.56	1232.94	1169.77	1164.93	1245.48	1239.00	1284.59	1290.46	-0.35	Increases
Perimeter-Area Ratio - Coefficient of Variation (%)	PARA_CV	54.63	45.09	49.62	47.03	56.68	52.93	47.25	47.17	48.29	49.28	50.63	52.63	53.98	-0.51	Increases
Shape Index - Mean	SHAPE_MN	1.78	1.53	1.51	1.50	1.64	1.57	1.50	1.48	1.49	1.46	1.49	1.55	1.53	**-0.83**	Increases
Shape Index - Area Weighted Mean	SHAPE_AM	5.08	5.59	6.86	6.70	5.27	5.74	6.46	8.52	8.46	8.37	7.90	6.74	7.25	0.66	Decreases
Shape Index - Median	SHAPE_MD	1.50	1.25	1.27	1.25	1.40	1.33	1.25	1.25	1.25	1.22	1.25	1.33	1.25	**-0.75**	Increases
Shape Index - Range	SHAPE_RA	5.87	6.39	8.13	7.63	6.29	6.55	7.53	10.28	10.39	10.06	9.52	8.05	8.69	0.65	Decreases
Shape Index - Standard Deviation	SHAPE_SD	0.97	0.82	0.81	0.85	0.80	0.80	0.83	0.81	0.78	0.82	0.78	0.80	0.86	**-0.73**	Increases
Shape Index - Coefficient of Variation (%)	SHAPE_CV	54.40	53.55	53.42	56.53	48.83	51.27	55.18	54.77	52.32	56.03	52.06	51.50	56.00	0.06	Decreases
Related Circumscribing Circle - Mean	CIRCLE_MN	0.676	0.610	0.601	0.598	0.651	0.621	0.603	0.602	0.602	0.582	0.603	0.616	0.605	**-0.79**	Increases
Related Circumscribing Circle - Area-Weighted Mean	CIRCLE_AM	0.836	0.865	0.859	0.863	0.867	0.852	0.854	0.862	0.859	0.863	0.861	0.862	0.864	0.40	Decreases
Related Circumscribing Circle - Median	CIRCLE_MD	0.714	0.618	0.618	0.618	0.692	0.641	0.618	0.618	0.618	0.593	0.618	0.638	0.620	**-0.80**	Increases
Related Circumscribing Circle - Range	CIRCLE_RA	0.651	0.630	0.638	0.653	0.587	0.674	0.617	0.608	0.654	0.647	0.661	0.654	0.661	0.03	Decreases
Related Circumscribing Circle - Standard Deviation	CIRCLE_SD	0.201	0.199	0.190	0.199	0.195	0.191	0.193	0.176	0.175	0.184	0.188	0.198	0.194	-0.63	Increases
Related Circumscribing Circle - Coefficient of Variation (%)	CIRCLE_CV	29.77	32.65	31.66	33.27	29.99	30.75	32.10	29.22	28.98	31.64	31.10	32.11	31.97	0.06	Decreases
Contiguity Index - Mean	CONTIG_MN	0.402	0.311	0.345	0.300	0.399	0.377	0.320	0.349	0.363	0.339	0.360	0.365	0.375	-0.40	Increases
Contiguity Index - Area-Weighted Mean	CONTIG_AM	0.906	0.892	0.882	0.907	0.901	0.903	0.896	0.875	0.866	0.880	0.884	0.885	0.891	**-0.72**	Increases
Contiguity Index - Median	CONTIG_MD	0.361	0.233	0.285	0.200	0.310	0.333	0.250	0.286	0.304	0.278	0.300	0.333	0.337	-0.22	Increases
Contiguity Index - Range	CONTIG_RA	0.947	0.946	0.938	0.943	0.948	0.947	0.941	0.928	0.927	0.930	0.933	0.940	0.937	-0.64	Increases
Contiguity Index - Standard Deviation	CONTIG_SD	0.300	0.283	0.296	0.303	0.317	0.304	0.294	0.280	0.280	0.300	0.299	0.307	0.311	-0.46	Increases
Contiguity Index - Coefficient of Variation (%)	CONTIG_CV	74.65	91.09	85.89	101.11	79.56	80.46	91.85	80.16	77.03	88.37	83.03	83.90	82.82	0.15	Decreases
ISOLATION AND PROXIMITY																
Proximity Index - Mean	PROX_MN	197.58	189.66	245.39	441.01	267.19	292.81	240.07	425.55	255.64	351.03	300.76	136.80	232.43	0.07	Decreases
Proximity Index - Area-Weighted Mean	PROX_AM	124.85	141.08	178.10	95.43	42.76	139.69	136.83	216.72	187.90	244.96	215.16	96.45	134.38	0.64	Decreases
Proximity Index - Median	PROX_MD	1.00	0.85	1.16	0.60	1.67	0.61	0.65	4.00	3.20	4.07	1.80	0.85	1.05	0.42	Decreases
Proximity Index - Range	PROX_RA	4386.80	6069.93	7099.11	7398.50	6271.61	6473.11	5397.34	7821.60	7881.02	7983.45	8037.61	6008.60	7513.61	0.49	Decreases
Proximity Index - Standard Deviation	PROX_SD	698.54	783.90	915.86	1372.74	1048.07	1030.51	912.80	1365.90	1029.94	1301.15	1214.07	683.93	924.11	0.22	Decreases
Proximity Index - Coefficient of Variation (%)	PROX_CV	353.55	413.32	373.22	311.27	392.26	351.93	380.22	320.97	402.89	370.67	403.67	499.96	397.58	0.24	Decreases
Similarity Index - Mean	SIMI_MN	8437.36	9420.28	9704.88	10316.26	7748.55	7083.17	9253.79	12811.47	11727.30	12614.96	12686.66	11257.04	10940.62	0.50	Decreases
Similarity Index - Area-Weighted Mean	SIMI_AM	14553.01	15238.93	12838.56	13891.20	12434.68	15295.14	15280.29	11906.44	11044.83	10884.69	10687.34	12349.70	12937.59	-0.33	Increases
Similarity Index - Median	SIMI_MD	8771.20	4869.39	3190.77	5657.18	2694.32	2627.73	4187.20	12512.80	6348.99	8132.05	6592.86	4806.05	7040.00	0.00	Decreases
Similarity Index - Range	SIMI_RA	24839.20	27721.39	23815.30	26637.30	23039.51	23325.95	29181.78	30732.39	28978.27	26146.15	29622.80	29101.70	27344.20	0.47	Decreases
Similarity Index - Standard Deviation	SIMI_SD	7205.05	8537.80	9884.19	9941.71	8149.80	7872.73	9419.69	11589.27	11490.41	11758.64	12286.53	11145.11	10432.92	0.64	Decreases
Similarity Index - Coefficient of Variation (%)	SIMI_CV	85.39	90.63	101.85	96.37	105.18	111.15	101.79	90.46	97.98	93.21	96.85	99.01	95.36	0.26	Decreases
Euclidian Nearest Neighbour Distance - Mean (m)	ENN_MN	98.15	62.90	77.90	91.08	104.72	97.91	68.74	64.28	67.00	61.47	63.83	82.22	80.22	**-0.75**	Increases
Euclidian Nearest Neighbour Distance - Area-Weighted Mean (m)	ENN_AM	42.73	28.34	38.90	32.25	33.33	35.62	26.58	25.94	30.72	26.57	29.86	29.39	30.75	-0.66	Increases
Euclidian Nearest Neighbour Distance - Median (m)	ENN_MD	30.00	22.36	30.00	31.62	22.36	28.28	31.62	28.28	25.32	28.28	28.28	31.62	30.00	0.06	Decreases
Euclidian Nearest Neighbour Distance - Range (m)	ENN_RA	1563.54	1696.07	3097.08	1326.18	4881.72	3043.49	1211.46	1860.00	1078.82	1424.20	1924.45	1659.20	3993.59	-0.28	Increases
Euclidian Nearest Neighbour Distance - Standard Deviation (m)	ENN_SD	196.84	134.58	235.70	178.19	498.13	302.96	111.57	145.89	130.35	149.86	160.13	176.61	269.23	-0.45	Increases
Euclidian Nearest Neighbour Distance - Coefficient of Variation (%)	ENN_CV	200.55	213.96	302.58	195.64	475.70	309.44	162.30	226.97	194.54	243.80	250.85	214.79	335.62	-0.25	Increases
CONTRAST																
Contrast-Weighted Edge Density (m/ha)	CWED	22.56	19.39	25.70	16.35	22.24	17.32	18.62	26.44	29.39	25.21	21.82	20.56	22.29	0.30	Decreases
Total Edge Contrast Index (%)	TECI	8.63	6.73	8.37	6.23	8.17	6.46	6.61	8.29	8.76	8.13	7.21	6.83	7.69	-0.08	Increases
Edge Contrast Index - Mean (%)	ECON_MN	13.39	10.69	12.92	11.92	13.24	11.92	11.76	12.53	12.85	12.43	10.48	11.22	13.30	-0.36	Increases
Edge Contrast Index - Area-Weighted Mean (%)	ECON_AM	14.30	10.16	11.35	9.10	12.57	9.20	9.53	11.41	11.86	11.20	10.34	9.74	10.90	-0.48	Increases
Edge Contrast Index - Median (%)	ECON_MD	11.00	10.00	10.95	10.00	11.11	10.00	10.00	10.00	10.84	10.00	10.00	10.00	10.85	-0.48	Increases
Edge Contrast Index - Range (%)	ECON_RA	47.39	45.00	40.00	50.00	40.00	40.00	45.71	47.78	60.00	46.00	48.18	46.67	46.67	0.10	Decreases
Edge Contrast Index - Standard Deviation (%)	ECON_SD	10.93	6.59	7.21	8.52	9.18	7.54	8.29	7.36	7.51	6.79	6.25	7.54	8.39	**-0.81**	Increases
Edge Contrast Index - Coefficient of Variation (%)	ECON_CV	81.57	61.67	55.76	71.43	69.36	63.27	70.51	58.74	58.40	54.62	59.60	67.17	63.06	**-0.78**	Increases
CONTAGION AND INTERSPERSION																
Percentage of Like Adjacencies (%)	PLADJ	91.72	90.38	89.44	91.66	91.19	91.39	90.70	88.85	88.02	89.28	89.65	89.74	90.30	**-0.73**	Increases
Contagion (%)	CONTAG	60.89	63.99	70.64	71.71	63.89	68.81	70.26	73.44	71.34	75.53	74.10	70.33	71.44	**0.71**	Decreases
Aggregation Index (%)	AI	92.55	91.16	90.12	92.35	91.97	92.11	91.40	89.48	88.66	89.90	90.30	90.42	90.99	**-0.74**	Increases
DIVERSITY																
Shannon's Diversity Index	SHDI	1.213	1.054	0.799	0.838	1.091	0.928	0.852	0.691	0.732	0.639	0.688	0.815	0.796	**-0.77**	Increases
Simpson's Diversity Index	SIDI	0.660	0.585	0.476	0.500	0.615	0.551	0.514	0.413	0.405	0.386	0.389	0.455	0.457	**-0.74**	Increases
Modified Simpson's Diversity Index	MSIDI	1.078	0.880	0.647	0.692	0.953	0.800	0.722	0.533	0.519	0.488	0.493	0.607	0.611	**-0.76**	Increases
Shannon's Evenness Index	SHEI	0.677	0.588	0.446	0.468	0.609	0.518	0.475	0.385	0.409	0.356	0.384	0.455	0.444	**-0.77**	Increases
Simpson's Evenness Index	SIEI	0.791	0.702	0.572	0.600	0.737	0.661	0.617	0.496	0.486	0.463	0.467	0.546	0.549	**-0.74**	Increases
Modified Simpson's Evenness Index	MSIEI	0.601	0.491	0.361	0.386	0.532	0.446	0.403	0.297	0.290	0.272	0.275	0.339	0.341	**-0.76**	Increases

* Correlation Coefficient between metric value and water level for each year, significant correlations (i.e. values ≥ |0.7|) are bolded
^ Total vegetated wetland area (excludes open water)

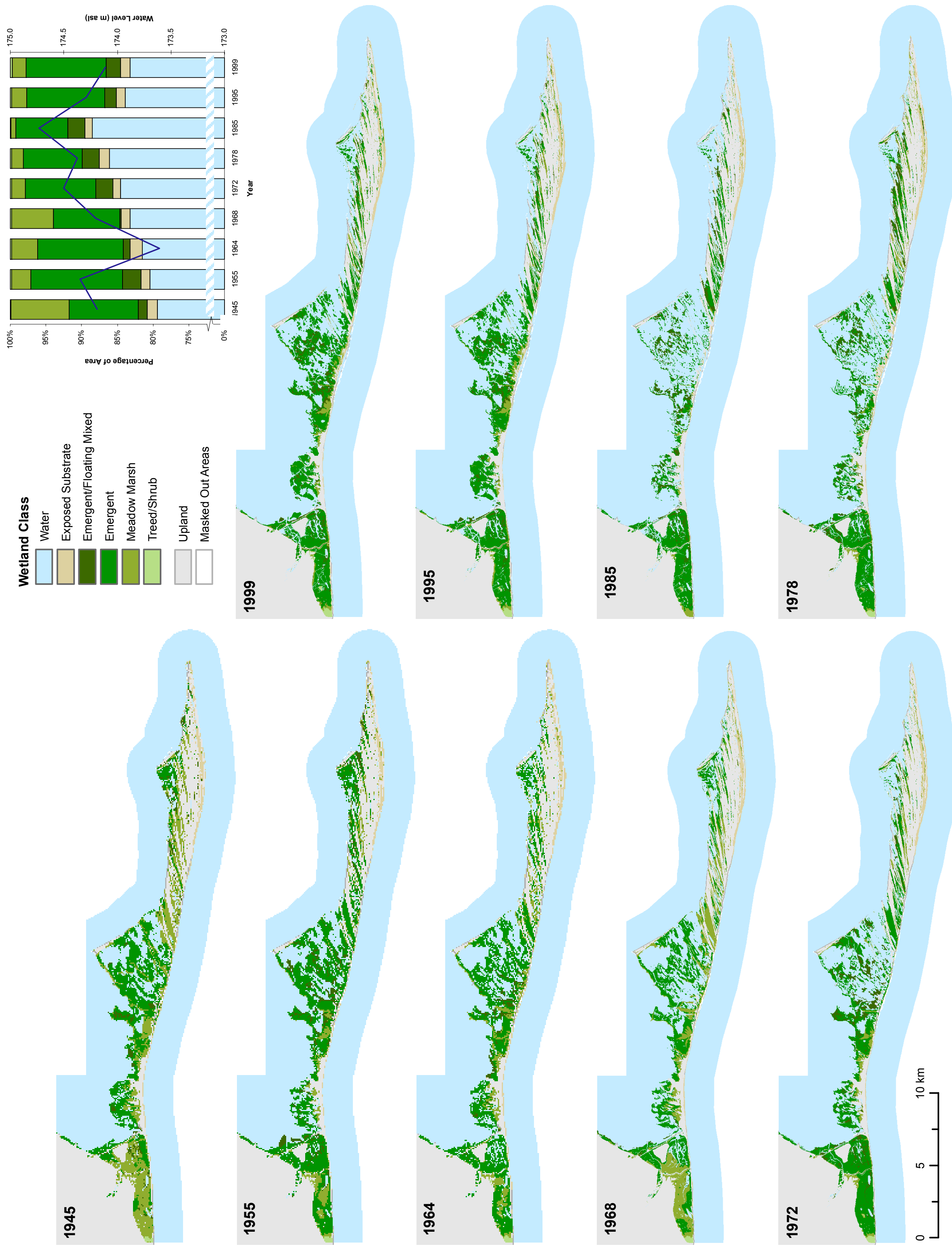

Figure 4.7 Historical distribution of wetland vegetation at Long Point, 1945-1999

Table 4.8 Landscape metrics for Long Point, 1945-1999

Landscape Metric	Code	Year and Water Level (m asl)									Correl. Coeff.*	Trend w/ ↓ Water
		1945	1955	1964	1968	1972	1978	1985	1995	1999		
		174.19	*174.35*	*173.61*	*174.20*	*174.50*	*174.37*	*174.73*	*174.29*	*174.11*		
AREA, DENSITY, EDGE												
Total Area^ (ha)	TA	4880.51	4634.7	4391.32	3978.37	3656.35	3286.29	2732.27	3822.68	3981.80	-0.62	Increases
Number of Patches (#)	NP	1652	2497	2336	2657	2758	3627	3887	3230	2934	0.57	Decreases
Patch Density (#/100 ha)	PD	6.97	10.54	9.86	11.21	11.64	15.30	16.40	13.63	12.38	0.57	Decreases
Patch Area - Mean (m)	AREA_MN	14.35	9.49	10.15	8.92	8.59	6.53	6.10	7.34	8.08	-0.45	Increases
Patch Area - Area-Weighted Mean (m)	AREA_AM	14502.65	14790.18	14958.03	15754.97	16312.84	16963.09	17809.00	16065.14	15860.16	0.67	Decreases
Patch Area - Median (m)	AREA_MD	0.15	0.07	0.11	0.08	0.09	0.11	0.08	0.08	0.12	-0.36	Increases
Patch Area - Range (m)	AREA_RA	18524.27	18710.66	18817.47	19315.84	19653.76	20044.91	20539.14	19503.06	19381.78	0.66	Decreases
Patch Area - Standard Deviation (m)	AREA_SD	455.93	374.57	389.44	374.78	374.32	332.88	329.48	343.27	357.85	-0.42	Increases
Patch Area - Coefficient of Variation (%)	AREA_CV	3177.78	3946.09	3838.28	4201.38	4355.70	5093.93	5403.35	4677.94	4429.79	0.59	Decreases
Radius of Gyration - Mean (m)	GYRATE_MN	64.94	47.59	51.80	44.43	42.48	39.78	32.95	38.96	44.71	-0.56	Increases
Radius of Gyration - Area-Weighted Mean (m)	GYRATE_AM	8088.42	8120.54	8140.55	8262.09	8330.81	8404.31	8566.60	8323.56	8266.34	0.68	Decreases
Radius of Gyration - Median (m)	GYRATE_MD	20.06	13.45	16.69	14.30	14.96	17.09	14.41	14.08	17.30	-0.38	Increases
Radius of Gyration - Range (m)	GYRATE_RA	10170.42	10140.94	10098.23	10020.38	9940.45	9857.36	9812.66	10006.35	10015.42	-0.62	Increases
Radius of Gyration - Standard Deviation (m)	GYRATE_SD	283.76	224.91	236.69	214.99	206.64	179.41	169.44	191.08	201.74	-0.52	Increases
Radius of Gyration - Coefficient of Variation (%)	GYRATE_CV	436.95	472.63	456.93	483.92	486.41	451.02	514.15	490.38	451.27	0.60	Decreases
Total Edge Length (m)	TE	700520	750230	840800	803340	790310	1042910	960300	905600	901330	0.24	Decreases
Edge Density (m/ha)	ED	29.56	31.65	35.47	33.89	33.34	44.00	40.52	38.21	38.03	0.24	Decreases
SHAPE												
Landscape Shape Index	LSI	23.39	24.20	25.67	25.06	24.85	28.95	27.61	26.72	26.65	0.24	Decreases
Perimeter-Area Ratio - Mean	PARA_MN	1976.65	2306.26	2129.07	2218.80	2214.61	2070.06	2213.34	2284.33	2077.25	0.32	Decreases
Perimeter-Area Ratio - Area-Weighted Mean	PARA_AM	90.35	94.55	102.19	99.03	97.93	119.24	112.27	107.66	107.30	0.24	Decreases
Perimeter-Area Ratio - Median	PARA_MD	1666.67	2333.33	2000.00	2000.00	2000.00	1800.00	2000.00	2250.00	1818.18	0.13	Decreases
Perimeter-Area Ratio - Range	PARA_RA	3977.16	3977.33	3973.32	3971.33	3971.21	3965.20	3964.04	3971.36	3971.03	-0.47	Increases
Perimeter-Area Ratio - Standard Deviation	PARA_SD	1288.34	1305.43	1262.61	1229.15	1270.18	1178.60	1171.58	1215.75	1229.56	-0.37	Increases
Perimeter-Area Ratio - Coefficient of Variation (%)	PARA_CV	65.18	56.60	59.30	55.40	57.35	56.94	52.93	53.22	59.19	-0.47	Increases
Shape Index - Mean	SHAPE_MN	1.69	1.57	1.62	1.56	1.55	1.59	1.52	1.58	1.60	-0.58	Increases
Shape Index - Area Weighted Mean	SHAPE_AM	7.11	7.02	8.28	8.92	9.09	11.09	11.80	9.13	9.03	0.51	Decreases
Shape Index - Median	SHAPE_MD	1.40	1.29	1.33	1.29	1.30	1.33	1.29	1.30	1.33	-0.41	Increases
Shape Index - Range	SHAPE_RA	8.18	7.43	10.14	9.05	9.66	11.32	12.32	12.17	13.06	0.13	Decreases
Shape Index - Standard Deviation	SHAPE_SD	0.98	0.83	0.92	0.84	0.80	0.83	0.79	0.86	0.87	-0.68	Increases
Shape Index - Coefficient of Variation (%)	SHAPE_CV	57.67	53.22	56.59	53.77	51.97	52.18	51.88	54.66	54.49	**-0.74**	Increases
Related Circumscribing Circle - Mean	CIRCLE_MN	0.654	0.622	0.639	0.629	0.622	0.635	0.614	0.628	0.630	-0.61	Increases
Related Circumscribing Circle - Area-Weighted Mean	CIRCLE_AM	0.849	0.849	0.851	0.851	0.848	0.847	0.845	0.843	0.842	-0.34	Increases
Related Circumscribing Circle - Median	CIRCLE_MD	0.682	0.633	0.673	0.644	0.642	0.657	0.618	0.644	0.637	-0.67	Increases
Related Circumscribing Circle - Range	CIRCLE_RA	0.780	0.751	0.784	0.744	0.799	0.780	0.769	0.779	0.733	0.07	Decreases
Related Circumscribing Circle - Standard Deviation	CIRCLE_SD	0.192	0.199	0.197	0.194	0.193	0.186	0.181	0.195	0.193	-0.60	Increases
Related Circumscribing Circle - Coefficient of Variation (%)	CIRCLE_CV	29.39	32.04	30.80	30.81	31.10	29.21	29.45	30.98	30.66	-0.21	Increases
Contiguity Index - Mean	CONTIG_MN	0.470	0.389	0.430	0.406	0.411	0.443	0.408	0.391	0.444	-0.32	Increases
Contiguity Index - Area-Weighted Mean	CONTIG_AM	0.974	0.973	0.971	0.972	0.972	0.966	0.968	0.969	0.969	-0.22	Increases
Contiguity Index - Median	CONTIG_MD	0.527	0.333	0.452	0.420	0.419	0.494	0.429	0.381	0.485	-0.24	Increases
Contiguity Index - Range	CONTIG_RA	0.993	0.993	0.992	0.992	0.992	0.990	0.990	0.992	0.992	-0.47	Increases
Contiguity Index - Standard Deviation	CONTIG_SD	0.319	0.318	0.310	0.300	0.310	0.289	0.285	0.295	0.302	-0.43	Increases
Contiguity Index - Coefficient of Variation (%)	CONTIG_CV	67.76	81.70	72.21	73.89	75.41	65.22	69.87	75.56	67.93	0.06	Decreases
ISOLATION AND PROXIMITY												
Proximity Index - Mean	PROX_MN	7865.17	5257.51	4612.00	10605.33	11346.82	8591.68	13560.64	8008.45	9077.87	**0.74**	Decreases
Proximity Index - Area-Weighted Mean	PROX_AM	1356.70	1081.42	1190.67	2457.56	3055.71	2347.91	2888.73	1200.62	1279.96	0.64	Decreases
Proximity Index - Median	PROX_MD	2.02	0.89	2.00	1.60	1.00	2.00	1.34	1.30	1.33	-0.51	Increases
Proximity Index - Range	PROX_RA	463108.00	467818.11	470438.13	482917.46	491436.75	501132.55	513698.16	487586.25	484551.25	0.66	Decreases
Proximity Index - Standard Deviation	PROX_SD	49353.77	41164.35	34964.41	56549.82	61377.81	53336.87	66983.97	49638.18	54635.22	**0.79**	Decreases
Proximity Index - Coefficient of Variation (%)	PROX_CV	627.50	782.96	758.12	533.22	540.93	620.80	493.96	619.82	601.85	-0.61	Increases
Similarity Index - Mean	SIMI_MN	242395.45	211745.96	254446.78	306789.27	405757.86	506722.01	626621.38	335925.12	380596.22	0.65	Decreases
Similarity Index - Area-Weighted Mean	SIMI_AM	349119.41	350944.43	328184.27	300661.16	272343.76	227077.75	200717.60	281445.42	292282.56	-0.62	Increases
Similarity Index - Median	SIMI_MD	2114.80	1136.80	1871.21	1862.47	2396.65	2279.45	35429.83	2173.28	2465.57	0.58	Decreases
Similarity Index - Range	SIMI_RA	1695486.60	1723224.30	1722671.23	1748451.01	1798445.14	1812065.94	1853796.38	1783022.28	1767143.43	0.69	Decreases
Similarity Index - Standard Deviation	SIMI_SD	498157.96	494954.14	510001.63	560175.87	638175.44	714042.00	762968.19	600805.77	631756.84	0.65	Decreases
Similarity Index - Coefficient of Variation (%)	SIMI_CV	205.51	233.75	200.44	182.59	157.28	140.91	121.76	178.85	165.99	-0.54	Increases
Euclidian Nearest Neighbour Distance - Mean (m)	ENN_MN	73.97	60.31	61.81	63.29	64.04	53.92	56.98	54.20	55.76	-0.19	Increases
Euclidian Nearest Neighbour Distance - Area-Weighted Mean (m)	ENN_AM	24.88	24.64	23.21	22.36	23.46	22.79	22.13	23.25	23.07	-0.18	Increases
Euclidian Nearest Neighbour Distance - Median (m)	ENN_MD	30.00	31.62	30.00	30.00	31.62	30.00	31.62	30.00	31.62	0.50	Decreases
Euclidian Nearest Neighbour Distance - Range (m)	ENN_RA	3110.26	2323.07	3958.54	3963.93	3674.89	2490.02	5126.75	3567.10	3545.28	0.14	Decreases
Euclidian Nearest Neighbour Distance - Standard Deviation (m)	ENN_SD	165.22	109.64	174.50	173.98	138.49	97.14	126.82	96.20	103.92	-0.45	Increases
Euclidian Nearest Neighbour Distance - Coefficient of Variation (%)	ENN_CV	223.37	181.79	282.31	274.87	216.25	180.14	222.55	177.50	186.39	-0.49	Increases
CONTRAST												
Contrast-Weighted Edge Density (m/ha)	CWED	6.84	6.33	8.49	8.53	6.82	9.10	7.76	7.95	7.77	-0.29	Increases
Total Edge Contrast Index (%)	TECI	11.25	10.07	12.73	13.10	10.56	12.10	10.81	11.44	11.22	-0.59	Increases
Edge Contrast Index - Mean (%)	ECON_MN	11.11	11.77	13.69	14.22	13.12	14.06	14.29	12.81	12.45	0.12	Decreases
Edge Contrast Index - Area-Weighted Mean (%)	ECON_AM	16.37	14.05	17.87	17.66	13.88	15.06	12.74	13.93	14.41	**-0.79**	Increases
Edge Contrast Index - Median (%)	ECON_MD	10.00	10.00	11.67	13.33	10.12	12.74	13.18	10.00	10.00	0.14	Decreases
Edge Contrast Index - Range (%)	ECON_RA	60.00	60.00	60.00	60.00	63.53	60.00	61.43	60.00	80.00	-0.09	Increases
Edge Contrast Index - Standard Deviation (%)	ECON_SD	10.26	9.49	10.70	10.37	9.68	9.31	9.67	9.39	9.09	-0.54	Increases
Edge Contrast Index - Coefficient of Variation (%)	ECON_CV	92.30	80.67	78.17	72.91	73.81	66.22	67.65	73.28	73.00	-0.37	Increases
CONTAGION AND INTERSPERSION												
Percentage of Like Adjacencies (%)	PLADJ	97.74	97.64	97.45	97.52	97.55	97.02	97.19	97.31	97.32	-0.24	Increases
Contagion (%)	CONTAG	77.11	78.12	78.68	80.16	80.84	81.11	83.78	80.29	79.69	0.64	Decreases
Aggregation Index (%)	AI	97.85	97.74	97.55	97.63	97.65	97.12	97.29	97.41	97.42	-0.24	Increases
DIVERSITY												
Shannon's Diversity Index	SHDI	0.736	0.696	0.667	0.618	0.596	0.563	0.479	0.607	0.627	-0.60	Increases
Simpson's Diversity Index	SIDI	0.353	0.335	0.320	0.295	0.274	0.250	0.211	0.284	0.294	-0.62	Increases
Modified Simpson's Diversity Index	MSIDI	0.436	0.408	0.386	0.350	0.320	0.288	0.237	0.334	0.348	-0.61	Increases
Shannon's Evenness Index	SHEI	0.411	0.389	0.372	0.345	0.332	0.314	0.267	0.339	0.350	-0.60	Increases
Simpson's Evenness Index	SIEI	0.424	0.402	0.384	0.354	0.329	0.300	0.254	0.341	0.353	-0.62	Increases
Modified Simpson's Evenness Index	MSIEI	0.243	0.228	0.215	0.195	0.179	0.161	0.133	0.186	0.194	-0.61	Increases

* Correlation Coefficient between metric value and water level for each year, significant correlations (i.e. values ≥ |0.7|) are bolded
^ Total vegetated wetland area (excludes open water)

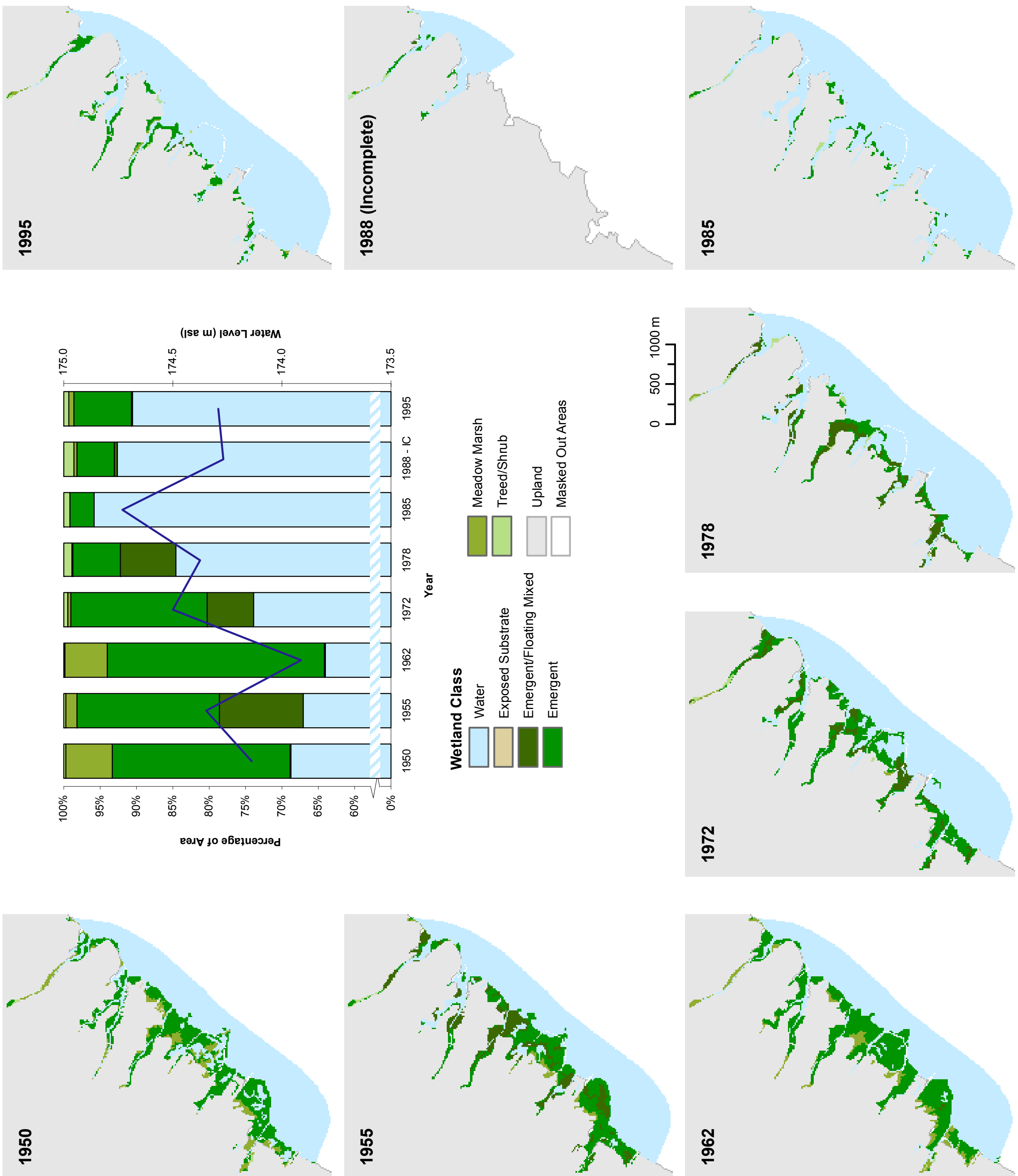

Figure 4.8 Historical distribution of wetland vegetation at Rondeau, 1950-1995

Table 4.9 Landscape metrics for Rondeau, 1955-1995

Landscape Metric	Code	Year and Water Level (m asl)								Correl. Coeff.*	Trend w/ ↓ Water
		1950	1955	1962	1972	1978	1985	1988[x]	1995		
		174.14	*174.35*	*173.91*	*174.50*	*174.37*	*174.73*	*174.27*	*174.29*		
AREA, DENSITY, EDGE											
Total Area^ (ha)	TA	95.93	100.86	110.23	80.01	47.3	12.79	4.65	29.07	**-0.70**	Increases
Number of Patches (#)	NP	194	142	115	119	145	127	51	125	-0.18	Increases
Patch Density (#/100 ha)	PD	63.34	46.36	37.54	38.85	47.34	41.46	86.73	40.81	-0.18	Increases
Patch Area - Mean (m)	AREA_MN	1.58	2.16	2.66	2.57	2.11	2.41	1.15	2.45	0.10	Decreases
Patch Area - Area-Weighted Mean (m)	AREA_AM	124.30	133.12	124.08	162.23	214.92	277.91	48.95	246.14	0.69	Decreases
Patch Area - Median (m)	AREA_MD	0.08	0.14	0.11	0.16	0.07	0.03	0.02	0.05	-0.22	Increases
Patch Area - Range (m)	AREA_RA	192.88	199.06	192.26	222.19	256.34	291.74	53.62	274.49	**0.70**	Decreases
Patch Area - Standard Deviation (m)	AREA_SD	13.92	16.81	17.98	20.27	21.20	25.78	7.42	24.44	0.64	Decreases
Patch Area - Coefficient of Variation (%)	AREA_CV	881.61	779.20	675.17	787.59	1003.71	1068.78	643.85	997.24	0.64	Decreases
Radius of Gyration - Mean (m)	GYRATE_MN	32.69	39.21	44.23	43.74	31.88	22.66	21.48	31.26	-0.56	Increases
Radius of Gyration - Area-Weighted Mean (m)	GYRATE_AM	723.72	753.92	711.32	742.86	869.18	968.19	356.51	894.98	0.69	Decreases
Radius of Gyration - Median (m)	GYRATE_MD	12.73	18.68	16.05	22.14	15.09	8.84	7.07	10.42	-0.14	Increases
Radius of Gyration - Range (m)	GYRATE_RA	1061.73	1063.45	1039.98	979.61	1017.62	1009.53	382.50	984.87	-0.47	Increases
Radius of Gyration - Standard Deviation (m)	GYRATE_SD	83.69	97.04	104.52	94.73	87.25	89.70	53.54	91.88	-0.39	Increases
Radius of Gyration - Coefficient of Variation (%)	GYRATE_CV	255.96	247.52	236.31	216.55	273.65	395.76	249.18	293.94	0.62	Decreases
Total Edge Length (m)	TE	39200	30980	27180	26780	21290	10300	4090	15610	-0.59	Increases
Edge Density (m/ha)	ED	127.98	101.14	88.74	87.43	69.51	33.63	69.56	50.96	-0.59	Increases
SHAPE											
Landscape Shape Index	LSI	11.38	10.21	9.67	9.61	8.83	7.26	5.30	8.02	-0.59	Increases
Perimeter-Area Ratio - Mean	PARA_MN	2391.28	2113.07	2271.38	2139.58	2410.19	2825.23	2870.25	2596.07	0.43	Decreases
Perimeter-Area Ratio - Area-Weighted Mean	PARA_AM	388.77	335.10	310.28	307.67	271.83	200.07	347.28	234.74	-0.59	Increases
Perimeter-Area Ratio - Median	PARA_MD	2000.00	1825.76	2000.00	1709.09	2444.44	3000.00	3000.00	2500.00	0.49	Decreases
Perimeter-Area Ratio - Range	PARA_RA	3851.94	3874.92	3850.31	3875.97	3868.15	3867.22	3785.19	3875.34	0.67	Decreases
Perimeter-Area Ratio - Standard Deviation	PARA_SD	1253.38	1244.65	1293.14	1312.03	1207.25	1120.81	1129.04	1223.92	-0.62	Increases
Perimeter-Area Ratio - Coefficient of Variation (%)	PARA_CV	52.41	58.90	56.93	61.32	50.09	39.67	39.34	47.14	-0.45	Increases
Shape Index - Mean	SHAPE_MN	1.61	1.64	1.71	1.76	1.57	1.37	1.43	1.55	-0.57	Increases
Shape Index - Area Weighted Mean	SHAPE_AM	4.50	4.20	4.28	4.11	4.79	5.48	3.74	4.87	0.54	Decreases
Shape Index - Median	SHAPE_MD	1.33	1.43	1.38	1.50	1.33	1.17	1.30	1.25	-0.29	Increases
Shape Index - Range	SHAPE_RA	4.19	5.10	4.40	3.61	4.26	4.66	2.92	4.63	0.02	Decreases
Shape Index - Standard Deviation	SHAPE_SD	0.84	0.82	0.85	0.82	0.66	0.57	0.57	0.76	**-0.74**	Increases
Shape Index - Coefficient of Variation (%)	SHAPE_CV	52.31	49.82	49.91	46.71	42.17	41.92	40.14	49.47	**-0.72**	Increases
Related Circumscribing Circle - Mean	CIRCLE_MN	0.611	0.642	0.639	0.646	0.640	0.577	0.590	0.604	-0.42	Increases
Related Circumscribing Circle - Area-Weighted Mean	CIRCLE_AM	0.824	0.830	0.804	0.818	0.819	0.801	0.829	0.812	-0.11	Increases
Related Circumscribing Circle - Median	CIRCLE_MD	0.641	0.682	0.682	0.682	0.682	0.576	0.587	0.608	-0.48	Increases
Related Circumscribing Circle - Range	CIRCLE_RA	0.633	0.592	0.681	0.592	0.623	0.571	0.553	0.617	**-0.95**	Increases
Related Circumscribing Circle - Standard Deviation	CIRCLE_SD	0.180	0.172	0.192	0.189	0.191	0.188	0.196	0.202	-0.05	Increases
Related Circumscribing Circle - Coefficient of Variation (%)	CIRCLE_CV	29.43	26.82	30.11	29.25	29.92	32.66	33.17	33.49	0.23	Decreases
Contiguity Index - Mean	CONTIG_MN	0.383	0.448	0.413	0.446	0.373	0.272	0.265	0.333	-0.44	Increases
Contiguity Index - Area-Weighted Mean	CONTIG_AM	0.894	0.908	0.916	0.916	0.926	0.946	0.906	0.936	0.58	Decreases
Contiguity Index - Median	CONTIG_MD	0.417	0.507	0.426	0.521	0.313	0.222	0.167	0.333	-0.38	Increases
Contiguity Index - Range	CONTIG_RA	0.959	0.966	0.959	0.966	0.964	0.964	0.941	0.966	0.65	Decreases
Contiguity Index - Standard Deviation	CONTIG_SD	0.295	0.298	0.310	0.314	0.290	0.260	0.264	0.288	-0.62	Increases
Contiguity Index - Coefficient of Variation (%)	CONTIG_CV	77.10	66.62	75.04	70.35	77.89	95.59	99.52	86.53	0.43	Decreases
ISOLATION AND PROXIMITY											
Proximity Index - Mean	PROX_MN	301.49	172.98	329.23	177.83	282.25	414.30	55.77	346.91	0.05	Decreases
Proximity Index - Area-Weighted Mean	PROX_AM	407.72	23.72	84.97	35.84	88.52	68.09	5.95	33.00	-0.36	Increases
Proximity Index - Median	PROX_MD	6.63	0.78	4.88	2.60	0.82	0.40	0.10	0.80	-0.69	Increases
Proximity Index - Range	PROX_RA	4824.78	4977.35	3853.00	4444.08	6408.75	7293.75	1073.01	5490.68	**0.75**	Decreases
Proximity Index - Standard Deviation	PROX_SD	854.06	790.32	895.10	759.54	1160.88	1458.23	225.47	1226.77	0.50	Decreases
Proximity Index - Coefficient of Variation (%)	PROX_CV	283.28	456.88	271.87	427.12	411.30	351.97	404.25	353.63	0.58	Decreases
Similarity Index - Mean	SIMI_MN	5652.28	6796.98	4600.03	7727.55	10346.71	14497.23	1672.64	10695.31	**0.82**	Decreases
Similarity Index - Area-Weighted Mean	SIMI_AM	8367.32	8797.39	9749.57	7746.55	5628.79	1722.22	442.08	3361.84	**-0.70**	Increases
Similarity Index - Median	SIMI_MD	1593.83	2131.33	1318.27	669.60	5329.00	11689.29	261.76	10980.00	0.53	Decreases
Similarity Index - Range	SIMI_RA	16633.95	19998.21	18561.52	21476.62	23687.44	26264.76	4832.28	24889.85	**0.73**	Decreases
Similarity Index - Standard Deviation	SIMI_SD	6474.32	7488.10	6005.58	8794.80	9673.79	9113.33	1870.86	9665.17	**0.72**	Decreases
Similarity Index - Coefficient of Variation (%)	SIMI_CV	114.54	110.17	130.56	113.81	93.50	62.86	111.85	90.37	**-0.81**	Increases
Euclidian Nearest Neighbour Distance - Mean (m)	ENN_MN	61.36	67.49	52.94	67.52	54.95	53.14	109.15	83.02	0.02	Decreases
Euclidian Nearest Neighbour Distance - Area-Weighted Mean (m)	ENN_AM	29.17	27.35	27.35	28.08	26.98	22.27	28.85	28.44	-0.66	Increases
Euclidian Nearest Neighbour Distance - Median (m)	ENN_MD	22.36	30.00	22.36	22.36	30.00	30.00	41.23	28.28	0.55	Decreases
Euclidian Nearest Neighbour Distance - Range (m)	ENN_RA	3317.44	2160.02	752.01	856.81	529.18	432.55	742.43	1641.84	-0.36	Increases
Euclidian Nearest Neighbour Distance - Standard Deviation (m)	ENN_SD	245.21	187.97	105.71	128.52	74.76	72.50	156.54	182.27	-0.37	Increases
Euclidian Nearest Neighbour Distance - Coefficient of Variation (%)	ENN_CV	399.63	278.53	199.68	190.35	136.05	136.42	143.41	219.56	-0.46	Increases
CONTRAST											
Contrast-Weighted Edge Density (m/ha)	CWED	30.23	15.01	19.46	14.49	14.54	10.07	28.95	13.90	-0.67	Increases
Total Edge Contrast Index (%)	TECI	11.59	6.42	8.79	6.58	7.19	6.05	10.42	7.56	**-0.70**	Increases
Edge Contrast Index - Mean (%)	ECON_MN	13.48	9.44	11.84	9.75	11.97	15.72	29.98	15.62	0.18	Decreases
Edge Contrast Index - Area-Weighted Mean (%)	ECON_AM	14.58	8.40	12.67	8.71	8.90	7.27	12.45	8.97	**-0.81**	Increases
Edge Contrast Index - Median (%)	ECON_MD	11.91	8.46	10.00	8.71	8.33	10.00	27.14	11.25	-0.34	Increases
Edge Contrast Index - Range (%)	ECON_RA	53.04	35.00	40.00	55.00	60.00	60.00	66.67	60.00	0.51	Decreases
Edge Contrast Index - Standard Deviation (%)	ECON_SD	8.49	6.67	7.09	7.43	11.18	12.64	18.72	12.37	0.50	Decreases
Edge Contrast Index - Coefficient of Variation (%)	ECON_CV	62.95	70.67	59.83	76.22	93.44	80.39	62.43	79.20	0.65	Decreases
CONTAGION AND INTERSPERSION											
Percentage of Like Adjacencies (%)	PLADJ	90.28	91.62	92.24	92.31	93.20	95.00	91.32	94.13	0.59	Decreases
Contagion (%)	CONTAG	69.94	64.29	67.73	70.06	79.26	92.31	85.43	84.56	0.61	Decreases
Aggregation Index (%)	AI	91.11	92.52	93.10	93.16	94.01	95.65	92.97	94.86	0.58	Decreases
DIVERSITY											
Shannon's Diversity Index	SHDI	0.807	0.919	0.831	0.764	0.577	0.197	0.363	0.373	-0.61	Increases
Simpson's Diversity Index	SIDI	0.465	0.499	0.498	0.415	0.275	0.081	0.149	0.175	-0.66	Increases
Modified Simpson's Diversity Index	MSIDI	0.625	0.690	0.689	0.536	0.321	0.084	0.161	0.192	-0.66	Increases
Shannon's Evenness Index	SHEI	0.450	0.571	0.517	0.475	0.322	0.110	0.203	0.232	-0.60	Increases
Simpson's Evenness Index	SIEI	0.557	0.623	0.622	0.519	0.330	0.097	0.179	0.218	-0.66	Increases
Modified Simpson's Evenness Index	MSIEI	0.349	0.429	0.428	0.333	0.179	0.047	0.090	0.119	-0.65	Increases

[x] 1988 was exluded from the trend analysis since the coverage was incomplete
* Correlation Coefficient between metric value and water level for each year, significant correlations (i.e. values ≥ |0.7|) are bolded
^ Total vegetated wetland area (excludes open water)

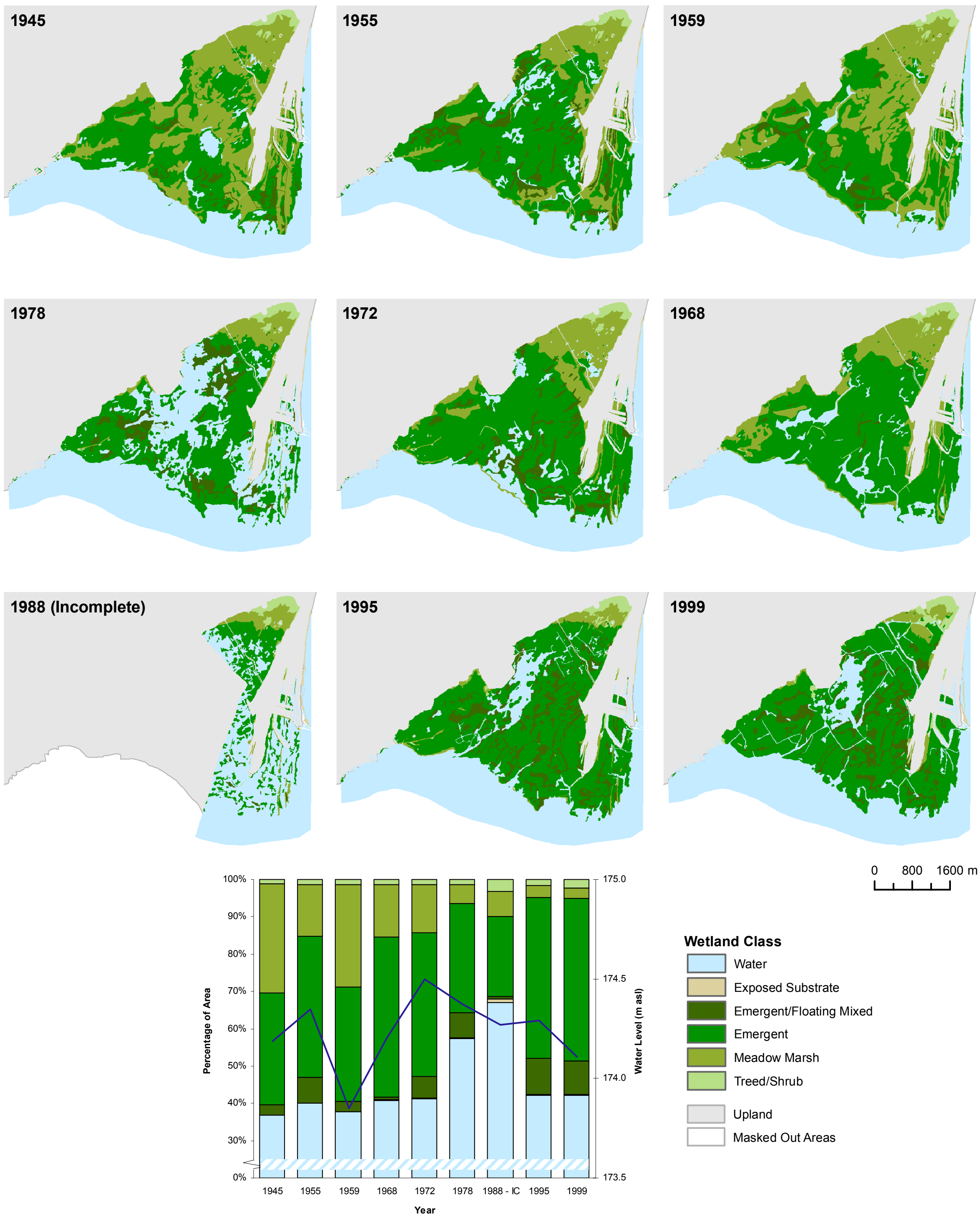

Figure 4.9 Historical distribution of wetland vegetation at Turkey Point, 1945-1999

Table 4.10 Landscape metrics for Turkey Point, 1945-1999

Landscape Metric	Code	Year and Water Level (m asl)									Correl.	Trend w/
		1945	1955	1959	1962	1972	1978	1988[x]	1995	1999	Coeff.*	↓ Water
		174.19	*174.35*	*173.85*	*173.91*	*174.50*	*174.37*	*174.27*	*174.29*	*174.11*		
AREA, DENSITY, EDGE												
Total Area^ (ha)	TA	1153.12	1096.72	1138.19	1080.45	1075.31	777.17	256.81	1055.28	1056.11	-0.39	Increases
Number of Patches (#)	NP	269	283	263	243	278	367	311	398	299	0.47	Decreases
Patch Density (#/100 ha)	PD	14.72	15.49	14.40	13.30	15.22	20.09	40.18	21.79	16.37	0.47	Decreases
Patch Area - Mean (m)	AREA_MN	6.79	6.46	6.95	7.52	6.57	4.98	2.49	4.59	6.11	-0.53	Increases
Patch Area - Area-Weighted Mean (m)	AREA_AM	337.18	424.21	365.08	575.63	467.68	584.55	254.79	373.08	348.20	0.13	Decreases
Patch Area - Median (m)	AREA_MD	0.21	0.31	0.19	0.11	0.28	0.22	0.15	0.16	0.25	0.64	Decreases
Patch Area - Range (m)	AREA_RA	633.55	649.68	661.44	739.86	651.04	994.29	430.01	682.61	706.00	0.15	Decreases
Patch Area - Standard Deviation (m)	AREA_SD	47.37	51.93	49.88	65.35	55.05	53.71	25.06	41.13	45.72	-0.20	Increases
Patch Area - Coefficient of Variation (%)	AREA_CV	697.48	804.44	718.02	869.28	837.66	1079.02	1006.87	895.97	748.25	0.42	Decreases
Radius of Gyration - Mean (m)	GYRATE_MN	64.24	66.23	64.40	54.54	62.55	48.47	38.49	49.43	64.95	-0.14	Increases
Radius of Gyration - Area-Weighted Mean (m)	GYRATE_AM	1078.97	1103.08	1069.56	1253.90	1173.31	1193.13	638.57	1069.91	1015.48	0.08	Decreases
Radius of Gyration - Median (m)	GYRATE_MD	23.20	29.98	21.92	15.92	27.01	22.05	18.39	21.04	25.92	0.60	Decreases
Radius of Gyration - Range (m)	GYRATE_RA	1790.86	1802.79	1785.94	1758.89	1799.49	1811.09	910.66	1786.38	1752.50	0.66	Decreases
Radius of Gyration - Standard Deviation (m)	GYRATE_SD	157.45	144.04	154.58	154.42	146.18	119.41	78.26	126.25	142.50	-0.54	Increases
Radius of Gyration - Coefficient of Variation (%)	GYRATE_CV	245.11	217.48	240.03	283.14	233.70	246.36	203.32	255.41	219.40	-0.40	Increases
Total Edge Length (m)	TE	167350	167790	150680	120430	149920	181440	101830	231020	201130	0.36	Decreases
Edge Density (m/ha)	ED	91.60	91.84	82.48	65.92	82.06	99.31	131.57	126.45	110.09	0.36	Decreases
SHAPE												
Landscape Shape Index	LSI	13.59	13.62	12.62	10.85	12.58	14.42	13.92	17.32	15.57	0.36	Decreases
Perimeter-Area Ratio - Mean	PARA_MN	1703.89	1620.45	1787.71	2025.22	1610.24	1648.61	1998.67	2027.10	1765.51	-0.55	Increases
Perimeter-Area Ratio - Area-Weighted Mean	PARA_AM	218.85	219.33	200.60	167.48	199.77	234.27	331.93	288.55	255.83	0.36	Decreases
Perimeter-Area Ratio - Median	PARA_MD	1463.41	1185.19	1500.00	1764.71	1335.84	1356.32	1571.43	1839.74	1384.62	-0.44	Increases
Perimeter-Area Ratio - Range	PARA_RA	3943.05	3942.83	3942.28	3929.75	3942.31	3873.96	3831.73	3918.37	3881.45	-0.11	Increases
Perimeter-Area Ratio - Standard Deviation	PARA_SD	1101.78	1136.06	1189.65	1197.70	1113.96	1074.81	1181.56	1145.15	1230.89	**-0.71**	Increases
Perimeter-Area Ratio - Coefficient of Variation (%)	PARA_CV	64.66	70.11	66.55	59.14	69.18	65.19	59.12	56.49	69.72	0.25	Decreases
Shape Index - Mean	SHAPE_MN	1.79	1.81	1.73	1.68	1.74	1.62	1.61	1.85	1.85	0.08	Decreases
Shape Index - Area Weighted Mean	SHAPE_AM	5.25	5.70	5.03	5.93	5.57	7.48	6.72	7.37	6.47	0.36	Decreases
Shape Index - Median	SHAPE_MD	1.43	1.50	1.40	1.33	1.43	1.33	1.36	1.49	1.47	0.32	Decreases
Shape Index - Range	SHAPE_RA	8.20	9.42	6.64	7.41	8.29	8.93	7.72	13.61	8.24	0.48	Decreases
Shape Index - Standard Deviation	SHAPE_SD	1.12	0.99	0.97	1.02	0.98	0.89	0.88	1.33	1.17	-0.02	Increases
Shape Index - Coefficient of Variation (%)	SHAPE_CV	62.89	54.58	55.81	60.93	56.36	55.08	54.68	72.11	62.84	-0.08	Increases
Related Circumscribing Circle - Mean	CIRCLE_MN	0.663	0.676	0.661	0.641	0.652	0.622	0.630	0.661	0.651	-0.02	Increases
Related Circumscribing Circle - Area-Weighted Mean	CIRCLE_AM	0.803	0.774	0.781	0.746	0.756	0.751	0.630	0.780	0.746	-0.04	Increases
Related Circumscribing Circle - Median	CIRCLE_MD	0.687	0.706	0.682	0.669	0.664	0.618	0.623	0.694	0.667	-0.16	Increases
Related Circumscribing Circle - Range	CIRCLE_RA	0.667	0.700	0.656	0.664	0.669	0.695	0.617	0.654	0.645	0.51	Decreases
Related Circumscribing Circle - Standard Deviation	CIRCLE_SD	0.170	0.179	0.182	0.185	0.175	0.168	0.178	0.187	0.184	-0.50	Increases
Related Circumscribing Circle - Coefficient of Variation (%)	CIRCLE_CV	25.56	26.43	27.57	28.86	26.84	26.94	28.31	28.21	28.22	-0.49	Increases
Contiguity Index - Mean	CONTIG_MN	0.537	0.559	0.517	0.461	0.562	0.551	0.463	0.459	0.530	0.54	Decreases
Contiguity Index - Area-Weighted Mean	CONTIG_AM	0.938	0.938	0.943	0.952	0.943	0.933	0.906	0.919	0.928	-0.37	Increases
Contiguity Index - Median	CONTIG_MD	0.576	0.653	0.574	0.500	0.614	0.609	0.557	0.500	0.611	0.47	Decreases
Contiguity Index - Range	CONTIG_RA	0.983	0.983	0.983	0.979	0.983	0.964	0.952	0.976	0.967	-0.12	Increases
Contiguity Index - Standard Deviation	CONTIG_SD	0.278	0.285	0.297	0.290	0.281	0.269	0.288	0.279	0.304	-0.65	Increases
Contiguity Index - Coefficient of Variation (%)	CONTIG_CV	51.76	50.99	57.41	63.04	50.02	48.75	62.20	60.81	57.27	**-0.70**	Increases
ISOLATION AND PROXIMITY												
Proximity Index - Mean	PROX_MN	718.10	813.70	643.39	1567.41	1155.09	1009.20	256.43	1003.48	1243.98	-0.08	Increases
Proximity Index - Area-Weighted Mean	PROX_AM	1441.34	701.49	508.86	663.21	1245.71	499.05	153.02	2597.37	3848.25	0.06	Decreases
Proximity Index - Median	PROX_MD	4.76	1.00	4.69	0.80	0.69	2.40	1.40	4.40	3.00	-0.34	Increases
Proximity Index - Range	PROX_RA	15839.00	16242.25	16536.25	18507.28	16432.69	24876.13	11360.95	17097.00	17655.80	0.16	Decreases
Proximity Index - Standard Deviation	PROX_SD	2013.24	2862.54	1957.25	4341.66	3495.91	3596.53	1314.77	2873.29	3303.27	0.14	Decreases
Proximity Index - Coefficient of Variation (%)	PROX_CV	280.36	351.79	304.21	277.00	302.65	356.37	512.71	286.33	265.54	0.47	Decreases
Similarity Index - Mean	SIMI_MN	18733.80	23774.88	17685.72	32511.80	28418.72	39323.07	15574.16	24266.80	25326.29	0.32	Decreases
Similarity Index - Area-Weighted Mean	SIMI_AM	56713.20	63131.50	54044.59	68315.98	65596.61	62320.21	17164.06	63568.38	59991.05	0.35	Decreases
Similarity Index - Median	SIMI_MD	12525.56	9275.20	7932.00	18497.08	13540.97	18638.80	7197.60	16538.46	13413.32	0.17	Decreases
Similarity Index - Range	SIMI_RA	93749.72	87787.49	99055.76	101104.16	120674.67	113994.89	41677.60	120984.84	101607.87	0.44	Decreases
Similarity Index - Standard Deviation	SIMI_SD	19904.17	25163.97	20206.17	28784.85	28196.68	36591.99	15890.49	23478.28	28397.77	0.37	Decreases
Similarity Index - Coefficient of Variation (%)	SIMI_CV	106.25	105.84	114.25	88.54	99.22	93.05	102.03	96.75	112.13	-0.25	Increases
Euclidian Nearest Neighbour Distance - Mean (m)	ENN_MN	59.98	65.69	88.85	77.33	73.23	64.31	51.54	62.34	66.75	-0.63	Increases
Euclidian Nearest Neighbour Distance - Area-Weighted Mean (m)	ENN_AM	25.60	29.45	29.28	25.94	29.85	26.25	24.91	23.86	27.26	0.09	Decreases
Euclidian Nearest Neighbour Distance - Median (m)	ENN_MD	31.62	41.23	31.62	31.62	41.23	36.06	30.00	28.28	31.62	0.63	Decreases
Euclidian Nearest Neighbour Distance - Range (m)	ENN_RA	782.12	384.97	3419.83	920.05	912.31	921.54	956.22	2612.30	665.93	-0.45	Increases
Euclidian Nearest Neighbour Distance - Standard Deviation (m)	ENN_SD	85.26	65.92	253.36	117.07	109.23	96.92	82.53	153.99	89.00	-0.58	Increases
Euclidian Nearest Neighbour Distance - Coefficient of Variation (%)	ENN_CV	142.15	100.35	285.16	151.39	149.16	150.71	160.11	247.00	133.34	-0.45	Increases
CONTRAST												
Contrast-Weighted Edge Density (m/ha)	CWED	17.72	16.84	17.28	14.12	15.36	18.67	32.11	19.76	18.99	0.19	Decreases
Total Edge Contrast Index (%)	TECI	13.93	13.21	14.63	13.91	13.05	13.83	16.03	12.19	13.03	-0.60	Increases
Edge Contrast Index - Mean (%)	ECON_MN	14.22	12.67	14.69	14.16	14.86	14.99	23.02	14.98	15.20	-0.05	Increases
Edge Contrast Index - Area-Weighted Mean (%)	ECON_AM	15.39	14.57	16.68	16.54	14.59	16.28	18.65	13.97	15.12	-0.68	Increases
Edge Contrast Index - Median (%)	ECON_MD	15.00	10.06	15.85	12.50	13.33	16.00	20.00	10.95	10.88	-0.18	Increases
Edge Contrast Index - Range (%)	ECON_RA	40.00	40.00	40.00	40.88	40.00	60.00	75.00	60.00	60.00	0.21	Decreases
Edge Contrast Index - Standard Deviation (%)	ECON_SD	8.66	8.11	8.87	9.21	9.58	8.25	14.55	10.16	11.11	-0.11	Increases
Edge Contrast Index - Coefficient of Variation (%)	ECON_CV	60.87	64.06	60.42	65.02	64.47	55.05	63.21	67.84	73.08	-0.10	Increases
CONTAGION AND INTERSPERSION												
Percentage of Like Adjacencies (%)	PLADJ	94.53	94.52	94.99	95.81	95.01	94.14	91.70	92.79	93.60	-0.36	Increases
Contagion (%)	CONTAG	59.36	58.64	59.57	64.50	59.75	63.25	69.50	60.48	61.22	-0.28	Increases
Aggregation Index (%)	AI	94.97	94.96	95.43	96.23	95.45	94.57	92.33	93.21	94.03	-0.36	Increases
DIVERSITY												
Shannon's Diversity Index	SHDI	1.243	1.258	1.249	1.103	1.236	1.083	0.987	1.145	1.141	0.08	Decreases
Simpson's Diversity Index	SIDI	0.688	0.673	0.687	0.628	0.662	0.578	0.502	0.626	0.622	-0.20	Increases
Modified Simpson's Diversity Index	MSIDI	1.165	1.118	1.163	0.989	1.085	0.862	0.698	0.983	0.973	-0.20	Increases
Shannon's Evenness Index	SHEI	0.693	0.702	0.697	0.616	0.690	0.604	0.475	0.639	0.637	0.08	Decreases
Simpson's Evenness Index	SIEI	0.826	0.808	0.825	0.754	0.794	0.693	0.574	0.751	0.747	-0.20	Increases
Modified Simpson's Evenness Index	MSIEI	0.650	0.624	0.649	0.552	0.605	0.481	0.336	0.548	0.543	-0.20	Increases

[x] 1988 was exluded from the trend analysis since the coverage was incomplete
* Correlation Coefficient between metric value and water level for each year, significant correlations (i.e. values ≥ |0.7|) are bolded
^ Total vegetated wetland area (excludes open water)

Table 4.11 Wetland vegetation response on Lake Erie

Declining Water Levels	Rising Water Levels	Indicated By
LANDSCAPE ANALYSIS		
• Vegetated wetland area increased (except Dunnville)	• Vegetated wetland area decreased (except Dunnville)	• TA
• Wetland less fragmented, fewer patches	• Wetland more fragmented, more patches	• NP, PD
• Patches larger in size	• Patches smaller in size	• AREA_MN
• Patches more elongated and less compact	• Patches more compact	• GYRATE_, SHAPE_
• Wetland more homogeneous (except Rondeau)	• Wetland more heterogeneous	• TE, ED
• Patches more aggregated, regular in shape	• Patches more disaggregated, irregular in shape	• LSI
• Patches more elongated and linear	• Patches more circular	• CIRCLE_MN
• Patches more isolated and fragmented in distribution (Long Point)	• Patches more contiguous and less fragmented in distribution (Long Point)	• CONTIG_MN
• Neighbourhood occupied by less patches of the same and similar class types	• Neighbourhood occupied by more patches of the same and similar class types	• PROX_, SIMI_
• Similar patches located further from each other	• Similar patches located closer to each other	• ENN_
• More edge contrast	• Less edge contrast	• CWED, TECI
• Patches more aggregated, the proportion of like adjacencies increased (i.e. patches of similar wetland classes were more equally adjacent to each other)	• Patches more disaggregated, the proportion of like adjacencies decreased	• PLADJ, AI
• Patch types more interspersed	• Patch types less interspersed	• CONTAG
• Wetlands more diverse, proportion and distribution of patch type area more even across landscape	• Wetlands less diverse, distribution of area among different patch types more uneven	• SHDI, SIDI, MSIDI, SHEI, SIEI, MSIEI
CLASS ANALYSIS		
• Open water and emergent/floating mixed decreased in area	• Open water and emergent/floating mixed increased in area	• CA
• Emergent and meadow marsh increased	• Emergent and meadow march decreased	
• Substrate exposed along shore (Long Point)	• Shoreline areas flooded by lake water	
• Patches of exposed substrate and emergent/ floating mixed (Long Point), open water, emergent, and meadow marsh (Rondeau) less fragmented	• Patches of exposed substrate and emergent/ floating mixed (Long Point), open water, emergent, and meadow marsh (Rondeau) more fragmented	• TE, ED
• Patches of meadow marsh (Long Point) and treed/shrub (Rondeau) more fragmented	• Patches of meadow marsh (Long Point) and treed/shrub (Rondeau) less fragmented	
• Patches of emergent/floating mixed (Long Point) more regular and aggregated in shape	• Patches of emergent/floating mixed (Long Point) more irregular and disaggregated in shape	• LSI, AREA_, GYRATE_, SHAPE
• Patches of emergent vegetation larger and contiguous in size; more elongated, less compact, and irregular in shape (Long Point, Rondeau)	• Patches of emergent vegetation smaller and more fragmented in size; more compact and regular in shape	
• Patches of exposed substrate larger, less compact, and elongated	• Patches of exposed substrate smaller, more compact	• GYRATE_MN
• Patches of meadow marsh larger, more contiguous, irregular, and less compact	• Patches of meadow marsh smaller, more compact	• AREA_, CONTIG_, PROX_
• Patches of open water had more contrast in class edge	• Patches of open water had less contrast in class edge	• TECI
• Patches of open water more isolated and less fragmented in distribution	• Patches of open water less isolated and more disaggregated in distribution	• PROX_, SIMI_,
• Patches of exposed substrate located closer to each other in distribution	• Patches of exposed substrate located further from each other in distribution	• ENN_MN
• Patches of emergent/floating mixed located further from each other (Long Point and Turkey Point)	• Patches of emergent/floating mixed located further from each other (Rondeau)	
• Patches of emergent more aggregated in distribution across the wetland and proportion of like adjacencies increases	• Patches of emergent and meadow marsh more dispersed in distribution across the wetland	• PLADJ, AI
• Patches of meadow marsh aggregated in distribution (distribution of patch type adjacencies less proportionate)		
SPATIAL ANALYSIS		
• Lakeward migration of wetland communities	• Landward migration of wetland communities	• Visual Observation
• Succession to drier communities	• Succession to wetter communities	
• Substrate exposed along shore	• Shoreline areas flooded by high lake water	

4.2.3 Lake Huron (unregulated water levels, fens)

Lake Huron wetland study sites consisted of both marsh and fen communities that were directly or indirectly affected by unregulated lake levels. Changes in wetland vegetation at Baie du Dore (Figure 4.10; Table 4.12) Howdenvale (Figure 4.11; Table 4.13), and Oliphant (Figure 4.12; Table 4.14) were strongly correlated with water level fluctuations at both the landscape and class level. Class metrics tables are provided in Appendix 4.5.

There were several key changes in the composition and configuration of the Lake Huron wetlands as water levels declined. The total area of vegetated wetland increased as drier wetland vegetation expanded; emergent, meadow marsh, and fen vegetation expanded forming larger patches of vegetation while alvar and substrate were exposed along the shore. Generally, the Lake Huron fens were more heterogeneous and communities within the landscape were more fragmented as water levels declined. Patches of vegetation were less compact, elongated, and linear in shape. There was more interspersion in the fens as patches of fen vegetation were disaggregated in distribution (i.e. located in neighbourhoods of less similar wetland communities). There was also more diversity, or evenness, in the Lake Huron fens as water levels declined. The proportion, distribution, and abundance of area among different wetland communities became more even, indicating that the wetland was dominated by a greater number of communities.

Community transitions were not as clear as in the marsh communities along Lakes Erie and Ontario. As water levels fluctuated, there were the typical transitions between the wetland marsh communities but also transitions between wetland and fen communities. Although changes within fen communities were likely more closely linked to groundwater supply, lake level fluctuations do influence groundwater levels which in turn affects the composition and configuration of fen communities along the Lake Huron shoreline, a few trends were observed in the fen communities in relation to lake level fluctuations. The findings of the spatiotemporal analysis for Lake Huron marsh and fen wetlands are summarized in Table 4.15.

4.2.4 Summary of Findings

Spatiotemporal analysis of marsh and fen communities within wetlands on Lakes Huron, Erie, and Ontario identified several similarities and differences in response to lower lake levels. On all three lakes, the total area of vegetated wetland increased. As water levels declined, drier wetland vegetation communities expanded lakeward along the moisture gradient. Emergent, meadow marsh, and treed/shrub vegetation expanded forming larger, more continuous stands of vegetation. Conversely, the area of open water and emergent/floating mixed vegetation communities decreased. As lake levels receded, areas of sandy substrate and alvar were also exposed along the shore. Patches of vegetation became more elongated, linear, and less compact in shape. The wetland landscapes of Lakes Erie and Ontario study sites became less fragmented and more homogeneous as water levels declined, whereas the wetlands on Lake Huron became more fragmented and heterogeneous. Patches within the Lake Ontario wetlands became more aggregated and were more likely located next to patches of similar vegetation communities (i.e. there was less contrast in class edge). The Lake Ontario wetlands also became less diverse and the distribution of area among different patch types was less proportionate. In contrast, the Lake Erie wetlands and Lake Huron fens were more diverse as the area of different wetland communities became more even in the landscape. Patches within these wetlands became more disaggregated and interspersed, and the proportion of like adjacencies decreased. Similar patches of vegetation were more fragmented and less contiguous. Differences in vegetation responses between Lakes Ontario and Erie may be due to the compression of extremes and alteration of seasonal cycles of Lake Ontario water level fluctuations from water level regulation. Finally, the wetlands on Lakes Erie and Huron were also more diverse in that single community dominance decreased. During high water level periods, the water class dominated the wetland landscape. As water levels declined, the water class was less dominant as the drier vegetation communities expanded. The proportion, distribution, and abundance of area between different community types became more equal, and as a result, evenness in the wetlands increased. This likely was due to the large amount of open water included in the analysis for wetlands in Lakes Erie and Huron. The key findings of the analysis provided an indication of how wetland vegetation may respond to water level declines with projected future climate change.

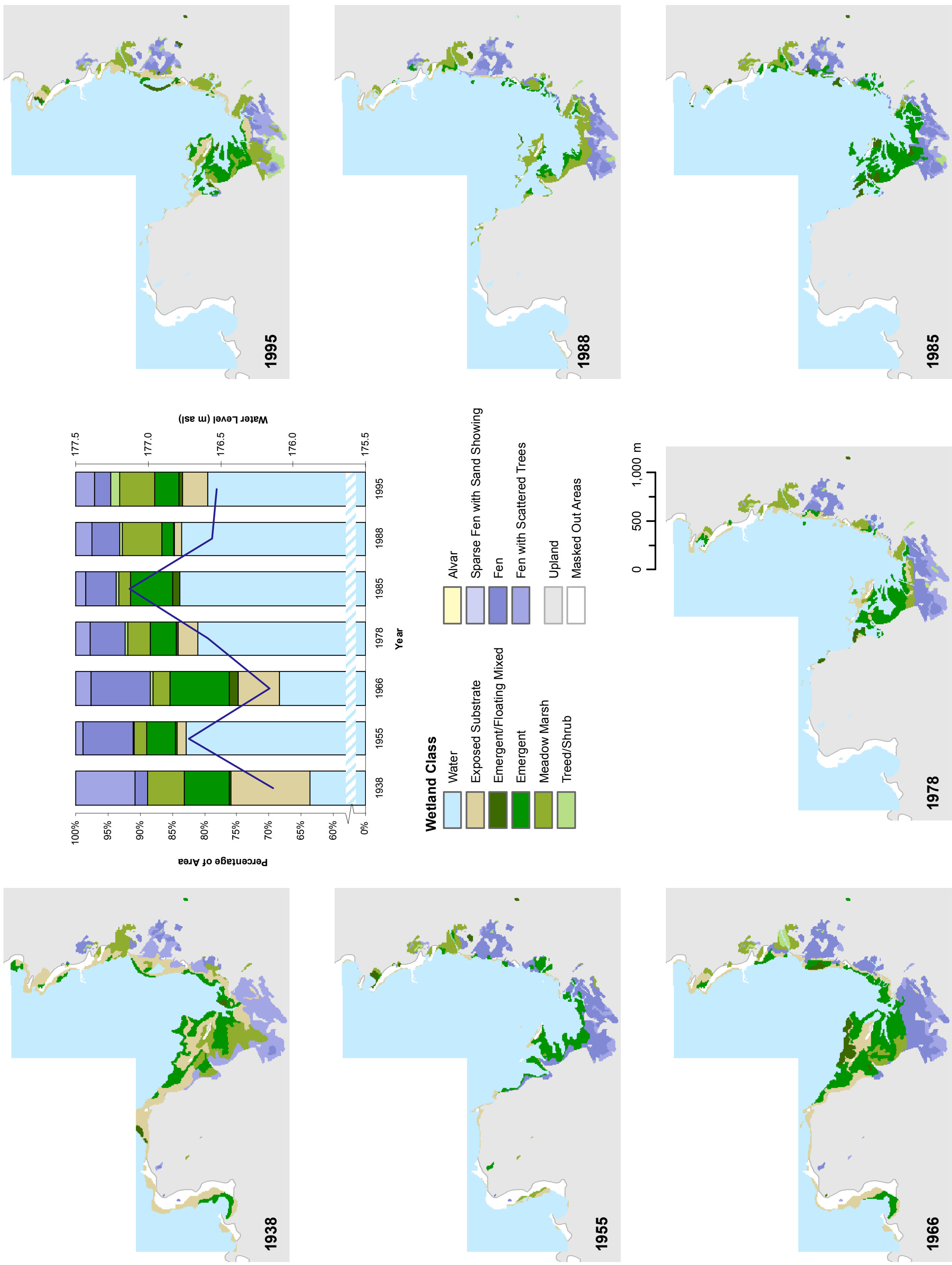

Figure 4.10 Historical distribution of wetland vegetation at Baie du Dore, 1938-1995

Table 4.12 Landscape metrics for Baie Du Dore, 1938-1995

Landscape Metric	**Code**	**Year and Water Level (m asl)**							**Correl.**	**Trend w/**
		1938	**1955**	**1966**	**1978**	**1985**	**1988**	**1995**	**Coeff.***	**↓ Water**
		176.14	***176.72***	***176.16***	***176.59***	***177.13***	***176.56***	***176.53***		
AREA, DENSITY, EDGE										
Total Area^ (ha)	TA	134.55	63.54	116.96	69.98	59.68	60.62	75.98	**-0.85**	Increases
Number of Patches (#)	NP	95	136	111	148	131	142	159	0.53	Decreases
Patch Density (#/100 ha)	PD	25.68	36.77	30.01	40.01	35.42	38.39	42.99	0.53	Decreases
Patch Area - Mean (m)	AREA_MN	3.89	2.72	3.33	2.50	2.82	2.60	2.33	-0.59	Increases
Patch Area - Area-Weighted Mean (m)	AREA_AM	99.51	254.15	173.97	242.14	259.45	257.01	232.92	**0.80**	Decreases
Patch Area - Median (m)	AREA_MD	0.29	0.10	0.19	0.10	0.10	0.10	0.11	**-0.75**	Increases
Patch Area - Range (m)	AREA_RA	181.57	306.11	251.39	298.73	309.39	307.95	293.14	**0.77**	Decreases
Patch Area - Standard Deviation (m)	AREA_SD	19.29	26.15	23.85	24.47	26.92	25.74	23.16	**0.79**	Decreases
Patch Area - Coefficient of Variation (%)	AREA_CV	495.55	961.49	715.59	979.21	953.35	988.26	995.60	**0.72**	Decreases
Radius of Gyration - Mean (m)	GYRATE_MN	56.66	31.50	48.34	31.85	30.53	32.41	33.07	**-0.82**	Increases
Radius of Gyration - Area-Weighted Mean (m)	GYRATE_AM	390.69	802.96	693.36	792.26	815.31	810.67	776.76	**0.70**	Decreases
Radius of Gyration - Median (m)	GYRATE_MD	24.15	15.33	25.29	14.28	14.44	17.01	15.67	**-0.84**	Increases
Radius of Gyration - Range (m)	GYRATE_RA	571.37	943.97	942.85	950.15	952.24	947.80	954.12	0.54	Decreases
Radius of Gyration - Standard Deviation (m)	GYRATE_SD	85.98	84.21	98.57	82.47	85.50	83.40	78.76	-0.44	Increases
Radius of Gyration - Coefficient of Variation (%)	GYRATE_CV	151.74	267.32	203.89	258.91	280.02	257.36	238.20	**0.87**	Decreases
Total Edge Length (m)	TE	31180	21690	30680	27180	25700	27320	31880	-0.66	Increases
Edge Density (m/ha)	ED	84.30	58.64	82.94	73.48	69.48	73.86	86.19	-0.66	Increases
SHAPE										
Landscape Shape Index	LSI	8.03	6.80	7.96	7.51	7.32	7.53	8.12	-0.66	Increases
Perimeter-Area Ratio - Mean	PARA_MN	1524.63	2072.54	1805.41	2130.95	2154.62	2157.41	2045.15	**0.79**	Decreases
Perimeter-Area Ratio - Area-Weighted Mean	PARA_AM	251.43	200.11	248.72	229.80	221.80	230.56	255.21	-0.66	Increases
Perimeter-Area Ratio - Median	PARA_MD	1103.45	2000.00	1333.33	1972.22	1800.00	1973.68	1636.36	**0.72**	Decreases
Perimeter-Area Ratio - Range	PARA_RA	3941.62	3926.11	3923.55	3908.62	3903.23	3909.86	3905.24	-0.68	Increases
Perimeter-Area Ratio - Standard Deviation	PARA_SD	1114.81	1024.83	1195.18	1106.74	1205.06	1057.34	1161.01	0.03	Decreases
Perimeter-Area Ratio - Coefficient of Variation (%)	PARA_CV	73.12	49.45	66.20	51.94	55.93	49.01	56.77	-0.67	Increases
Shape Index - Mean	SHAPE_MN	1.70	1.53	1.73	1.54	1.52	1.63	1.58	**-0.89**	Increases
Shape Index - Area Weighted Mean	SHAPE_AM	2.33	3.13	3.01	3.68	4.02	3.76	3.72	**0.77**	Decreases
Shape Index - Median	SHAPE_MD	1.40	1.33	1.57	1.33	1.33	1.41	1.40	**-0.70**	Increases
Shape Index - Range	SHAPE_RA	3.03	3.36	3.97	2.95	3.38	3.57	3.19	-0.15	Increases
Shape Index - Standard Deviation	SHAPE_SD	0.78	0.60	0.71	0.62	0.68	0.66	0.62	-0.51	Increases
Shape Index - Coefficient of Variation (%)	SHAPE_CV	45.79	38.95	41.19	39.88	44.92	40.67	39.08	-0.02	Increases
Related Circumscribing Circle - Mean	CIRCLE_MN	0.621	0.640	0.659	0.626	0.585	0.639	0.634	-0.69	Increases
Related Circumscribing Circle - Area-Weighted Mean	CIRCLE_AM	0.670	0.720	0.768	0.733	0.719	0.728	0.735	0.03	Decreases
Related Circumscribing Circle - Median	CIRCLE_MD	0.608	0.666	0.682	0.647	0.608	0.682	0.671	-0.26	Increases
Related Circumscribing Circle - Range	CIRCLE_RA	0.554	0.590	0.633	0.541	0.547	0.596	0.577	-0.44	Increases
Related Circumscribing Circle - Standard Deviation	CIRCLE_SD	0.146	0.155	0.168	0.157	0.153	0.161	0.155	-0.13	Increases
Related Circumscribing Circle - Coefficient of Variation (%)	CIRCLE_CV	23.45	24.20	25.43	25.01	26.25	25.18	24.51	0.58	Decreases
Contiguity Index - Mean	CONTIG_MN	0.591	0.443	0.524	0.433	0.433	0.427	0.457	**-0.78**	Increases
Contiguity Index - Area-Weighted Mean	CONTIG_AM	0.929	0.943	0.930	0.935	0.938	0.935	0.928	0.68	Decreases
Contiguity Index - Median	CONTIG_MD	0.683	0.459	0.600	0.477	0.500	0.462	0.533	**-0.72**	Increases
Contiguity Index - Range	CONTIG_RA	0.983	0.978	0.977	0.973	0.972	0.974	0.972	-0.66	Increases
Contiguity Index - Standard Deviation	CONTIG_SD	0.280	0.251	0.288	0.268	0.290	0.259	0.283	-0.11	Increases
Contiguity Index - Coefficient of Variation (%)	CONTIG_CV	47.36	56.60	54.96	61.84	66.92	60.48	61.88	**0.83**	Decreases
ISOLATION AND PROXIMITY										
Proximity Index - Mean	PROX_MN	36.83	129.70	244.48	398.34	109.89	185.61	174.78	-0.03	Increases
Proximity Index - Area-Weighted Mean	PROX_AM	18.42	16.51	39.51	17.88	13.85	15.13	7.02	-0.54	Increases
Proximity Index - Median	PROX_MD	0.75	0.70	2.63	1.73	0.52	1.86	1.15	-0.53	Increases
Proximity Index - Range	PROX_RA	1396.77	7653.00	6285.86	7469.56	7735.00	6159.20	7328.75	0.68	Decreases
Proximity Index - Standard Deviation	PROX_SD	147.94	855.50	1017.86	1465.68	710.49	816.17	927.35	0.21	Decreases
Proximity Index - Coefficient of Variation (%)	PROX_CV	401.73	659.60	416.33	367.95	646.58	439.73	530.58	**0.74**	Decreases
Similarity Index - Mean	SIMI_MN	3782.11	10613.09	7153.98	10929.11	11745.92	10328.41	9048.54	**0.85**	Decreases
Similarity Index - Area-Weighted Mean	SIMI_AM	4887.37	5409.90	8503.36	5365.69	5752.00	3887.73	5591.55	-0.28	Increases
Similarity Index - Median	SIMI_MD	520.53	14174.38	2034.30	11208.85	9400.90	9295.75	2509.48	**0.70**	Decreases
Similarity Index - Range	SIMI_RA	16543.20	27591.30	25223.30	27088.48	29275.38	27721.60	27275.41	**0.74**	Decreases
Similarity Index - Standard Deviation	SIMI_SD	5584.93	9477.19	7926.12	9222.40	11244.30	9702.61	9425.37	**0.89**	Decreases
Similarity Index - Coefficient of Variation (%)	SIMI_CV	147.67	89.30	110.79	84.38	95.73	93.94	104.16	-0.66	Increases
Euclidian Nearest Neighbour Distance - Mean (m)	ENN_MN	133.39	89.18	101.04	71.85	72.79	75.15	76.20	**-0.72**	Increases
Euclidian Nearest Neighbour Distance - Area-Weighted Mean (m)	ENN_AM	76.48	24.63	29.41	25.51	24.18	27.17	27.06	-0.60	Increases
Euclidian Nearest Neighbour Distance - Median (m)	ENN_MD	36.06	36.06	31.62	31.62	31.62	30.00	31.62	-0.22	Increases
Euclidian Nearest Neighbour Distance - Range (m)	ENN_RA	1494.33	1432.45	1652.60	1724.13	640.00	790.25	1561.14	-0.65	Increases
Euclidian Nearest Neighbour Distance - Standard Deviation (m)	ENN_SD	257.25	167.07	225.10	162.79	108.87	135.12	159.08	**-0.90**	Increases
Euclidian Nearest Neighbour Distance - Coefficient of Variation (%)	ENN_CV	192.85	187.34	222.78	226.57	149.57	179.79	208.78	-0.67	Increases
CONTRAST										
Contrast-Weighted Edge Density (m/ha)	CWED	29.03	15.36	23.90	22.30	13.64	20.46	24.42	**-0.89**	Increases
Total Edge Contrast Index (%)	TECI	17.37	10.86	14.41	14.27	8.96	13.06	14.45	**-0.90**	Increases
Edge Contrast Index - Mean (%)	ECON_MN	17.66	15.56	18.80	20.98	13.44	16.44	17.58	-0.64	Increases
Edge Contrast Index - Area-Weighted Mean (%)	ECON_AM	20.61	16.32	18.73	18.20	11.48	19.09	19.09	**-0.89**	Increases
Edge Contrast Index - Median (%)	ECON_MD	15.00	17.42	20.00	20.00	12.02	17.78	17.78	-0.53	Increases
Edge Contrast Index - Range (%)	ECON_RA	57.21	60.00	60.00	55.00	45.00	43.33	50.00	-0.56	Increases
Edge Contrast Index - Standard Deviation (%)	ECON_SD	14.66	10.87	13.80	13.94	9.26	10.79	11.56	**-0.85**	Increases
Edge Contrast Index - Coefficient of Variation (%)	ECON_CV	83.00	69.82	73.40	66.47	68.91	65.61	65.77	-0.59	Increases
CONTAGION AND INTERSPERSION										
Percentage of Like Adjacencies (%)	PLADJ	93.71	95.00	93.78	94.26	94.46	94.24	93.62	0.66	Decreases
Contagion (%)	CONTAG	65.85	79.43	67.53	76.29	78.35	78.96	73.83	**0.84**	Decreases
Aggregation Index (%)	AI	94.75	95.86	94.82	95.17	95.30	95.09	94.56	0.58	Decreases
DIVERSITY										
Shannon's Diversity Index	SHDI	1.212	0.708	1.142	0.805	0.678	0.700	0.877	**-0.88**	Increases
Simpson's Diversity Index	SIDI	0.563	0.305	0.510	0.335	0.289	0.294	0.361	**-0.86**	Increases
Modified Simpson's Diversity Index	MSIDI	0.829	0.364	0.713	0.409	0.342	0.348	0.448	**-0.85**	Increases
Shannon's Evenness Index	SHEI	0.583	0.340	0.549	0.387	0.349	0.337	0.422	**-0.84**	Increases
Simpson's Evenness Index	SIEI	0.644	0.349	0.583	0.383	0.338	0.336	0.413	**-0.85**	Increases
Modified Simpson's Evenness Index	MSIEI	0.399	0.175	0.343	0.196	0.176	0.168	0.215	**-0.82**	Increases

* Correlation Coefficient between metric value and water level for each year, significant correlations (i.e. values ≥ |0.7|) are bolded
^ Total vegetated wetland area (excludes open water)

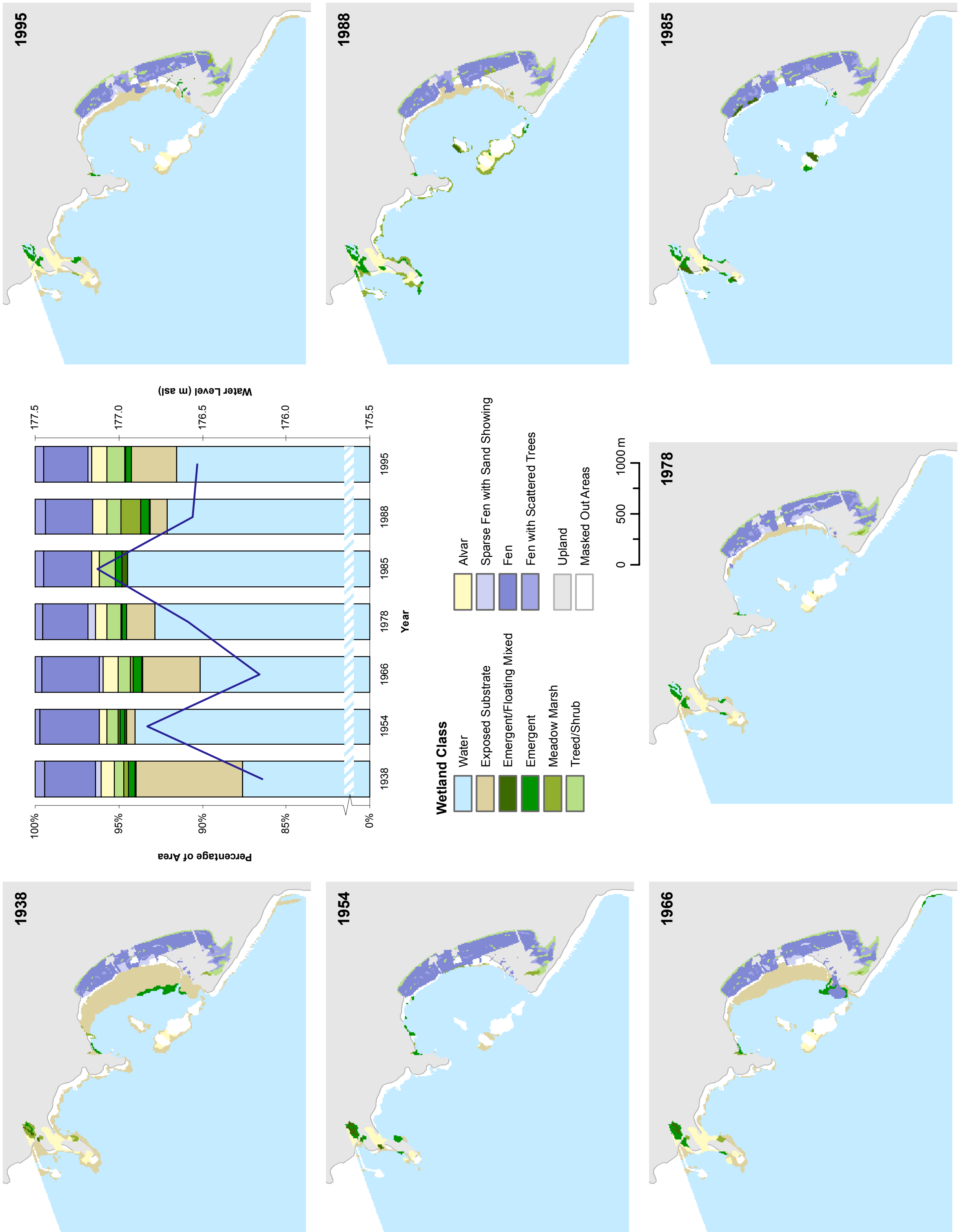

Figure 4.11 Historical distribution of wetland vegetation at Howdenvale, 1938-1995

Table 4.13 Landscape metrics for Howdenvale, 1938-1995

Landscape Metric	Code	Year and Water Level (m asl)							Correl.	Trend w/
		1938	**1954**	**1966**	**1978**	**1985**	**1988**	**1995**	**Coeff.***	**↓ Water**
		176.14	***176.83***	***176.16***	***176.59***	***177.13***	***176.56***	***176.53***		
AREA, DENSITY, EDGE										
Total Area^ (ha)	TA	83.17	40.05	66.04	48.08	37	52.79	56.75	**-0.92**	Increases
Number of Patches (#)	NP	144	113	145	151	104	139	131	**-0.83**	Increases
Patch Density (#/100 ha)	PD	21.45	16.84	21.60	22.50	15.49	20.71	19.52	**-0.83**	Increases
Patch Area - Mean (m)	AREA_MN	4.66	5.94	4.63	4.45	6.45	4.83	5.12	**0.85**	Decreases
Patch Area - Area-Weighted Mean (m)	AREA_AM	516.47	593.16	545.75	576.58	597.29	568.00	561.24	**0.93**	Decreases
Patch Area - Median (m)	AREA_MD	0.06	0.06	0.08	0.06	0.05	0.09	0.10	-0.32	Increases
Patch Area - Range (m)	AREA_RA	587.85	630.66	604.90	621.98	633.04	617.30	613.60	**0.92**	Decreases
Patch Area - Standard Deviation (m)	AREA_SD	48.84	59.06	50.05	50.43	61.75	52.15	53.38	**0.92**	Decreases
Patch Area - Coefficient of Variation (%)	AREA_CV	1047.87	994.29	1081.19	1134.51	956.80	1079.94	1041.81	-0.62	Increases
Radius of Gyration - Mean (m)	GYRATE_MN	33.46	30.43	31.29	29.07	31.07	32.65	35.37	-0.33	Increases
Radius of Gyration - Area-Weighted Mean (m)	GYRATE_AM	887.65	957.65	908.21	936.26	957.30	929.05	921.85	**0.94**	Decreases
Radius of Gyration - Median (m)	GYRATE_MD	10.75	10.42	12.99	11.44	11.12	13.96	14.96	-0.25	Increases
Radius of Gyration - Range (m)	GYRATE_RA	980.07	1002.81	989.49	998.12	1004.02	997.05	993.62	**0.91**	Decreases
Radius of Gyration - Standard Deviation (m)	GYRATE_SD	90.90	98.51	86.55	85.03	100.82	87.90	91.45	**0.73**	Decreases
Radius of Gyration - Coefficient of Variation (%)	GYRATE_CV	271.68	323.76	276.59	292.49	324.49	269.26	258.59	**0.77**	Decreases
Total Edge Length (m)	TE	23870	12330	20750	19920	12260	21430	22720	**-0.84**	Increases
Edge Density (m/ha)	ED	35.56	18.37	30.91	29.68	18.27	31.93	33.85	**-0.84**	Increases
SHAPE										
Landscape Shape Index	LSI	5.28	4.16	4.98	4.90	4.16	5.04	5.17	**-0.84**	Increases
Perimeter-Area Ratio - Mean	PARA_MN	2375.66	2369.39	2213.78	2389.98	2340.62	2173.06	2030.89	0.20	Decreases
Perimeter-Area Ratio - Area-Weighted Mean	PARA_AM	117.16	82.78	107.86	105.39	82.57	109.89	113.73	**-0.84**	Increases
Perimeter-Area Ratio - Median	PARA_MD	2400.00	2222.22	2000.00	2400.00	2400.00	2000.00	1857.14	0.28	Decreases
Perimeter-Area Ratio - Range	PARA_RA	3963.70	3966.64	3965.32	3965.82	3965.66	3965.30	3965.29	0.67	Decreases
Perimeter-Area Ratio - Standard Deviation	PARA_SD	1126.32	1040.71	1056.81	1137.33	1140.06	1024.36	1001.02	0.14	Decreases
Perimeter-Area Ratio - Coefficient of Variation (%)	PARA_CV	47.41	43.92	47.74	47.59	48.71	47.14	49.29	-0.11	Increases
Shape Index - Mean	SHAPE_MN	1.55	1.48	1.51	1.50	1.43	1.59	1.58	-0.63	Increases
Shape Index - Area Weighted Mean	SHAPE_AM	2.27	2.14	2.16	2.17	2.18	2.19	2.21	-0.45	Increases
Shape Index - Median	SHAPE_MD	1.25	1.25	1.30	1.29	1.17	1.40	1.31	-0.46	Increases
Shape Index - Range	SHAPE_RA	3.54	2.92	3.40	3.14	3.28	3.58	3.39	-0.54	Increases
Shape Index - Standard Deviation	SHAPE_SD	0.72	0.60	0.61	0.63	0.63	0.65	0.72	-0.37	Increases
Shape Index - Coefficient of Variation (%)	SHAPE_CV	46.49	40.34	40.14	42.25	44.37	41.16	45.60	-0.07	Increases
Related Circumscribing Circle - Mean	CIRCLE_MN	0.617	0.623	0.625	0.619	0.576	0.646	0.628	-0.56	Increases
Related Circumscribing Circle - Area-Weighted Mean	CIRCLE_AM	0.592	0.548	0.571	0.554	0.541	0.560	0.564	**-0.91**	Increases
Related Circumscribing Circle - Median	CIRCLE_MD	0.641	0.644	0.632	0.646	0.532	0.693	0.645	-0.56	Increases
Related Circumscribing Circle - Range	CIRCLE_RA	0.606	0.602	0.650	0.593	0.616	0.595	0.633	-0.33	Increases
Related Circumscribing Circle - Standard Deviation	CIRCLE_SD	0.175	0.166	0.166	0.168	0.163	0.168	0.168	-0.68	Increases
Related Circumscribing Circle - Coefficient of Variation (%)	CIRCLE_CV	28.32	26.63	26.48	27.08	28.35	25.97	26.76	0.17	Decreases
Contiguity Index - Mean	CONTIG_MN	0.377	0.373	0.409	0.372	0.378	0.418	0.451	-0.25	Increases
Contiguity Index - Area-Weighted Mean	CONTIG_AM	0.967	0.977	0.970	0.970	0.977	0.969	0.968	**0.84**	Decreases
Contiguity Index - Median	CONTIG_MD	0.333	0.367	0.417	0.354	0.369	0.433	0.478	-0.09	Increases
Contiguity Index - Range	CONTIG_RA	0.989	0.990	0.990	0.990	0.990	0.990	0.990	0.67	Decreases
Contiguity Index - Standard Deviation	CONTIG_SD	0.267	0.245	0.258	0.269	0.279	0.249	0.248	0.18	Decreases
Contiguity Index - Coefficient of Variation (%)	CONTIG_CV	70.99	65.70	63.06	72.17	73.67	59.38	55.05	0.26	Decreases
ISOLATION AND PROXIMITY										
Proximity Index - Mean	PROX_MN	124.68	459.65	222.49	369.87	494.32	384.38	366.10	**0.94**	Decreases
Proximity Index - Area-Weighted Mean	PROX_AM	8.89	25.65	14.09	21.12	14.87	17.34	17.61	0.52	Decreases
Proximity Index - Median	PROX_MD	0.91	0.55	1.45	0.91	0.65	1.38	1.65	-0.55	Increases
Proximity Index - Range	PROX_RA	11757.20	15768.51	15122.75	15550.64	15826.25	15432.75	15340.25	0.66	Decreases
Proximity Index - Standard Deviation	PROX_SD	1021.38	2552.03	1762.96	2251.30	2549.47	2295.02	2292.81	**0.84**	Decreases
Proximity Index - Coefficient of Variation (%)	PROX_CV	819.19	555.21	792.39	608.68	515.76	597.08	626.28	**-0.95**	Increases
Similarity Index - Mean	SIMI_MN	13601.44	15270.49	16151.37	16774.33	16920.69	18986.70	16974.03	0.36	Decreases
Similarity Index - Area-Weighted Mean	SIMI_AM	5287.30	3022.87	3539.44	1776.93	2625.55	2340.21	2632.17	-0.59	Increases
Similarity Index - Median	SIMI_MD	1551.42	1741.27	835.55	2307.78	1622.00	9275.08	2018.95	0.06	Decreases
Similarity Index - Range	SIMI_RA	48671.70	50766.32	49446.99	49826.40	57098.07	55574.61	49417.70	**0.71**	Decreases
Similarity Index - Standard Deviation	SIMI_SD	16727.50	18317.62	18038.75	18279.29	20777.72	18422.42	18225.91	**0.87**	Decreases
Similarity Index - Coefficient of Variation (%)	SIMI_CV	122.98	119.95	111.69	108.97	122.79	97.03	107.38	0.19	Decreases
Euclidian Nearest Neighbour Distance - Mean (m)	ENN_MN	61.97	64.78	53.64	62.78	105.16	60.24	71.18	**0.79**	Decreases
Euclidian Nearest Neighbour Distance - Area-Weighted Mean (m)	ENN_AM	24.04	21.19	21.84	21.84	21.65	22.00	21.57	-0.62	Increases
Euclidian Nearest Neighbour Distance - Median (m)	ENN_MD	30.00	31.62	29.14	22.36	36.06	28.28	22.36	0.41	Decreases
Euclidian Nearest Neighbour Distance - Range (m)	ENN_RA	1111.59	1189.34	903.15	910.59	1605.61	1353.83	2136.41	0.37	Decreases
Euclidian Nearest Neighbour Distance - Standard Deviation (m)	ENN_SD	136.75	131.64	102.51	121.75	229.59	138.13	219.76	0.59	Decreases
Euclidian Nearest Neighbour Distance - Coefficient of Variation (%)	ENN_CV	220.67	203.21	191.09	193.94	218.31	229.32	308.76	0.03	Decreases
CONTRAST										
Contrast-Weighted Edge Density (m/ha)	CWED	12.34	5.50	10.48	10.43	5.16	10.78	11.87	**-0.84**	Increases
Total Edge Contrast Index (%)	TECI	15.13	8.54	13.62	13.78	8.02	13.83	14.86	**-0.83**	Increases
Edge Contrast Index - Mean (%)	ECON_MN	21.48	21.24	22.77	23.61	21.89	22.24	22.06	-0.19	Increases
Edge Contrast Index - Area-Weighted Mean (%)	ECON_AM	21.11	7.21	17.64	14.52	4.23	17.79	19.18	**-0.89**	Increases
Edge Contrast Index - Median (%)	ECON_MD	20.00	16.00	20.00	20.00	20.00	20.00	20.00	-0.34	Increases
Edge Contrast Index - Range (%)	ECON_RA	60.00	60.00	60.00	60.00	60.00	60.00	60.00	0.00	-
Edge Contrast Index - Standard Deviation (%)	ECON_SD	14.78	16.69	16.05	14.87	17.52	14.72	14.50	0.63	Decreases
Edge Contrast Index - Coefficient of Variation (%)	ECON_CV	68.80	78.58	70.51	62.99	80.04	66.21	65.75	0.60	Decreases
CONTAGION AND INTERSPERSION										
Percentage of Like Adjacencies (%)	PLADJ	97.07	97.93	97.30	97.37	97.94	97.25	97.16	**0.84**	Decreases
Contagion (%)	CONTAG	86.09	91.98	87.93	89.83	92.36	88.77	88.96	**0.94**	Decreases
Aggregation Index (%)	AI	97.70	98.48	97.92	97.94	98.47	97.84	97.74	**0.82**	Decreases
DIVERSITY										
Shannon's Diversity Index	SHDI	0.556	0.310	0.480	0.380	0.294	0.417	0.431	**-0.95**	Increases
Simpson's Diversity Index	SIDI	0.227	0.114	0.185	0.137	0.106	0.150	0.160	**-0.93**	Increases
Modified Simpson's Diversity Index	MSIDI	0.258	0.122	0.204	0.147	0.112	0.162	0.175	**-0.92**	Increases
Shannon's Evenness Index	SHEI	0.241	0.141	0.209	0.173	0.134	0.190	0.187	**-0.95**	Increases
Simpson's Evenness Index	SIEI	0.253	0.129	0.205	0.154	0.120	0.169	0.178	**-0.93**	Increases
Modified Simpson's Evenness Index	MSIEI	0.112	0.055	0.089	0.067	0.051	0.074	0.076	**-0.92**	Increases

* Correlation Coefficient between metric value and water level for each year, significant correlations (i.e. values ≥ |0.7|) are bolded
^ Total vegetated wetland area (excludes open water)

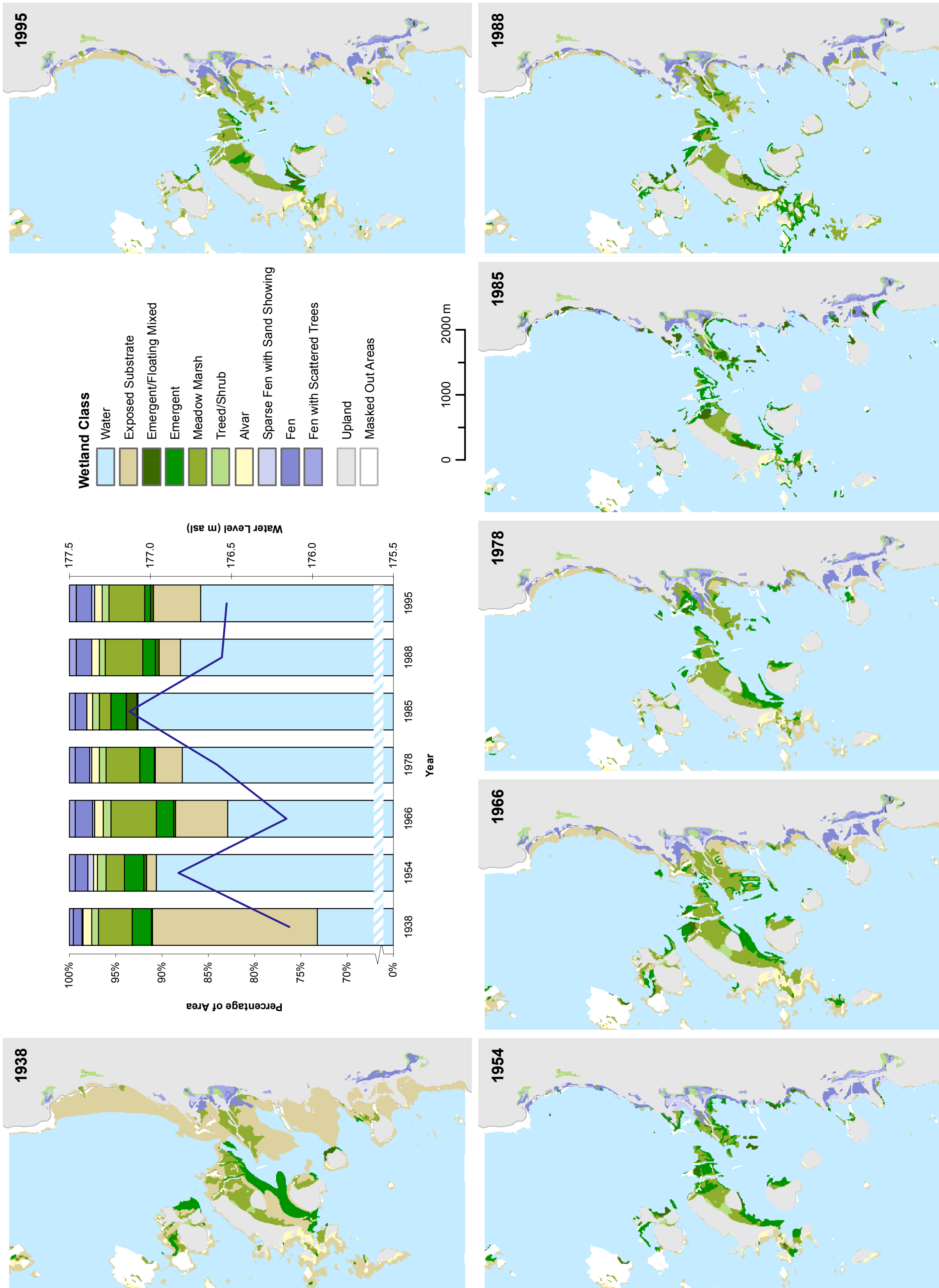

Figure 4.12 Historical distribution of wetland vegetation at Oliphant, 1938-1995

Table 4.14 Landscape metrics for Oliphant, 1938-1995

Landscape Metric	Code	Year and Water Level (m asl) 1938 *176.14*	1954 *176.83*	1966 *176.16*	1978 *176.59*	1985 *177.13*	1988 *176.56*	1995 *176.53*	Correl. Coeff.*	Trend w/ ↓ Water
AREA, DENSITY, EDGE										
Total Area^ (ha)	TA	507.51	178.21	324.87	231.89	140.47	227.31	268.93	**-0.87**	Increases
Number of Patches (#)	NP	331	494	452	497	438	526	490	0.40	Decreases
Patch Density (#/100 ha)	PD	17.44	26.02	23.81	26.18	23.07	27.71	25.81	0.40	Decreases
Patch Area - Mean (m)	AREA_MN	5.74	3.84	4.20	3.82	4.33	3.61	3.87	-0.44	Increases
Patch Area - Area-Weighted Mean (m)	AREA_AM	526.19	1558.72	1304.45	1462.43	1621.78	1465.93	1395.73	**0.74**	Decreases
Patch Area - Median (m)	AREA_MD	0.13	0.09	0.13	0.12	0.10	0.12	0.12	**-0.87**	Increases
Patch Area - Range (m)	AREA_RA	777.17	1720.05	1572.99	1665.93	1754.54	1667.99	1627.31	0.67	Decreases
Patch Area - Standard Deviation (m)	AREA_SD	54.63	77.30	73.90	74.64	83.73	72.65	73.43	**0.78**	Decreases
Patch Area - Coefficient of Variation (%)	AREA_CV	952.62	2011.51	1759.52	1954.16	1931.80	2012.91	1895.43	0.64	Decreases
Radius of Gyration - Mean (m)	GYRATE_MN	46.64	28.17	38.75	32.01	27.99	31.66	34.58	**-0.89**	Increases
Radius of Gyration - Area-Weighted Mean (m)	GYRATE_AM	931.22	1927.19	1824.84	1880.24	1950.21	1875.94	1846.84	0.63	Decreases
Radius of Gyration - Median (m)	GYRATE_MD	17.54	14.24	17.72	16.54	15.39	16.42	16.76	**-0.85**	Increases
Radius of Gyration - Range (m)	GYRATE_RA	1213.22	2113.71	2167.35	2123.30	2099.64	2117.93	2126.35	0.48	Decreases
Radius of Gyration - Standard Deviation (m)	GYRATE_SD	110.63	98.26	110.83	100.21	102.11	96.99	103.69	**-0.70**	Increases
Radius of Gyration - Coefficient of Variation (%)	GYRATE_CV	237.18	348.81	286.05	313.08	364.86	306.34	299.87	**0.94**	Decreases
Total Edge Length (m)	TE	103010	75050	113300	94100	68130	103920	106730	**-0.89**	Increases
Edge Density (m/ha)	ED	54.26	39.53	59.68	49.57	35.89	54.74	56.22	**-0.89**	Increases
SHAPE										
Landscape Shape Index	LSI	11.39	9.79	11.98	10.88	9.39	11.45	11.61	**-0.89**	Increases
Perimeter-Area Ratio - Mean	PARA_MN	1934.96	2228.42	1971.32	2085.33	2127.63	2100.63	2040.70	**0.84**	Decreases
Perimeter-Area Ratio - Area-Weighted Mean	PARA_AM	158.94	129.48	169.78	149.55	122.19	159.90	162.86	**-0.89**	Increases
Perimeter-Area Ratio - Median	PARA_MD	1750.00	2000.00	1623.56	1846.15	1958.33	1804.76	1777.78	**0.89**	Decreases
Perimeter-Area Ratio - Range	PARA_RA	3956.23	3954.75	3945.72	3949.35	3950.63	3947.41	3948.04	0.09	Decreases
Perimeter-Area Ratio - Standard Deviation	PARA_SD	1135.47	1165.95	1136.19	1112.47	1103.79	1116.37	1073.67	-0.14	Increases
Perimeter-Area Ratio - Coefficient of Variation (%)	PARA_CV	58.68	52.32	57.64	53.35	51.88	53.14	52.61	**-0.88**	Increases
Shape Index - Mean	SHAPE_MN	1.66	1.49	1.64	1.56	1.50	1.59	1.61	**-0.94**	Increases
Shape Index - Area Weighted Mean	SHAPE_AM	3.43	4.47	4.93	4.84	4.95	5.03	4.90	0.43	Decreases
Shape Index - Median	SHAPE_MD	1.45	1.33	1.42	1.38	1.33	1.33	1.38	**-0.87**	Increases
Shape Index - Range	SHAPE_RA	4.43	4.89	4.62	4.45	4.17	4.93	4.24	-0.16	Increases
Shape Index - Standard Deviation	SHAPE_SD	0.75	0.59	0.73	0.64	0.57	0.71	0.71	**-0.93**	Increases
Shape Index - Coefficient of Variation (%)	SHAPE_CV	44.95	39.48	44.69	41.12	38.15	44.51	44.23	**-0.89**	Increases
Related Circumscribing Circle - Mean	CIRCLE_MN	0.644	0.615	0.640	0.635	0.624	0.631	0.645	**-0.77**	Increases
Related Circumscribing Circle - Area-Weighted Mean	CIRCLE_AM	0.654	0.637	0.674	0.652	0.629	0.650	0.664	**-0.85**	Increases
Related Circumscribing Circle - Median	CIRCLE_MD	0.668	0.638	0.682	0.643	0.644	0.655	0.654	**-0.85**	Increases
Related Circumscribing Circle - Range	CIRCLE_RA	0.649	0.637	0.657	0.628	0.571	0.629	0.660	**-0.81**	Increases
Related Circumscribing Circle - Standard Deviation	CIRCLE_SD	0.165	0.171	0.174	0.169	0.162	0.168	0.166	-0.44	Increases
Related Circumscribing Circle - Coefficient of Variation (%)	CIRCLE_CV	25.59	27.70	27.23	26.53	26.01	26.65	25.73	0.09	Decreases
Contiguity Index - Mean	CONTIG_MN	0.485	0.411	0.475	0.446	0.434	0.441	0.455	**-0.85**	Increases
Contiguity Index - Area-Weighted Mean	CONTIG_AM	0.955	0.964	0.952	0.958	0.966	0.955	0.954	**0.89**	Decreases
Contiguity Index - Median	CONTIG_MD	0.521	0.457	0.542	0.480	0.478	0.500	0.500	**-0.84**	Increases
Contiguity Index - Range	CONTIG_RA	0.987	0.987	0.984	0.986	0.986	0.985	0.985	0.13	Decreases
Contiguity Index - Standard Deviation	CONTIG_SD	0.279	0.280	0.277	0.270	0.266	0.270	0.262	-0.44	Increases
Contiguity Index - Coefficient of Variation (%)	CONTIG_CV	57.60	68.07	58.44	60.54	61.21	61.25	57.63	0.60	Decreases
ISOLATION AND PROXIMITY										
Proximity Index - Mean	PROX_MN	208.13	51.20	106.46	207.90	567.71	465.36	425.42	0.47	Decreases
Proximity Index - Area-Weighted Mean	PROX_AM	2827.59	3.03	6.86	15.43	78.02	84.71	34.16	-0.52	Increases
Proximity Index - Median	PROX_MD	1.80	0.44	2.18	0.95	0.63	0.86	1.53	**-0.88**	Increases
Proximity Index - Range	PROX_RA	19429.50	21500.75	39325.00	33318.80	43863.75	41700.00	40684.20	0.29	Decreases
Proximity Index - Standard Deviation	PROX_SD	1209.10	966.28	1847.34	2348.64	4371.70	4018.50	3333.47	0.50	Decreases
Proximity Index - Coefficient of Variation (%)	PROX_CV	580.94	1887.21	1735.26	1129.71	770.05	863.52	783.57	0.00	Increases
Similarity Index - Mean	SIMI_MN	18429.34	61685.50	51744.15	58350.64	73454.88	58660.63	52487.37	**0.81**	Decreases
Similarity Index - Area-Weighted Mean	SIMI_AM	25639.50	13662.96	23682.67	17951.54	11643.74	17788.95	19767.78	**-0.98**	Increases
Similarity Index - Median	SIMI_MD	4210.29	86026.63	47357.94	83417.42	87813.59	83466.14	48985.59	**0.80**	Decreases
Similarity Index - Range	SIMI_RA	86944.29	155411.92	143061.48	150360.91	158249.62	150988.50	148343.28	0.69	Decreases
Similarity Index - Standard Deviation	SIMI_SD	20942.71	53877.56	49402.30	50473.80	60091.98	51779.50	49194.78	**0.74**	Decreases
Similarity Index - Coefficient of Variation (%)	SIMI_CV	113.64	87.34	95.47	86.50	81.81	88.27	93.73	**-0.81**	Increases
Euclidian Nearest Neighbour Distance - Mean (m)	ENN_MN	109.22	76.92	71.23	84.53	102.55	83.38	81.78	0.11	Decreases
Euclidian Nearest Neighbour Distance - Area-Weighted Mean (m)	ENN_AM	37.21	32.88	25.14	28.28	25.25	24.34	26.83	-0.29	Increases
Euclidian Nearest Neighbour Distance - Median (m)	ENN_MD	30.00	36.06	30.00	31.62	36.06	30.00	30.00	**0.85**	Decreases
Euclidian Nearest Neighbour Distance - Range (m)	ENN_RA	4780.38	2195.78	1606.35	2700.90	2060.87	2492.87	1598.05	-0.39	Increases
Euclidian Nearest Neighbour Distance - Standard Deviation (m)	ENN_SD	353.99	150.77	131.86	198.74	214.70	204.27	166.80	-0.27	Increases
Euclidian Nearest Neighbour Distance - Coefficient of Variation (%)	ENN_CV	324.11	196.00	185.13	235.10	209.36	245.00	203.96	-0.42	Increases
CONTRAST										
Contrast-Weighted Edge Density (m/ha)	CWED	21.63	11.18	20.82	16.55	9.43	17.83	20.16	**-0.94**	Increases
Total Edge Contrast Index (%)	TECI	20.66	12.43	18.91	16.55	10.93	16.96	18.90	**-0.94**	Increases
Edge Contrast Index - Mean (%)	ECON_MN	23.56	18.39	21.84	21.27	17.54	20.66	22.74	**-0.89**	Increases
Edge Contrast Index - Area-Weighted Mean (%)	ECON_AM	28.91	16.04	25.99	22.78	12.73	23.16	26.33	**-0.94**	Increases
Edge Contrast Index - Median (%)	ECON_MD	24.00	19.33	20.00	20.00	16.28	20.00	20.00	**-0.84**	Increases
Edge Contrast Index - Range (%)	ECON_RA	60.00	60.00	60.00	60.00	60.00	60.00	60.00	0.00	-
Edge Contrast Index - Standard Deviation (%)	ECON_SD	13.05	10.88	12.45	13.16	11.55	13.04	13.54	-0.59	Increases
Edge Contrast Index - Coefficient of Variation (%)	ECON_CV	55.38	59.17	56.99	61.84	65.86	63.11	59.56	**0.83**	Decreases
CONTAGION AND INTERSPERSION										
Percentage of Like Adjacencies (%)	PLADJ	96.03	96.76	95.76	96.26	96.95	96.00	95.93	**0.89**	Decreases
Contagion (%)	CONTAG	77.50	86.85	80.30	84.36	88.55	84.18	82.76	**0.97**	Decreases
Aggregation Index (%)	AI	96.45	97.14	96.17	96.65	97.29	96.38	96.32	**0.87**	Decreases
DIVERSITY										
Shannon's Diversity Index	SHDI	0.896	0.503	0.758	0.595	0.407	0.589	0.654	**-0.96**	Increases
Simpson's Diversity Index	SIDI	0.430	0.178	0.307	0.227	0.142	0.222	0.259	**-0.91**	Increases
Modified Simpson's Diversity Index	MSIDI	0.561	0.196	0.366	0.257	0.153	0.252	0.299	**-0.88**	Increases
Shannon's Evenness Index	SHEI	0.389	0.218	0.329	0.259	0.185	0.256	0.284	**-0.95**	Increases
Simpson's Evenness Index	SIEI	0.477	0.197	0.341	0.252	0.159	0.247	0.287	**-0.90**	Increases
Modified Simpson's Evenness Index	MSIEI	0.244	0.085	0.159	0.112	0.070	0.109	0.130	**-0.87**	Increases

* Correlation Coefficient between metric value and water level for each year, significant correlations (i.e. values ≥ |0.7|) are bolded
^ Total vegetated wetland area (excludes open water)

Table 4.15 Wetland vegetation response on Lake Huron

Declining Water Levels	Rising Water Levels	Indicated By
LANDSCAPE ANALYSIS		
• Vegetated wetland area increased	• Vegetated wetland area decreased	• TA
• Wetland more fragmented (Howdenvale only) • Patches larger in size • Patches more elongated and less compact (except Howdenvale)	• Wetland less fragmented (Howdenvale) • Patches smaller in size • Patches more compact in shape (except Howdenvale)	• NP, PD • AREA_ • GYRATE_
• Wetland more heterogeneous, communities fragmented	• Wetland more homogeneous, communities less fragmented	• TE, ED
• Patches more disaggregated, irregular in shape • Patches more elongated and linear • Patches larger and more contiguous; patches less fragmented in distribution	• Patches more aggregated, regular in shape • Patches more circular and compact • Patches smaller and more fragmented in distribution	• LSI, SHAPE_ • CIRCLE_, PARA_ • CONTIG_
• Patches isolated and fragmented; neighbourhood occupied by fewer patches of the same class	• Neighbourhood increasingly occupied by patches of the same class	• PROX_, ENN_
• More edge contrast	• Less edge contrast	• CWED, TECI, ECON_
• Patches more disaggregated, the proportion of like adjacencies decreased (i.e. patches of similar wetland classes were less equally adjacent to each other) • Patch types more interspersed	• Patches more aggregated, the proportion of like adjacencies increased (i.e. patches of similar wetland classes were more equally adjacent to each other) • Patch types less interspersed	• PLADJ, AI • CONTAG
• Wetlands more diverse, proportion and distribution of patch type area more even across landscape	• Wetlands less diverse, distribution of area among different patch types more uneven	• SHDI, SIDI, MSIDI, SHEI, SIEI, MSIEI
CLASS ANALYSIS		
• Open water, emergent/floating mixed decreased • Exposed substrate, meadow marsh, alvar, sparse fen with sand, and fen with scattered trees increased in area • Fewer patches of emergent/floating mixed and emergent vegetation	• Open water, emergent/floating mixed increased • Exposed substrate, meadow marsh, alvar, sparse fen with sand showing, and fen with scattered trees decreased in area • More patches of emergent/floating mixed and emergent vegetation	• CA • NP, PD
• Patches of exposed substrate less fragmented	• Patches of exposed substrate more fragmented	• TE, ED
• Patches of exposed substrate larger and contiguous in size; more elongated, less compact, and irregular in shape but more complex and convoluted in shape • Patches of emergent, meadow marsh, treed/shrub larger and more contiguous, simpler, less compact, and more elongated in shape. • Fewer but larger and contiguous patches of fen and fen with scattered trees; (Oliphant only) patches are irregular, elongated, and linear shaped • Patches of sparse fen with sand showing more complex and convoluted in shape	• Patches of exposed substrate smaller, more fragmented in distribution; more compact, regular, and simple in shape • Patches of emergent, meadow marsh, treed/shrub smaller and more fragmented, more complex, compact in shape • More, smaller patches of fen and fen with scattered trees; (Oliphant only) patches are regular, and more circular in shape • Patches of sparse fen with sand showing simpler in shape	• LSI, AREA_, GYRATE_, CIRCLE_, SHAPE_, PARA_, CONTIG_
• Patches of open water, exposed substrate, emergent, and sparse fen with sand showing had more contrast with neighbouring class edge • Patches of fen with scattered trees had less contrast with neighbouring class edge	• Patches of open water, exposed substrate, emergent, and sparse fen with sand showing had less contrast with neighbouring class edge • Patches of fen with scattered trees had more contrast with neighbouring class edge	• TECI, ECON_, CWED
• Patches of exposed substrate located closer to each other in distribution • Patches of emergent, meadow marsh, alvar, and fen located further from each other • Patches of fen with scattered trees located among areas of less similar patch types	• Patches of exposed substrate located further from each other in distribution • Patches of emergent, meadow marsh, alvar, and fen located closer to each other • Patches of fen with scattered trees located in areas with more similar patch types	• PROX_, ENN_ • SIMI_
• Patches of exposed substrate, emergent/floating mixed, emergent, and all three fen communities more aggregated in distribution across the wetland and proportion of like adjacencies increases	• Patches of exposed substrate, emergent/floating mixed, emergent, and all three fen communities more dispersed in distribution across the wetland	• PLADJ, AI
SPATIAL ANALYSIS		
• Lakeward migration of wetland communities • Succession to drier communities • Alvar and substrate exposed along shore	• Landward migration of wetland communities • Succession to wetter communities • Alvar and substrate flooded by high lake water	• Visual Observation

4.3 VEGETATION MODELLING

A wetland vegetation model was developed to simulate vegetation response to historic water level fluctuations in marshes on Lakes Erie and Ontario. Lake Huron wetlands were excluded as they were particularly difficult to model (Sabila 2005) because of fen and alvar communities that were less directly influenced by lake levels and more influenced by regional groundwater supplies. The following section describes the development, validation, and output of the model.

4.3.1 Model Development

A rule-based model was developed to simulate the generalized wetland vegetation classes observed during air photo interpretation (e.g. open water, exposed substrate, emergent/floating mixed, emergent, meadow, treed/shrub). Model rules were based on water depth, duration of hydrologic condition (i.e. number of years flooded or dewatered), and tolerance ranges of the different wetland vegetation communities to water level conditions. As such, detailed topographic models were required for each wetland site. The following section describes in detail, the compilation of elevation and bathymetry data used to create topographic models of the wetlands, the creation of the input grids for the model, and the decision rules that govern the model.

4.3.1.1 Elevation Models

For Lake Ontario, two sources of data were used to create elevation models of the wetland sites. OMNR provided land spot height elevations contained in the Natural Resources and Values Information System's (NRVIS) digital terrain model (DTM). To complement land information, hydrographic survey points of the lake, collected by Ocean Surveys, Inc. (OSI) for the IJC LOSLR Study were provided by CWS.

For Lake Erie, the topographic models were compiled from a number of different data sources. OMNR spot and five-metre contour data of the land were obtained for Long Point, Turkey Point, and Rondeau. For Dunnville, half-metre base mapping contours from Ontario's Flood Damage Reduction Program (FDRP) were obtained from the Grand River Conservation Authority (GRCA) and used instead of the OMNR data because of higher resolution. FDRP points, rather than contours, were available for Rondeau from the Lower Thames Valley Conservation Authority (LTVCA) and added data to the nearshore.

Bathymetry field sheets were obtained from Fisheries and Oceans Canada (DFO) for all four Lake Erie sites. The mapped bathymetry spot values were manually digitized into a GIS. To calculate the lake floor elevation, spot values were subtracted from the corrected lake level indicated by vertical datum on the field sheets. The shoreline was also digitized and assigned an elevation value equal to lake level (as indicated by the vertical datum). Supplementary bathymetry information, which added coverage to nearshore areas, was developed from water level observations collected at various stations at Long Point and Rondeau and available from DFO's Marine Environmental Data Service (MEDS); water level measurements at these stations were generated into point coverages using UTM coordinates for the stations. Elevation surveys completed by the CWS were also incorporated to add detail to the Big Creek Marshes and the dyked National Wildlife Area (NWA) at Long Point (see Section 8.2.4.2). In addition, National Oceanic and Atmospheric Administration's (NOAA) one-metre bathymetry contour data added supplementary information for the shoreline and Outer Bay at Long Point, which was not covered by the bathymetry field sheets. The elevation data collected for all of the wetland sites is summarized in Table 4.16.

Preliminary interpolation of the bathymetry and land elevation data identified several problems with some wetland sites. Lack of bathymetry and elevation data in wetland vegetated areas severely limited model construction for Turkey Point (and Lake Huron fen sites). Interpolated elevation surfaces were interrupted with triangle-shaped areas of erroneous elevation in areas with few or missing data points. Smaller areas at Long Point, Rondeau, and Dunnville were also affected. Therefore, additional points were strategically placed in Lake Erie wetlands to supplement existing elevation information. Bathymetry field sheets, historical wetland data, and existing OMNR spot data were used to estimate the location and value of these additional points. Points added in known water areas were assigned spot heights equivalent to lake level, while points added in known marsh habitat were assigned elevations that were half a metre higher than lake level; this value was consistent with OMNR spot values that occurred in similar locations within the study sites.

Table 4.16 GIS elevation data layers

Description	Date of Capture	Scale	Horizontal and Vertical Reference	Source
NRVIS OBM-DTM	1993 to 1996	1:10000	NAD83(CGVD28)	OMNR
OBM Spot and 5 m Contours	1977 to 1996	1:10000	NAD83(CGVD28)	OMNR
FDRP Base Mapping				
Dunnville (0.5 m Contours)	1994	1:2000	NAD83(IGLD85)	GRCA
Rondeau (Spots)	1986	1:2000	Mean Water Level (year of capture)	LTVCA
OSI Hydrographic Survey Points - 10 soundings every 0.03 to 0.1 m in lines 0.5 m apart - survey lines 100 m apart	Summer 2003	n/a	NAD83(IGLD85)	U.S. Army Corps of Engineers
Bathymetry Field Sheets (Spots)	1965 to 2001	1:1000 to 1:30000	Varies; NAD27(IGLD55), NAD83(IGLD85)	DFO
Bathymetry Contours of Lake Erie (1 m intervals)	1903 to 1998	1:2500 to 1:100000	WGS84(IGLD55)	NOAA
Water Level Data (MEDS)	2004	Daily, Weekly, Monthly	NAD83(IGLD85)	DFO
Promark Elevation Surveys - 200 randomly selected points on 20 m grid	April to June 2004	n/a	NAD83(IGLD85)	CWS

All the elevation data layers were projected into UTM, NAD83 and referenced to International Great Lakes Datum of 1985 (IGLD85). A triangular irregular network (TIN) was generated for each wetland site using all available datasets (Figure 4.13A). A TIN surface was represented by a series of adjacent but non-overlapping triangles computed for irregular spaced points (ESRI 2003). The TIN interpolation method was selected because it produced the best results compared to other interpolation methods tested (e.g. TOPOGRID, kriging, IDW). To eliminate edge effects, all bathymetry and land elevation data within a one-kilometre (km) buffer of the wetland sites were used as input in creation of the TIN. The TIN was then converted to a lattice with a 10-m cell size resolution compatible with the wetland vegetation data. The same mask applied to the historical wetland vegetation data (Section 4.1.2) was also applied to the elevation lattice (Figure 4.13B).

4.3.1.2 Water Depth and Duration Grids

Elevation models were used to create two input grids for the vegetation model: a water depth grid and duration of hydrologic condition grid. For each year of existing historical wetland air photo data, a water depth grid was created by subtracting the historic mean water level for that observation year from the interpolated elevation values (Figure 4.13C). Cell values in the water depth grid represented the substrate height in metres above or below the historic mean water level for a particular year. Similarly, water depth grids were created for one year prior, to understand and characterize the previous year's conditions, and then annually referring back another 40 years prior, to derive duration of hydrologic condition grids since antecedent water level conditions influence wetland vegetation community development.

An ARC Macro Language (AML) program was developed in *ARC/INFO* to determine duration of hydrologic conditions of each cell within the water depth grids. The AML queried each cell in the historical water depth grid on a year-by-year basis to determine how long that particular cell was flooded (i.e. below lake level) or dewatered (i.e. above lake level) and then classified the cells into one of 14 duration categories. Duration categories represent cells that were flooded or dewatered for: the current year; 1 to 5 years; 6 to 10 years; 11 to 20 years; 21 to 30 years; 31 to 40 years; or over 40 years. Duration grids were produced for each year of historical wetland data (Figure 4.13D), as well as one year prior to the historical dates. Both the current (i.e. year of historical wetland data) and previous year's duration and depth grids were used as input into the wetland model.

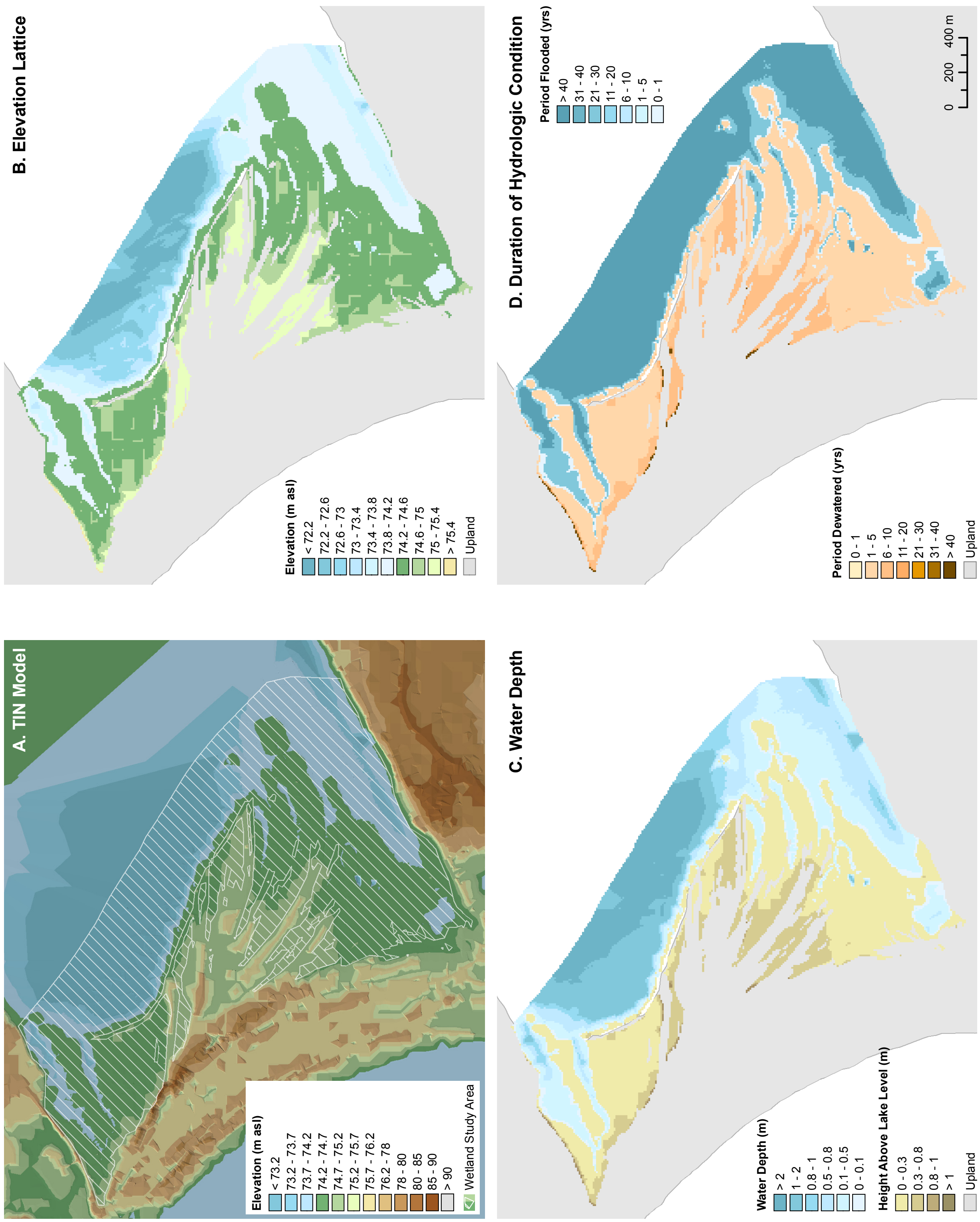

Figure 4.13 Example of the TIN, elevation lattice and input grids created for Presqu'ile, 1965

4.3.1.3 Decision Rules for Model

The rule-based vegetation model was developed using AML programming. A series of if-then statements were applied to predict wetland vegetation class based on water depth, duration of hydrologic condition, and tolerance ranges of the different wetland communities to water level conditions. Vegetation community hydrologic tolerance ranges were synthesized based upon published literature and field survey data (see Section 3.1; Wilcox *et al.* 2005). The model processed if-then statements on a cell-by-cell basis with *ARC/INFO*'s DOCELL command. It should be noted that the model did not use any initial (i.e. pre-existing) vegetation grids to simulate wetland class.

The wetland vegetation classes assigned to a cell based on the duration of hydrologic condition, flooded or dewatered (x-axis), and substrate height above or below the current year lake level (y-axis) are illustrated in Figure 4.14 - Matrix 1. The bottom-left corner of the matrix represented cells that were flooded for more than 40 years, and were currently at a water depth of greater than 200 cm. The top-right corner of the matrix represented cells that were dewatered for more than 40 years with a current year substrate elevation greater than 100 cm above the lake level. For example, if the cell was flooded for 6 to 10 years and was less than 80 cm below the lake level, the vegetation model would assign emergent vegetation (E) to that cell.

Rules for the model were constructed to reflect the interrelationship between lake levels and lag effects as transitions between wetland communities were not immediate and often occurred after one or more years of persistent flooding or dewatering. Where cells had recently become flooded or dewatered (i.e. cells that had been flooded or dewatered for less than a year), conditions from the previous year were examined to determine what wetland class should be assigned (Figure 4.14 - Matrix 2). For example, cells that had recently been flooded (i.e. flooded for less than one year) were previously dry the year before. Therefore, these cells could be assigned to one of three wetland classes – emergent, meadow, or treed/shrub – depending on the length of time the cells were dewatered and the height above lake level because these communities likely persist during such a short period of flooding. For instance, if in the previous year, the cell was dewatered from 1 to 5 years and the height of the cell was less than 30 cm above lake level, the cell was assigned to emergent vegetation. Water-tolerant emergent vegetation persists after one year of flooding. If that cell was greater than 30 cm above lake level, meadow marsh was assigned to the cell; it was assumed that meadow marsh existed in this cell prior to the current years flooding and persisted in this cell during the short period of high water levels.

4.3.1.4 Model Evaluation

The rule-based vegetation model was evaluated by comparing the area and spatial distribution of the simulated wetland vegetation community surfaces to the historical air interpreted wetland vegetation data. Simulations were run for each year of historical wetland data for all eight wetlands on Lakes Erie and Ontario. The model's simulated wetland surfaces were overlaid with the actual historical wetland data to provide an indication of spatial accuracy. The percentage of correctly predicted cells was computed for each wetland and year at the landscape and class level. At the class level, communities were predicted with good success if the percentage of correctly predicted cells was greater than 50%, with moderate success if the percentage ranged from 20 to 50%, and with poor success if the percentage was less than 20%. Actual and simulated areas of wetland vegetation communities were also compared to determine how well the model simulated the areal extent of each community. Finally, *FRAGSTATS* was rerun on the simulated wetland surfaces and then trends in the landscape and class metric values were compared to the actual historic trends to determine how well the model performed at simulating changes in the spatial pattern and structure of the wetland with water level changes.

MATRIX 1 (Current Year) — Duration of Hydrologic Condition (years)

	Height Above/Below Lake Level (cm)	Flooded > 40	Flooded 31-40	Flooded 21-30	Flooded 11-20	Flooded 6-10	Flooded 1-5	Flooded 0	Dewatered 0	Dewatered 1-5	Dewatered 6-10	Dewatered 11-20	Dewatered 21-30	Dewatered 31-40	Dewatered > 40
Above	> 101								See Matrix 2	M	T	T	T	T	T
Above	91-100									M	M	T	T	T	T
Above	81-90									M	M	T	T	T	T
Above	71-80									M	M	M	M	T	T
Above	61-70									M	M	M	M	T	T
Above	51-60									M	M	M	M	T	T
Above	41-50									M	M	M	M	T	T
Above	31-40									M	M	M	M	T	T
Above	21-30									E	E	E	M	M	T
Above	11-20									E	E	E	M	M	T
Above	0-10									E	E	E	M	M	T
Below	1-10	W	E	E	E	E	E	See Matrix 2							
Below	11-20	W	EF	E	E	E	E								
Below	21-30	W	EF	E	E	E	E								
Below	31-40	W	EF	E	E	E	E								
Below	41-50	W	EF	E	E	E	E								
Below	51-60	W	W	E	E	E	E								
Below	61-70	W	W	E	E	E	E								
Below	71-80	W	W	E	E	E	E								
Below	81-90	W	W	EF	EF	EF	EF								
Below	91-100	W	W	EF	EF	EF	EF								
Below	101-110	W	W	EF	EF	EF	EF								
Below	111-120	W	W	EF	EF	EF	EF								
Below	121-130	W	W	EF	EF	EF	EF								
Below	131-140	W	W	EF	EF	EF	EF								
Below	141-150	W	W	EF	EF	EF	EF								
Below	151-160	W	W	EF	EF	EF	EF								
Below	161-170	W	W	EF	EF	EF	EF								
Below	171-180	W	W	EF	EF	EF	EF								
Below	181-190	W	W	EF	EF	EF	EF								
Below	191-200	W	W	EF	EF	EF	EF								
Below	≥ 201	W	W	W	W	W	W								

MATRIX 2 (Previous Year) — Duration of Hydrologic Condition (years)

	Height Above/Below Lake Level (cm)	Flooded > 40	Flooded 31-40	Flooded 21-30	Flooded 11-20	Flooded 6-10	Flooded 1-5	Flooded 0	Dewatered 0	Dewatered 1-5	Dewatered 6-10	Dewatered 11-20	Dewatered 21-30	Dewatered 31-40	Dewatered > 40
Above	> 101								M	M	T	T	T	T	T
Above	91-100								M	M	M	T	T	T	T
Above	81-90								M	M	M	T	T	T	T
Above	71-80								M	M	M	M	M	T	T
Above	61-70								M	M	M	M	M	T	T
Above	51-60								M	M	M	M	M	T	T
Above	41-50								M	M	M	M	M	T	T
Above	31-40								M	M	M	M	M	T	T
Above	21-30								E	E	E	E	M	M	T
Above	11-20								E	E	E	E	M	M	T
Above	0-10								E	E	E	E	M	M	T
Below	1-10	Ex	E	E	E	E	E	E							
Below	11-20	Ex	E	E	E	E	E	E							
Below	21-30	Ex	E	E	E	E	E	E							
Below	31-40	Ex	E	E	E	E	E	E							
Below	41-50	Ex	E	E	E	E	E	E							
Below	51-60	Ex	Ex	E	E	E	E	E							
Below	61-70	Ex	Ex	E	E	E	E	E							
Below	71-80	Ex	Ex	E	E	E	E	E							
Below	81-90	Ex	Ex	E	E	E	E	EF							
Below	91-100	Ex	Ex	E	E	E	E	EF							
Below	101-110	Ex	Ex	E	E	E	E	EF							
Below	111-120	Ex	Ex	E	E	E	E	EF							
Below	121-130	Ex	Ex	E	E	E	E	EF							
Below	131-140	Ex	Ex	E	E	E	E	EF							
Below	141-150	Ex	Ex	E	E	E	E	EF							
Below	151-160	Ex	Ex	E	E	E	E	EF							
Below	161-170	Ex	Ex	E	E	E	E	EF							
Below	171-180	Ex	Ex	E	E	E	E	EF							
Below	181-190	Ex	Ex	E	E	E	E	EF							
Below	191-200	Ex	Ex	E	E	E	E	EF							
Below	≥ 201	Ex	Ex	Ex	Ex	Ex	Ex	Ex							

Figure 4.14 Decision rule matrix for the wetland vegetation model, where W = water, Ex = exposed substrate, EF = emergent/floating mixed, E = emergent, M = meadow marsh, T = treed/shrub

4.3.2 Results of the Historic Wetland Modelling

Overall, the rule-based vegetation model was 57% accurate in predicting wetland vegetation community response in Lake Ontario marshes that were influenced by regulated water levels. Presqu'ile was the most successfully simulated wetland on Lake Ontario as 69% of the cells were correctly predicted on average over all years. The modelled wetland data for the low and high water level period are provided for Presqu'ile in Figure 4.15. The least accurately predicted wetland on Lake Ontario was Lynde Creek with 45% of the cells correctly predicted. The vegetation model performed better at simulating wetland response on the unregulated marsh communities along Lake Erie. Overall, 74% of the cells were correctly predicted. The most accurately modelled wetland was Long Point, where 84% of the cells were predicted correctly overall. The modelled wetland vegetation surfaces for a low and high water level for Long Point are provided in Figure 4.16. Vegetation modelling results for Turkey Point were the least accurate of the Lake Erie wetlands with only 55% of the cells correctly predicted. The results of the historic wetland modelling, including an assessment of correctly predicted cells at the community level and comparisons of the predicted and actual areal extent and *FRAGSTATS* metrics, are summarized in Table 4.17. Results are discussed, followed by recommendations for future modelling. The accuracy assessment for each wetland is provided in Appendix 4.6 and the landscape and class metrics computed for the modelled surfaces are provided in Appendices 4.7 and 4.8, respectively.

4.3.2.1 Discussion

The wetland vegetation model developed for this analysis proved moderately successful in simulating historical wetland community response to water levels. The model was able to capture the dynamic nature of wetland vegetation change. As water levels rose, the model successfully simulated an increase in open water and a landward migration of the emergent vegetation community. During periods of declining water levels, the model simulated a lakeward expansion of the emergent community, and in several wetlands, an increase in treed/shrub and meadow marsh communities. The model performed better in simulating wetland community response at Presqu'ile on Lake Ontario and Long Point on Lake Erie, two lacustrine protected embayment wetlands. The model generally performed better during historically wet years (i.e. mid 1950s, late 1960s/early 1970s, mid 1980s) as compared to drier years. Water and emergent communities were often simulated with greater success, which increased the overall accuracy of many wetland sites during these wetter years. Meadow marsh and treed/shrub vegetation were predicted with moderate success depending on the wetland and year. Emergent/floating mixed was predicted with moderate success on the Lake Ontario wetlands but with poor success on Lake Erie. Exposed substrate was poorly simulated in the model on both lakes. This wetland community was only simulated in certain wetlands (all four Lake Ontario wetlands, Rondeau, Dunnville) when water levels were relatively low for several years or more. Exposed substrate may have been difficult to model because the community does not exist in all wetlands and years and because of a lack of detail within nearshore areas of the elevation models.

The landscape and class metrics computed for the modelled wetland surfaces did not consistently simulate the temporal trends that were observed in the historical wetland data. Analyses of the metrics computed for the modelled wetland surfaces were comparable for certain wetlands (i.e. Long Point, Turkey Point, Rondeau, Presqu'ile) and communities, but were inconsistent in other wetlands (i.e. Dunnville, South Bay, Hay Bay, Lynde Creek) and communities. In many wetlands, trends in the metrics computed for the simulated wetland communities were opposite to those identified in the historic datasets, or the metrics were not as strongly correlated to water level fluctuations (e.g. South Bay). The landscape and class metrics computed for some wetland communities in certain years and wetlands were also identical once various water depth and durations thresholds were reached. For example, metrics for open water at Hay Bay were identical in 1953 and 1962 during periods of lower water levels, and again in 1978, 1986, and 1999 during a period when water levels rose to record highs. Metrics computed for treed/shrub vegetation at Hay Bay were also identical for all years from 1953 to 1986. Similar patterns also occurred in the open water and treed/shrub communities in all other wetlands, in emergent vegetation at Long Point and Turkey Point, and in meadow marsh at Lynde Creek and South Bay. At this point of model development, the temporal trend analysis was not the best indicator of model performance and therefore temporal trends analyses was not used to assess impacts of future water level changes on wetland vegetation in Chapter 7 of this report.

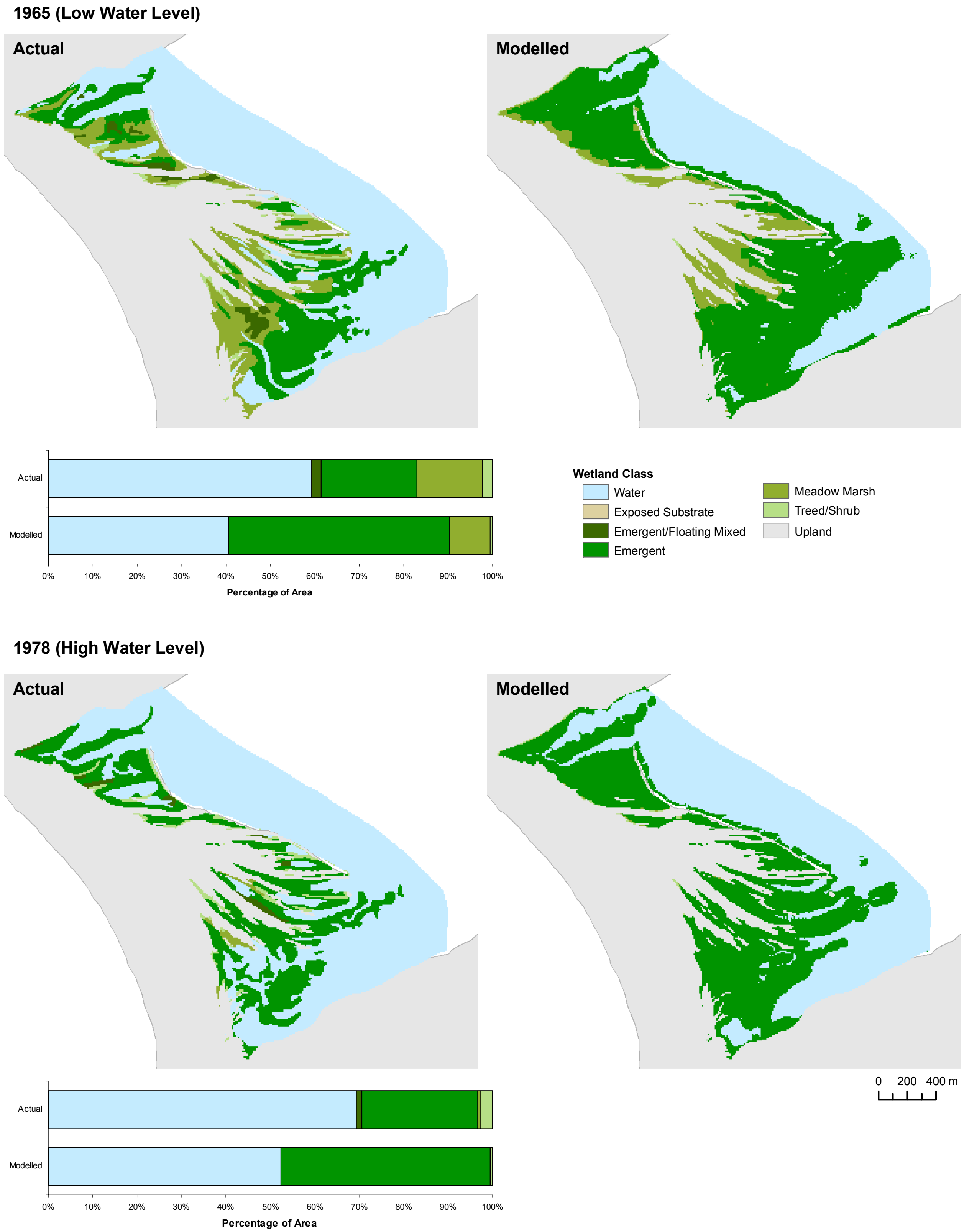

Figure 4.15 Actual and modelled wetland vegetation community surfaces for Presqu'ile, 1965 (low water year) and 1978 (high water year)

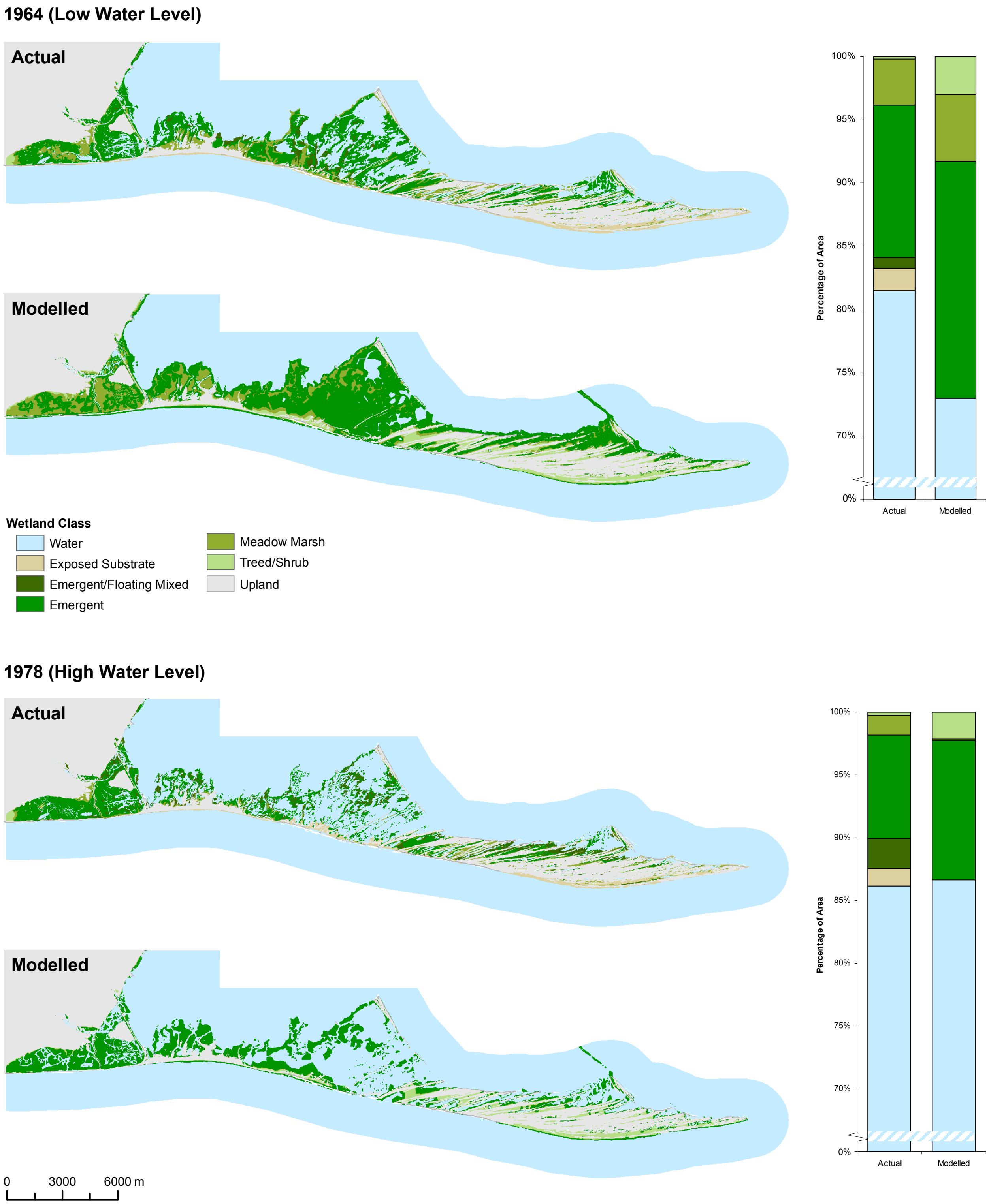

Figure 4.16 Actual and modelled wetland vegetation community surfaces for Long Point, 1964 (low water year) and 1978 (high water year)

Table 4.17 Summary of the wetland model assessment for Lakes Ontario and Erie wetlands

Wetland	Percent of Correctly Predicted Cells (%)*				Areal Comparisons	Landscape and Class Metric Comparisons	Other Observations
		By Community					
	Overall	Good	Moderate	Poor			
LAKE ONTARIO							
Hay Bay	60.4%	W: 90.2% E: 59.8%	M: 22.9%	EF: 7.4% T: 1.1%	• Over-estimated area of W • Under-estimated area of E • Area of all wetland communities predicted within 18 and 28 ha in 1978 and 1986, respectively	• Landscape metrics correlated well with water level fluctuations; trends consistent with historical trends • Class metrics for W, EF, E, M not as strongly correlated and opposite to historical trends; metrics for T more strongly correlated and similar trends	• Model performed better during high water years (1978, 1953, 1986) • Ex did not exist in any year, predicted in 1931, 1953
Lynde Creek	45.1%	E: 65.4%	M: 45.1% T: 25.9%	W: 13.8% Ex: 0.7% EF: 13.3%	• Under-estimated area of W (except 1927) • Under-estimated area of E in low water years (1929, 1959, 2001) and over-estimated in high water years (1954, 1978, 1986)	• Trends in landscape metric inconsistent with historical trends • Trends in class metrics for W, ED inconsistent and not as strongly correlated to water level fluctuations • Class metrics for E, M more strongly correlated but inconsistent with historical trends; trends in T opposite to historical trends	• Ex not predicted (except 2001)
Presqu'ile	68.5%	W: 77.6% E: 82.7%		Ex: 1.3% EF: 10.6% M: 15.5% T: 2.9%	• Over-estimated area of E • Under-estimated area of W (except 1999)	• Trends in landscape metrics responded similarly to historical trends (but no strong correlations) • Trends in class metrics for Ex, M, T consistent with historical values • Class metrics for W, E opposite to historical trends (but no strong correlations)	• M predicted with moderate success in 1965, 1971, 1999 • EF predicted with good success in 1999 • Ex only predicted in 1931, 1953
South Bay	55.7%	W: 60.7% E: 68.8%	M: 22.4% T: 33.3%	Ex: 0.0% EF: 7.9%	• Over-estimated area of E (except 1999) • Under-estimated area of W (except 1929)	• Landscape metrics more strongly correlated to water level fluctuations, generally opposite to historical trends • Class metrics inconsistent with historical values, opposite to historical trends	• Model performed better during high water years (1978, 1986)
LAKE ERIE							
Dunnville	74.7%	W: 84.0% E: 69.1%		Ex: 1.2% EF: 2.8% M: 15.6% T: 14.1%	• Over-estimated area of E (except 1995, 1999) and T • Under-estimated area of W, M	• Landscape metrics opposite to historical trends • Class metrics for W opposite to historical trends; EF similar; E similar and stronger; and M, T stronger and opposite	• Ex only predicted in 1934
Long Point	84.5%	W: 94.3% E: 51.7%	T: 22.1%	Ex: 0.0% EF: 14.2% M: 2.2%	• Over-estimated area of T • Under-estimated area of M	• Trends in class metrics consistent to historical trends; O, EF, T not as strongly correlated to water level fluctuations; E, M more strongly correlated	• Ex not predicted • EF not predicted in 1964, 1968, 1978, 1995
Rondeau	80.1%	W: 89.1%	E: 39.4%	Ex: 0.0% EF: 12.1% M: 0.8% T: 2.4%	• All wetland classes predicted within 15 ha of actual areas (1978 to 1995)	• Landscape metrics similar to historical trends • Class metrics for O, EF, T consistent with historical trends, but not as strongly correlated • Class metrics for E, M similar to historical values and more strongly correlated with water level fluctuations	• Captured loss of wetland vegetation in nearshore after 1978 • EF predicted with good success in 1955 and moderate success in 1985
Turkey Point	55.5%	W: 67.6% E: 71.0%	T: 85.0%	Ex: 8.4% EF: 12.2% M: 1.9%	• Over-estimated area of E (except 1995, 1999) and T • Under-estimated area of W, M	• Trends in landscape and class metrics similar to historical trends, few strongly correlated metrics	• Ex not predicted • M not predict in 1945, 1955, 1971

* The percent of correctly predicted cells overall is an average of all years per wetland; the percent of correctly predicted cells by community are listed according to good success (over 50% of the cells correctly predicted), moderate success (20 to 50% of the cells are correctly predicted) and poor success (<20% of the cells are correctly predicted)
W = water, Ex = exposed substrate, EF = emergent/floating mixed, E = emergent, M = meadow marsh, T = treed/shrub

The historical temporal analysis indicated that the spatial distribution and pattern of wetland classes were correlated, to some extent, to water level changes on Lake Ontario, particularly at Hay Bay and Lynde Creek. The model, however, did not perform well in simulating wetland vegetation response in these wetlands. In fact, the modelling results for Lake Ontario were less accurate than for Lake Erie even though the temporal trend analysis of the historic data indicated that wetland response was more strongly correlated with water level fluctuations on Lake Ontario. This was likely due to a lack of detail in the elevation surfaces for the Lake Ontario wetlands. Available elevation information did not have enough vertical resolution to accurately capture local variations in the elevation surface. For example, at Lynde Creek, all elevation spot heights located in the marsh north of the causeway had the same value (175 m). Spot height data did not capture the micro-topography that is known to occur in this area, and therefore the model did not accurately simulate the different wetland classes that have historically occurred in this area. It should also be noted, that there is also a source error associated with using a present day elevation model to predict historical hydrologic conditions. There would have been changes in the wetland bathymetry due to accretion from deposition and storm/ice scouring over time.

On Lake Erie, the historical temporal analysis indicated that there was less correlation between wetland classes and water level fluctuations compared to Lake Ontario, but the model performed better in simulating wetland communities correctly for marshes on Lake Erie. The more accurate results for Lake Erie could be attributed to two main factors. First, there was more bathymetry and land elevation data in the nearshore at Long Point, Dunnville, and Rondeau on Lake Erie compared to the Lake Ontario sites. Second, the success at Rondeau and Long Point may be related to the amount of lake (or water) area included within the study area. If open water had comprised a smaller proportion of the wetland area, the modelling results would have been less accurate.

Lake Ontario water level regulation did not appear to affect the wetland community model results. There was no difference in model accuracy pre- or post-1958 when water level regulation was initiated on Lake Ontario. The wetland community model results were, however, affected by hydrogeomorphic wetland type. The model performed significantly better on protected embayment wetlands. In fact, Long Point and Rondeau on Lake Erie and Presqu'ile on Lake Ontario were the most successfully modelled wetlands. Differences in the accuracies were likely due to these wetlands having more elevation information compared to other sites. However, the model rules may have also been more suited for protected embayment wetlands. The results based on the drowned river-mouth wetlands on Lake Ontario were inconclusive. Hay Bay had the second highest percentage of cells correctly predicted on Lake Ontario, while Lynde Creek had the lowest. Dunnville on Lake Erie, also a drowned river-mouth, was simulated more successfully, but this could be due to the better land elevation and detailed river basin bathymetry compared to the Lake Ontario sites. The model may not be as successful at simulating wetland response on drowned river-mouth wetlands since these wetlands are also influenced by river discharge and overland flows from precipitation events.

Visually, the simulated wetland vegetation community surfaces appeared simpler and less complex compared to the historical data. Patches of open water and emergent vegetation and, in some wetlands, meadow marsh and treed/shrub, were larger and more continuous. Depending on the topography of the wetland, patches did form in narrow bands along elevation gradients. Patches of emergent/floating mixed vegetation simulated by the model typically occurred in elongated but narrow patches along the shoreline coinciding with elevation gradients in the topography model. Exposed substrate also formed in narrow bands or was sporadically distributed in small clusters of one or more cells. Historical wetland vegetation community surfaces were more fragmented with patches of different communities interspersed across the landscape. The shapes of the patches in the historical data also appeared more realistic in shape and distribution compared with the simulated model. It was easy to identify the important role that the horizontal resolution of the elevation model had in simulating the location of wetland communities. The locations of elevation errors quickly became apparent in the modelling results.

4.3.2.2 Recommendations

Obtaining more accurate bathymetry and elevation data at a higher vertical and horizontal resolution is critical to improving accuracy in the wetland vegetation community response model. Detailed bathymetry and land elevation data simply do not exist for many wetland sites and are likely the primary constraints in accurately modelling wetland vegetation response for the study sites. Detailed bathymetry field sheets were obtained and provided valuable information regarding the elevation of the lake floor in deeper areas, but there was limited information closer to the shoreline, specifically in areas designated as marsh or swamp. Often the land contour and spot data provided limited information for these areas. Improving elevation data in the nearshore area is important, especially when modelling site-specific impacts of projected lake level declines under climate change.

Future modelling efforts require detailed hydrographic surveys of nearshore areas for wetland sites. If vegetation patch interspersion and micro-topography are important to the study, data points should be collected at least every ten metres (depending on study scale and objectives) and that the vertical resolution be captured to the nearest centimetre. Collection of elevation data collection also needs to extend out to a depth sufficient to capture the water level range projected under climate change. Under current 2050 climate change projections, lake levels may decline by one metre in the lower Great Lakes. A depth of three metres from the benchmark would be sufficient for the model to capture a lakeward migration of wetland communities.

Another step in future vegetation model development is to re-examine the decision rules. The model performed significantly better during high water level years indicating that the decision rules regarding emergent, meadow marsh, and treed/shrub communities on the dewatered component of the matrix may need to be modified. Since the rules were based on literature review and field surveys, it would be best to first acquire more detailed and accurate elevation data for some of the study sites to test the rules further. The model may not have captured community response to small changes in water levels due to the lack of detailed elevation data within the wetland area. Furthermore, water depth and duration, are not the only important variables that influence wetland vegetation response especially within the drier regions of the wetland. There may be other constraining factors that influence growth and response and should therefore be explored such as slope, soil, and the seedbank of wetland vegetation communities.

The wetland response model developed in this study provides the framework for building a more complex and integrated wetland model. Future versions of the model could incorporate soil and substrate data (e.g. clay, silt, sand, mud) to better identify the types of wetland vegetation communities that would grow at particular locations in the wetland; substrate type may also impede vegetation growth lakeward. Another consideration would be to include slope and aspect; slope could also limit the lakeward and landward migration of wetland vegetation communities, and aspect may be important when considering fetch, and wind and wave speed and their role in defining the spatial distribution of vegetation communities within the wetland. Substrate and fetch could also be included to model the extent of submergent vegetation more accurately which would significantly aid the fish modelling component of the project.

The current model does not consider land use changes, or human influences and modifications at the wetland sites. All non-wetland areas were masked out for the analysis, including wooded upland areas. As the wetland migrates lakeward, areas of treed/shrub in the upper extent of the wetland would eventually transition to wooded upland as moisture conditions decline. The model does not define what the upper wetland extent would be, or provide an indication of how much of the treed/shrub community would be lost as the community transitions to wooded upland or due to land use change from humans. The model assumes that there will be no future modifications to the wetland as most of the study sites are protected and development in the wetlands is limited. Land use changes could be incorporated into the model by defining or masking out the areas where human structures or modifications may be built (i.e. dyke) or by gradually expanding upland areas on a cell-by-cell basis per year.

4.4 Summary

The spatiotemporal analysis demonstrated that Great Lake coastal wetlands responded to hydrologic changes due to climate variability. As water levels declined, several changes in the distribution and area of wetland vegetation communities occurred: vegetation in the wetland tended to drier communities; there was a lakeward migration of the wetland, especially in the emergent community; there was an expansion of emergent, meadow marsh, and treed/shrub communities while open water and emergent/floating mixed communities declined; there were larger solid and continuous patches of drier vegetation in the wetland; and the wetland was less fragmented, complex, and interspersed.

The rule-based model developed in this chapter worked reasonably well in simulating historic wetland response to water levels fluctuations. To summarize, the wetland response model developed in this chapter and applied to historical wetland data:

- Captured the dynamic nature of wetland change in response to declining and rising water level conditions;
- Performed best in simulating wetland response at two lacustrine protected embayment wetlands, Presqu'ile on Lake Ontario and Long Point on Lake Erie;
- Performed better in simulating wetland response during historically wet years;
- Predicted the open water and emergent communities with good success (i.e. ≥50% of the cells within these communities were correctly predicted);
- Over-estimated the area of emergent vegetation and under-estimated the area of open water in most wetlands;
- Predicted treed/shrub vegetation and meadow marsh (for some wetlands) with moderate success (i.e. the percentage of correctly predicted cells ranged from 20 to 50%); and
- Predicted the exposed substrate and emergent/floating mixed communities with little or no success.

It is critical, however, for future modelling efforts that more accurate bathymetry and elevation data of wetlands be collected as input for the model. Other information, such as soil and substrate, could also be incorporated into the model to improve the results (provided the data exist). Nevertheless, the wetland model developed in this chapter was applied to model future wetland response to projected climate change and water level declines. When analyzing the results of the climate change modelling presented in Chapter 7, it is important to keep in mind the results of the historic wetland vegetation community analysis summarized above. The simulated wetland vegetation surfaces were also used to model the impacts of water level fluctuations on bird and fish communities. Development of the bird and fish models is presented in Chapters 5 and 6 respectively. In Chapter 7, the wetland response model along with the bird and fish suitability models were used to assess the impacts of projected climate change and water level fluctuations on vegetation, bird, and fish communities in coastal wetlands on Lakes Erie and Ontario.

4.5 REFERENCES

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5.0 Vulnerability of Marsh Birds in Great Lakes Coastal Wetlands to Climate-Induced Hydrological Change

Shawn Meyer, Joel Ingram, and Krista Holmes

A combination of literature review and predictive model development were used to obtain a better understanding of vulnerabilities and potential responses of marsh bird communities to climate-induced hydrological change (Figure 5.1). The literature review was undertaken to document and assess the vulnerability of common Great Lakes coastal marsh breeding birds and allow for the development of hydrological vulnerability indices for these birds. In addition, bird survey and habitat data were used to develop quantitative relationships between species presence and relative abundance in relation to wetland plant communities and water levels. These quantitative relationships were used to predict potential changes to bird communities at several coastal wetlands under various climate change scenarios. These results, aided in the identification of vulnerable bird species and nesting guilds for which monitoring, and potential management strategies could be designed to alleviate the potential effects of climate change.

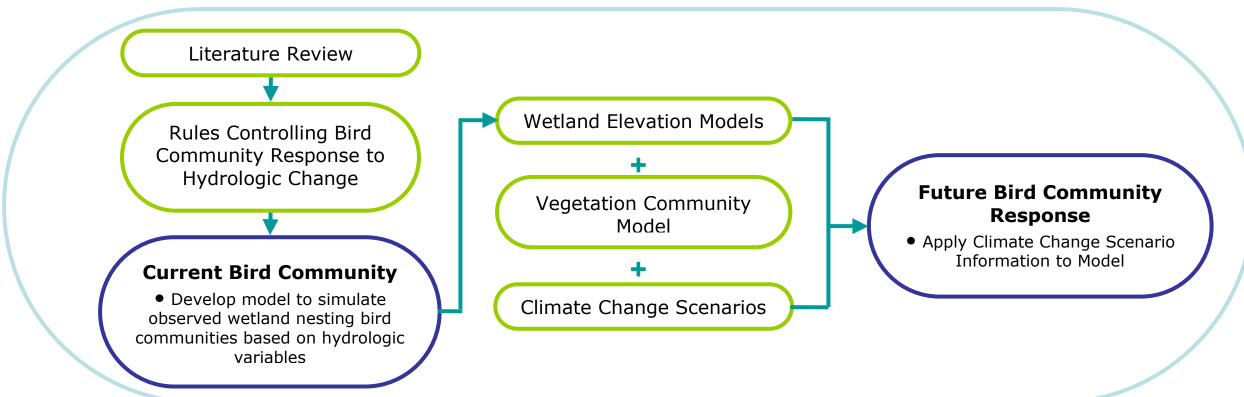

Figure 5.1 Flow diagram of approach used to evaluate marsh bird vulnerabilities and potential response to climate-induced hydrological change

5.1 Assessment of the Hydrological Vulnerability of Selected Marsh Birds Breeding in Coastal Wetlands on the Lower Great Lakes

Climate change has the potential to alter wetland habitat along the Great Lakes shoreline if water levels change. Habitats for many marsh birds, particularly marsh nesting obligate birds (i.e. birds that nest exclusively in marshes with rare exceptions) will be affected. These birds have several breeding requirements that are vulnerable to any change in habitat. Many marsh nesting obligate birds breed in Great Lakes coastal marshes including, Pied-billed Grebe (*Podilymbus podiceps*), American Bittern (*Botaurus lentiginosus*), Least Bittern (*Ixobrychus exilis*), Yellow Rail (*Coturnicops noveboracensis*), King Rail (*Rallus elegans*), Virginia Rail (*Rallus limicola*), Sora (*Porzana carolina*), Common Moorhen (*Gallinula chloropus*), American Coot (*Fulica americana*), Forster's Tern (*Sterna forsteri*), Black Tern (*Chlidonias niger*), Marsh Wren (*Cistothorus palustris*), and Swamp Sparrow (*Melospiza georgiana*) (Peck and James 1983; Timmermans 2001; Tozer 2003). Facultative marsh nesting birds, such as Red-winged Blackbird (*Agelaius phoeniceus*), Common Grackle (*Quiscalus quiscula*), Canada Goose (*Branta canadensis*), and several other species of waterfowl, also use coastal wetlands for nesting habitat. These birds readily nest in marsh and upland habitats, and thus, are less vulnerable to changes in coastal wetland habitat (Peck and James 1983, 1987; Poole and Gill 1992 - ongoing; Tozer 2003).

Wetland habitat suitability for most marsh breeding birds can be divided into landscape and local factors. At the landscape level, wetland habitat suitability for many marsh breeding birds is determined by wetland size, habitat availability, and interspersion. These dynamic landscape factors affect marsh bird community diversity and abundance by influencing the availability of breeding habitat requirements, such as patch size and open water. At the local level, the requirement by many marsh birds for specific plants, or standing water, will also affect habitat suitability within certain areas of a wetland. However, the life history of a marsh breeding bird will ultimately determine its hydrological vulnerability to climate change by influencing breeding habitat requirements and recruitment rate (i.e. population growth). Life history traits (e.g. body size) also determine wetland habitat suitability by affecting breeding habitat requirements, such as the structure of vegetation required to support a nest. The potential alteration of recruitment rate, which is related to a species' life history, may affect its hydrological vulnerability more than breeding habitat requirements. For example, nest vulnerability to flooding, dislodgement, or stranding, is affected by the time required to egg-lay, incubate, and brood rear (or total nest exposure period). Therefore, if projected hydrological changes occur because of climate change, such as increased flooding, some populations of marsh bird species with long nest exposure periods may decline due to reduced recruitment rates. Moreover, the responsiveness of a breeding bird population to compensate for hydrological changes (e.g. increasing recruitment during favourable breeding conditions) is also related to life history, and consequently, may affect a species' hydrological vulnerability by influencing population persistence. Current population size also influences population persistence through dispersal rates and capacity to buffer environmental changes, and therefore, influences a species' hydrological vulnerability. To assess the vulnerability of marsh bird species to climate induced hydrological changes on the Great Lakes, each of these habitat requirements, life history traits, and population parameters need to be considered.

5.1.1 Landscape Habitat Suitability

Many marsh birds are area-sensitive species (i.e. species that require wetlands of a certain size for suitable breeding habitat). Most marsh nesting obligate birds, such as Pied-billed Grebe, Black Tern, American Bittern, Virginia Rail, Sora, and Swamp Sparrow, are wetland area-dependent species (Johnson and Dinsmore 1986; Naugle *et al.* 2000, 2001; Riffell *et al.* 2001) and many others, such as Marsh Wren and Forster's Tern, are likely area-dependent (Johnson and Dinsmore 1986). Similarly, marsh breeding birds, such as Common Snipe (*Gallinago gallinago*), Spotted Sandpiper (*Actitis macularia*), Northern Harrier (*Circus cyaneus*), Eastern Kingbird (*Tyrannus caudifasciatus*), and Red-winged Blackbird also depend on large wetlands for breeding habitat (Gibbs *et al.* 1991; Riffell *et al.* 2001). Therefore, a species' minimum-area requirement determines whether a wetland provides suitable breeding habitat.

Hydrological changes, due to climate change, may affect wetland habitat suitability for many area-dependent marsh bird species if a reduction in the area of coastal wetland habitat occurs. Projected lower average water levels may reduce the size of many wetlands particularly where aquatic plant communities may not be able to advance lakeward due to barriers and/or exposure to high energy coastal processes (see Chapter 3). In these instances, bird communities may shift from area-sensitive marsh obligate and facultative marsh nesting birds, such as American Bittern, Pied-billed Grebe, Black Tern, Red-winged Blackbird, and Eastern Kingbird, to area-independent facultative marsh nesting birds, such as Common Grackle, Alder Flycatcher (*Empidonax alnorum*), and Common Yellowthroat (*Geothlypis trichas*) (Naugle *et al.* 1999; Riffell *et al.* 2001). This shift in bird communities, however, may not affect overall marsh bird species richness even though some bird species may be displaced. Consequently, use of simple indicators such as overall species richness during evaluation of climate-induced hydrological changes may not capture shifts within, and between nesting guilds.

The importance of wetland area for many marsh birds is likely due to the distribution and composition of wetland plants at a landscape level. A wetland supporting diverse plant communities interspersed with open water provides suitable nesting habitat for a diverse marsh bird community. Moreover, increased horizontal zonation (due to wetland plants of differing heights) and a wider range of water depths in a highly interspersed wetland optimizes foraging habitat for many marsh birds. For example, American and Least Bitterns nest within tall, dense cattail (*Typha* spp.) stands but require interspersed habitats with deep, open water for wading and foraging (Weller 1961; Gibbs *et al.* 1992a,b; Post and Seals 2000). Pied-billed Grebe (Muller and Storer 1999), Common Moorhen (Griej 1994; Bannor and Kiviat 2002), Virginia Rail (Conway 1995), and Sora (Melvin and Gibbs 1996) also require more vegetative cover for nesting than for foraging. Rails, however, use emergent vegetation in shallower water than grebes and moorhens because of different nesting and foraging behaviour (e.g. grebes and moorhens primarily swim whereas rails walk) (Conway 1995; Melvin and Gibbs 1996; Muller and Storer 1999). Therefore, a diverse and well interspersed wetland plant community (landscape) provides many more ecological niches (or spaces) for marsh birds than a monotypic wetland plant community. This explains why highly interspersed wetlands are positively related to marsh bird diversity (Kantrud and Stewart 1984; Craig and Beal 1992).

Many marsh birds also use habitat interspersion as a proximate cue of high quality breeding habitat (Kaminski and Prince 1981, 1984). During breeding, territorial conflicts often occur between birds and are usually related to visual display. Highly interspersed wetlands that support many micro-habitats with a high degree of patchiness between habitat and open water, will support a higher density of territorial breeding pairs than a monotypic wetland (Murkin *et al.* 1982; Kaminski and Prince 1984). A diverse mosaic of wetland plants and open water also benefits many marsh birds by providing more vegetation (e.g. submerged aquatic vegetation) and aquatic invertebrates for foraging birds than a monotypic wetland (Krull 1970; Murkin *et al.* 1982; Kantrud 1986; Angradi *et al.* 2001). In addition, canopy openings and open water within interspersed wetlands may improve mobility and stand accessibility for some marsh birds, such as bitterns, and thereby improve foraging efficiency (Benoit and Askins 1999). Consequently, marsh bird communities may change, if aquatic plant diversity and habitat interspersion change because of altered hydrology due to climate change.

Expansion of monotypic vegetation, such as cattail or common reed, because of lower water levels associated with climate change (see Chapter 3), may affect aquatic plant diversity and wetland habitat interspersion. This expansion may also displace some marsh birds as open emergent marsh and meadow marsh are replaced by monotypic stands of tall, dense vegetation. Pied-billed Grebe, Least Bittern, Black Tern, American Coot, and Mallard (*Anas platrhynchos*) (Gibbs *et al.* 1992b; Dunn and Agro 1995; Brisbin and Mowbray 2002; Drilling *et al.* 2002) depend on open emergent marsh within coastal wetlands for breeding habitat, whereas Northern Harrier, Eastern Kingbird, Swamp Sparrow, Le Conte's Sparrow (*Ammodramus leconteii*), and Sedge Wren (*Cistothorus platensis*) (Gibbs *et al.* 1991; MacWhirter and Bildstein 1996; Riffell *et al.* 2001) rely primarily on meadow marsh. These marsh birds may be replaced by marsh birds that use robust emergent vegetation (e.g. Virginia Rail, Sora, Marsh Wren, Red-winged Blackbird, Common Grackle, and Common Yellowthroat) as cattail and common reed expand (Conway 1995; Yasukawa and Searcy 1995; Melvin and Gibbs 1996; Kroodsma and Verner 1997; Peer and Bollinger 1997; Meyer 2003). This expansion, however, may negatively affect some of these marsh birds if structural changes in vegetation stands occur. For example, Virginia Rail and Sora tend to use tall, monotypic stands of vegetation with low stem density and litter accumulation (Johnson and Dinsmore 1986; Meyer 2003). If lower water levels occur due to climate change, stem density within some tall, monotypic vegetation stands may increase (Grace 1989) and may negatively affect Virginia Rail and Sora by reducing mobility and/or stand accessibility.

5.1.2 Local Habitat Suitability

The distribution and abundance of a bird species within a marsh are also affected by local plant species composition. Many marsh birds depend on a number of different plants during breeding and may shift from one habitat to another as requirements change. Therefore, these plants, or micro-habitats, are a crucial factor influencing breeding habitat suitability (Sutherland and Maher 1987) and can determine whether a bird uses a specific habitat within a wetland.

A species' life history determines the way that it interacts within a wetland. Due to contrasting life histories, marsh bird species depend on different wetland plants for nesting material and/or nest support (Steen *et al.* 2005). For example, small marsh passerines, such as Swamp Sparrow, build and support elevated nests with fine stemmed graminoid vegetation (e.g. sedges (*Carex* spp.)) (Peck and James 1987; Mowbray 1997) whereas relatively heavier non-passerine species, such as American Bittern (Peck and James 1983; Gibbs *et al.* 1992a), Least Bittern (Weller 1961; Peck and James 1983; Gibbs *et al.* 1992b; Tozer 2003), and Northern Harrier, require either strong structural vegetation (e.g. cattail or common reed) to support the weight of adults, eggs and chicks, or they nest on the ground.

Evolved habitat preferences may also affect habitat use by many marsh birds. Marsh Wrens nest in tall cattail or bulrush (*Scirpus* spp.) high above the water to prevent nest flooding (Peck and James 1987; Naugle *et al.* 2001; Riffell *et al.* 2001). Many platform nesting birds also have specific nesting habitat requirements. Virginia Rail and Sora require cattail or bulrush stands with a mixture of living and dead stems to build nests, ramps, or platforms with a relatively dense canopy (Johnson and Dinsmore 1986; Eddleman *et al.* 1988;

Kaufmann 1989; Gibbs *et al.* 1992a,b; Meanley 1992; Conway 1995; Tozer 2003). Other platform nesting birds, such as Pied-billed Grebe (Muller and Storer 1999), American Coot (Arnold *et al.* 1993), Common Moorhen (Helm *et al.* 1987) and Forster's and Black Terns (Cuthbert 1954; Bergman *et al.* 1970; Dunn 1979; Chapman Mosher 1986; Linz *et al.* 1994; McNicholl *et al.* 2001), require floating mats of vegetative litter in areas of deeper water and less emergent vegetation to build and support nesting platforms. Therefore, nesting habitat suitability for most marsh breeding birds is primarily determined by local wetland plant communities. Most wetland plants, however, are affected by hydrology (see Chapter 3). Consequently, any change in wetland plant communities (e.g. loss of a species), due to hydrological changes associated with climate change, may affect the distribution and abundance of a bird species within a wetland.

A second local habitat characteristic that affects nesting habitat suitability for marsh breeding birds is the presence of standing water within vegetation. All marsh nesting obligate birds require standing water for breeding and will switch nesting habitats if water levels change. For example, Marsh Wren only nest above water (Peck and James 1987) and will switch from cattails to bulrushes as water levels drop (Verner and Engelsen 1970). Similarly, American and Least Bitterns primarily nest over water (Middleton 1949; Peck and James 1983; Post 1998; Tozer 2003) or at least near open pools of water (Weller 1961; Gibbs *et al.* 1992a, 1992b), and nests of American Coot (Sutherland and Maher 1987), Common Moorhen (Fredrickson 1971), and Swamp Sparrow (Greenberg 1988; Mowbray 1997) correlate positively with the presence of standing water. Johnson and Dinsmore (1986) also showed that breeding Virginia Rail and Sora only occupied sites with standing water. In addition, Black and Forster's Terns require standing water around nesting substrates (e.g. mats of residual vegetation, woody debris, or muskrat lodges) (Peck and James 1987; Dunn and Agro 1995). Lower water levels in Great Lakes coastal wetlands, due to climate change, have the potential to affect nesting habitat suitability of all marsh nesting obligate birds, if vegetation with standing water becomes less abundant.

Nest flooding, dislodgement, or stranding, commonly result in reproductive failure for many marsh birds, may also become more frequent due to climate change. The vulnerability of a bird nest to flooding, or stranding, is directly related to species-specific nest characteristics, including whether the nest is anchored or free-floating, and nest height above the water (Steen *et al.* 2005). Many marsh breeding birds build platform nests; some are anchored to surrounding vegetation including those of Pied-billed Grebe (Steen *et al.* 2005), American Coot (Alisauskas and Arnold 1994; Brisbin and Mowbray 2002), Common Moorhen (Peck and James 1983; Helm *et al.* 1987), and Virginia Rail (Peck and James 1983) while others are free-floating, such as Black Tern (Cuthbert 1954; Bergman *et al.* 1970; Dunn 1979; Peck and James 1983; Linz *et al.* 1994). Anchored nests may be more vulnerable to flooding because of their relatively fixed position. For example, frequent flooding is a common reason for nest failure in American Coot (Sooter 1945; Weller 1971), Common Moorhen (Helm *et al.* 1987), and Virginia Rail (Walkinshaw 1937; Griese *et al.* 1980). Although free floating nests rise and fall with water level changes, nest break-up, or dislodgement from surrounding vegetation, may occur when water levels rise (Bergman *et al.* 1970; Dunn and Agro 1995). Elevated nests, such as those built by Marsh Wren (Peck and James 1987; Poole and Gill 1992 - ongoing), Swamp Sparrow (Peck and James 1987; Mowbray 1997), American Bittern (Gibbs *et al.* 1992a), Least Bittern (Gibbs *et al.* 1992b), and Red-winged Blackbird (Yasukawa and Searcy 1995), may also be vulnerable to increases in water levels. Weller (1961) and Kale (1965) documented nest collapse and/or flooding as common reasons for nest failure in Least Bittern and Marsh Wren, respectively. Moreover, lowering of water levels during nesting, due to climate change, may also result in reproductive failure because of increased nest predation due to nest stranding (Post 1998). Consequently, most platform nesting birds and some elevated nesters may be displaced, if hydrological changes result in more nest flooding, dislodgement, or stranding, than occurred historically.

The vulnerability of a nest to flooding, dislodgement, or stranding, is also related to adaptations in egg morphology and nesting behaviour. Although nest location and total nest exposure period (see Life History Traits Section 5.1.3) likely contribute most to the overall hydrological vulnerability of a marsh bird, other nesting adaptations, such as egg hardiness and the addition of nest material while incubating, also affect a species' hydrological vulnerability. Highly porous eggs, such as those produced by Black Tern and Pied-billed Grebe, may improve hatching success, particularly in nests that are constantly water-soaked (Davis and Ackerman 1985; Muller and Storer 1999). Thus, slight increases in water levels may not affect Black Terns and Pied-billed Grebes as much as other marsh birds that do not produce porous eggs. In addition, adaptable nesting behaviour, such as adding nest material to prevent egg flooding as water levels rise, or readily accepting nest disturbance associated with flooding, may reduce a species' hydrological vulnerability. Many marsh birds, such as American Coot (Brisbin and Mowbray 2002), Common Moorhen (Burtch 1917), Virginia Rail (Walkinshaw 1937), Sora (Walkinshaw 1940), and Black Tern (Richardson 1967), are highly adaptable birds while nesting, and hence, may be less vulnerable to hydrological changes than less adaptable birds. However, severe hydrological events, such as flash flooding, may result in nest flooding, or dislodgement for which existing morphological and behavioural adaptations fail to compensate.

The frequency and severity of flash floods will also affect many ground nesting marsh birds. Although most marsh nesting obligate birds build elevated nests, American Bittern, Common Moorhen, Virginia Rail, and Swamp Sparrow occasionally nest on the ground (Peck and James 1983, 1987; Helm *et al.* 1987; Greenberg 1988; Greenberg and Droege 1990; Mowbray 1997). Many shorebirds and waterfowl also depend on the upland plant community of wetlands for nesting habitat (Peck and James 1983; Poole and Gill 1992 - ongoing). Flash floods and other hydrological changes, such as seiches or storm surges, often result in nest failure for many ground nesting birds because of nest flooding (Greenberg 1988; Greenberg and Droege 1990). This reproductive failure may become more common if flooding becomes more frequent and severe because of more intense precipitation events due to climate change.

Many marsh birds do not depend on coastal wetlands for nesting habitat but do rely on them for foraging habitat. Many waterfowl (Bellrose 1980; Steen *et al.* 2005), herons (Peck and James 1983; Butler 1992; Davis and Kushlan 1994), and blackbirds (Peck and James 1987; Yasukawa and Searcy 1995) nest in upland habitats but forage and brood rear in coastal wetlands. Therefore, hydrological changes will impact these marsh birds and marsh nesting obligate birds if changes in foraging habitat occur.

Most marsh birds forage on aquatic vegetation and/or invertebrates (Poole and Gill 1992 - ongoing). Breeding Red-winged Blackbird, Black Tern, and King Rail primarily forage on aquatic invertebrates (Bergman *et al.* 1970; Meanley 1992; Turner and McCarty 1998) whereas Canada Goose, Virginia Rail, Sora, and Common Moorhen feed on both aquatic invertebrates and vegetation (Horak 1970; Kaminski and Prince 1981; Johnson and Dinsmore 1986). Wild rice (*Zizania palustris*) and many aquatic plants, such as wild celery (*Vallisneria americana*) and sago pondweed (*Potamogeton pectinatus*), are important forage plants for many marsh birds (Horak 1970; Petrie 1998; Knapton and Petrie 1999). Similarly, many aquatic invertebrates, such as dragonflies and damselflies (Odonata), and mayflies (Ephemeroptera), are vital aquatic insects for many marsh birds (Poole and Gill 1992 - ongoing; Turner and McCarty 1998). Many of these aquatic plants, insects, and other animals, such as amphibians and small mammals, are also affected by wetland plant communities (Krull 1970; Kurta 1995; Harding 1997; Angradi *et al.* 2001; Meyer 2003). Therefore, climate-induced water level changes may affect foraging habitat for many marsh birds, if wetland plant communities and other food resources change. This may influence avian reproductive success as chick survival of many marsh birds (e.g. Black Tern, Northern Harrier, and Red-winged Blackbird) is commonly related to food availability (Chapman Mosher 1986; Welham and Ydenberg 1993; MacWhirter and Bildstein 1996; Zimmerling 2002).

Lower water levels, due to climate change, may reduce the distribution and abundance of certain aquatic plants, such as wild rice, as other species, such as cattail and common reed, expand because of wider water tolerances for germinating and growing (Kelsall and Leopold 2002, also see Chapter 3). Wetland plants, however, differ in their nutritional value for marsh birds. For example, common reed is neither a valued food item (Petrie 1998), nor does it support a diverse aquatic invertebrate community (Angradi *et al.* 2001). Moreover, common reed is likely to expand and replace open emergent marsh and meadow marsh as water levels drop (Wilcox *et al.* 2003; also see Chapter 3). Consequently, foraging habitat may decline, particularly within an open emergent marsh. More robust emergent marsh birds, such as Virginia Rail, Red-winged Blackbird, and Common Yellowthroat may replace open emergent marsh bird species, such as Black Tern, American Coot, and Common Moorhen.

5.1.3 Life History Traits

Reproductive life history traits also determine the vulnerability of some marsh birds to hydrological changes. Specifically, reproductive traits, such as clutch size and the length of incubation and nestling periods (time required until chicks fledge the nest), determine total exposure time of nests to any potential water level change. Generally, birds that produce young that have to be fed by the parents until fledging (altricial young), such as Swamp Sparrow and Marsh Wren, have smaller clutches than birds that produce young that are capable of foraging shortly after hatching (precocial young), such as Pied-billed Grebe and American Coot due to energetic requirements (Lack 1968; Ehrlich *et al.* 1988). In addition, precocial birds produce nutritionally richer eggs than altricial birds. Consequently, precocial chicks are capable of moving around on their own after hatching, while altricial birds produce blind, helpless young that are incapable of movement (Ehrlich *et al.* 1988). Not all precocial birds, however, are fully independent. Ehrlich *et al.* (1988) categorized precocial birds into four levels in relation to self-foraging ability. For example, Virginia Rail produce precocial chicks that stay near the nest for about three to four days after hatching and are fed by the parents during that time, while Mallard precocial chicks leave the nest within 24 hours and are capable of self-feeding. Therefore, greater independence of Mallard chicks may reduce their vulnerability to hydrological changes due to climate change.

The number of days required to successfully hatch a nest (i.e. lay a full clutch of eggs and incubate) is generally longer for precocial birds in comparison to altricial birds (Poole and Gill 1992 - ongoing). Although altricial birds have a long nestling period, greater in-egg development of precocial birds results in a period of incubation that generally equals, or exceeds, the combined incubation and nestling periods of altricial birds. For example, total exposure time of Swamp Sparrow nests to flooding is approximately 29 days given a clutch size of four eggs (four days required to lay a clutch) and an incubation and nestling period of approximately 14 and 11 days respectively (Peck and James 1987; Mowbray 1997). In contrast, the American Coot, which produce highly precocial young, require approximately seven days for egg laying, 24 days for incubation, and two days for young to leave the nest (total exposure time of 33 days) (Peck and James 1983; Brisbin and Mowbray 2002). Semi-altricial birds, however, are even more vulnerable to nest destruction than both precocial and altricial birds. American Bittern and Least Bittern lay an average of four eggs over four and six days respectively; they incubate for approximately 23 and 20 days, and brood rear for a minimum of seven and six days (nest exposure times of 34 and 32 days respectively) (Peck and James 1983; Gibbs *et al.* 1992b; Davis and Kushlan 1994). This may explain why nest flooding, or collapse, are more common causes of nest failure in semi-altricial birds, such as Least Bittern (Weller 1961), and precocial birds, such as American Coot (Sooter 1945; Weller 1971) and Common Moorhen (Helm *et al.* 1987), than in the altricial Marsh Wren (Kale 1965) and Swamp Sparrow (Greenberg 1988; Greenberg and Droege 1990). Those birds with longer nest exposure periods (e.g. semi-altricial birds) may become less common and be replaced by less hydrologically vulnerable marsh birds (e.g. altricial birds).

5.1.4 Reproductive Strategy

The hydrological vulnerability of a marsh bird species to climate change is also affected by its maximum population recruitment rate. The ability of a population of marsh bird species to respond to favourable breeding habitat conditions is determined by life history traits, such as clutch size and number of broods per breeding season. In general, a population's intrinsic (annual) rate of growth is calculated by dividing the numbers of births from deaths (Ricklefs 1990). This calculation indicates whether a population is increasing, decreasing, or stable (i.e. if births equal deaths, population is stable). Annual reproductive success within freshwater coastal wetlands may be highly variable, due to inter-annual changes in water levels. However, breeding populations of marsh bird species can be maintained over the long-term provided that the life-time reproductive success of a breeding pair is sufficient to replace themselves within the population.

Population birth and death rates, however, are difficult to obtain for many bird species due to the challenge of tracking and identifying highly mobile, relatively small-bodied animals. This may explain why avian studies primarily use reproductive output (i.e. clutch size and number of broods per year), or long-term monitoring trends, as a "rough" index of population trends. For example, Vance *et al.* (2003) showed that populations of forest breeding birds characterized by high reproductive output (i.e. large clutches and multiple broods) are capable of persistence even with habitat loss or degradation. Population persistence, however, may also be affected by dispersal rates, such as immigration and emigration. Therefore, the size of current regional populations should also be considered when assessing a species' hydrological vulnerability (see Population Trends Section 5.1.5).

Age at first reproduction and average longevity are also life history traits that affect population parameters, such as birth and death rates (Ricklefs 1990), as well as population persistence (Eriksson and Kiviniemi 1999; Fagan *et al.* 2001; Fahrig 2001). Moreover, these traits may differentiate hydrological vulnerabilities among marsh birds that have similar reproductive outputs during a breeding season (i.e. similar number of broods per year and clutch size). For example, Least Bittern, Northern Harrier, and Common Grackle all produce one brood with an average clutch size of 4.5 eggs per breeding season, but have different ages for sexual maturity and maximum lifespan (Peck and James 1983, 1987; Poole and Gill 1992 - ongoing). Therefore, maximum lifetime reproductive output may provide a better index when assessing hydrological vulnerability than maximum annual reproductive output. For example, Least Bittern have a maximum lifetime reproductive output of approximately 27 young given a clutch size of 4.5, two years to reach sexual maturity, and a life expectancy of about eight years (Peck and James 1983; Poole and Gill 1992 - ongoing). Northern Harrier and Common Grackle have a maximum reproductive output of approximately 18 and 13.5 young during their lifespan, respectively (Section 5.2; Table 5.2). Birds that produce only one, relatively small clutch per year, and are short-lived, will be more vulnerable to hydrological changes (i.e. prolonged unfavourable breeding conditions) than birds that either produce many young during a short life, or a few young each year but live a very long life.

5.1.5 Population Trends

The hydrological vulnerability of a marsh bird species is also affected by current population size. Small populations, such as species at risk, have a higher risk of extinction due to demographic and environmental stochasticity than large populations (Pimm *et al.* 1988; Caughley and Gunn 1996). For example, a chance event, such as increased nest destruction during the breeding season because of flooding (i.e. environmental stochasticity), affects a small population more than a large population because there are fewer individuals to buffer, or compensate, any potential reduction in reproductive success. Furthermore, small more isolated populations have reduced rates of immigration and emigration which may affect population persistence, or viability. Generally, the long-term viability of a population that is not producing enough young to maintain itself (sink population) is reliant on immigration from populations producing surplus young (source population) (Ricklefs 1990). Consequently, even without changes in marsh habitat, different hydrological vulnerabilities may exist between regional populations of marsh bird species because of different immigration and/or emigration rates.

Currently, COSEWIC (2003) has identified Least Bittern and King Rail as threatened and endangered species, respectively, because of their small population sizes. In addition, Marsh Monitoring Program survey results suggest that many other populations of marsh bird species, such as Black Tern, Common Moorhen, Pied-billed Grebe, and Sora are declining within selected areas of Great Lakes coastal wetlands (Timmermans 2001). Consequently, modification of the hydrological regime, due to climate change, may affect the Great Lakes populations of Least Bittern, King Rail, Black Tern, Common Moorhen, Pied-billed Grebe, Sora, and other small, or declining, coastal marsh bird populations more than populations that are increasing or stable (e.g. Canada Goose and Common Grackle). This may result in reduced distribution and numbers of some marsh bird populations, and loss of some species completely.

5.1.6 Other Potential Effects of Climate Change on Marsh Birds in Great Lakes Coastal Wetlands

Great Lakes coastal wetlands provide some of the most important staging habitat in southern Canada for migrating birds (Hummel 1981; Dennis *et al.* 1984; Prince *et al.* 1992; Petrie 1998). For example, up to 30,000 Tundra Swans and up to 8% of the world's Canvasbacks migrate through Lake Erie coastal wetlands (Petrie 1998). Spring and fall bird migrations are energetically demanding, and many migrating marsh birds forage extensively on submerged and emergent aquatic vegetation within coastal wetlands during migration stopovers (Thayer *et al.* 1984; Petrie 1998; Knapton and Petrie 1999; Badzinski 2003). Any potential changes in aquatic plant availability may affect many populations of marsh bird species.

Despite their ecological importance for many marsh birds, Great Lakes coastal wetlands continue to be compromised or lost because of anthropogenic stressors and/or coastal development (Herdendorf 1987, 1992; Snell 1987). As many wetlands become degraded and/or unsuitable for marsh birds, remaining coastal wetlands become even more important habitat for migratory birds (Petrie 1998). Although coastal wetland loss has likely contributed to large concentrations of waterfowl using the remaining wetlands, environmental factors, such as warmer spring, fall, and winter temperatures, due to climate change, may also affect avian migration patterns. For example, some birds may migrate earlier in spring (Mason 1995; Butler 2003; Hussell 2003) or shift into more northerly habitats during winter (Thomas and Lennon 1999; Petrie and Francis 2003). These temporal and geographical shifts in migration patterns may affect coastal wetland habitat as more waterfowl and other marsh birds use these habitats.

Large congregations of waterfowl and other wetland birds can deplete wetland vegetation (Anderson and Low 1976; Giroux and Bedard 1987; Beekman *et al.* 1991; Evers *et al.* 1998; Idestam-Almquist 1998; Petrie and Francis 2003) and in some habitats, alter species composition of vegetation communities (Jefferies *et al.* 1994; Kotanen and Jefferies 1997). It is possible that aquatic plant communities in Great Lakes coastal wetlands may be altered by increasing populations of staging and wintering waterfowl. These changes may affect the carrying capacity of these wetlands, and hence, the reproductive success of many marsh birds as the quality of important nesting and foraging habitat declines.

Warmer winter temperatures, due to climate change, may also result in the growth and expansion of some wildlife populations (Kling *et al.* 2003; Petrie and Francis 2003). Currently, Virginia opossum (*Deldephidia virginianus*) and exotic Mute Swan (*Cygnus olor*) are restricted to southern Ontario because cold winter temperatures result in high winter mortality (Kurta 1995; Ciaranca *et al.* 1997). Warmer winter temperatures may benefit these wildlife and other important nest predators including northern raccoon (*Procyon lotor*) and striped skunk (*Mephitis mephitis*) which could negatively affect many marsh breeding birds because of increased nest loss due to predation. Ground nesting marsh birds (e.g. Mallard, Common Snipe, and Northern Harrier) (MacWhirter and Bildstein 1996; Sovada *et al.* 2001; Drilling *et al.* 2002) and marsh birds susceptible to nest stranding (e.g. Least Bittern) (Post 1998) are particularly vulnerable to nest predation from these mammals. Furthermore, an increasing population of Mute Swans may negatively affect many marsh bird species, particularly waterbirds (e.g. bitterns, waterfowl, terns), because of reduced food availability (e.g. submerged aquatic vegetation) and increased interspecific competition for wetland habitat (Petrie and Francis 2003).

5.2 HYDROLOGICAL VULNERABILITY INDEX

Hydrological vulnerability indices were developed for selected Great Lakes coastal marsh birds using survey data of marsh bird communities collected for the IJC LOSLR Study (DesGranges *et al.* 2005). The index scores were used to rank the hydrological vulnerability of marsh bird species and nesting guilds. The *Breeding Birds of Ontario Nidiology and Distribution* (Peck and James 1983, 1987) was used to select key marsh birds that fit one of the following criteria:

- Nest in Great Lakes coastal marshes, and
- Forage primarily in coastal marsh vegetation (e.g. open emergent marsh, emergent marsh, or meadow marsh vegetation) while brood rearing.

Indices were calculated for each key marsh bird by summing vulnerability scores for several breeding habitat requirements, life history traits, and population trends/status variables. Species habitat requirements were determined from nidiological data provided by *Breeding Birds of Ontario Nidiology and Distribution* using Peck and James (1983, 1987) and/or Steen *et al.* (2005). Breeding habitat requirements were divided into: Marsh Nesting Requirements; Nest Habitat Availability (vegetation); Nest Suitability (nest location); and Foraging Habitat Availability/Suitability (vegetation). Only coastal wetland habitat that was likely to be impacted by projected lake water level changes was included (i.e. from open emergent marsh to transitional meadow marsh/shrub). Life history traits (average clutch size, length of incubation and nestling period, age at first reproduction and average longevity) were obtained from Peck and James (1983, 1987) and/or Poole and Gill (1992 - ongoing). When number of days to lay a clutch was not given, it was assumed that egg laying required a minimum of one day per egg. Total exposure time of nests to flooding was determined by summing the total number of days to complete an average sized clutch, and to incubate and brood rear. Population trends were obtained from Bird Studies Canada - Marsh Monitoring Program (Timmermans 2001). Population status was obtained from COSEWIC (COSEWIC 2003). Habitat requirements, life history traits, and population trends/status were categorized and rated in relation to projected hydrological changes to develop the vulnerability index. Table 5.1 outlines the vulnerability index components and their scoring.

Table 5.1 Codes and hydrological vulnerability weightings for selected marsh bird species breeding in coastal wetlands on the lower Great Lakes

Habitat Use of Marsh Breeding Birds	Hydrological Vulnerability Score
Marsh Dependency	
Marsh nesting obligate bird	8
Facultative marsh nesting bird	0
Nesting Habitat	
OEM = Open emergent marsh	15
EMW = Emergent marsh, standing water required	13
MMW = Meadow marsh, standing water required	11
EMD = Emergent marsh, standing water not required	8
MMD = Meadow marsh, standing water not required	6
MMT = Meadow marsh transition with shrubs, standing water not required	4
TRS = Treed swamp	1
Nest Location	
OPL = Over water, platform nest (may be anchored or free floating)	15
OE<15 = Over water, elevated nest less than 15 cm from water	13
DGR = No standing water, nest on ground	11
OE15-60 = Over water, elevated nest between 15 and 60 cm	8
DE<15 = No standing water, elevated nest less than 15 cm from ground	6
DE15-60 = No standing water, elevated nest between 15 and 60 cm	4
OE>60 = Over water, elevated nest > 60 cm from water	2
DE>60 = No standing water, elevated nest > 60 cm from ground	1
Foraging Habitat	
OEM = Open emergent marsh with submerged aquatic vegetation	5
EMW = Emergent marsh with standing water	4
EMD = Emergent marsh without standing water	3
MM = Meadow marsh	2
MMT = Meadow marsh transition with shrubs	1
Nest Exposure	
High Nest Exposure = nest exposure > 44 days	16
Moderate Nest Exposure = nest exposure 35 < x ≤ 44 days	12
Low Nest Exposure = nest exposure 26 < x ≤ 35 days	8
Minimal Nest Exposure = nest exposure 17 < x ≤ 26 days	4
None = ≤ 17 days	0
Reproductive Strategy	
Low Reproductive Output over Lifespan ("r" < 18)	10
Moderate Reproductive Output over Lifespan (18 ≤ "r" < 36)	5
High Reproductive Output over Lifespan ("r" ≥ 36)	0
Population Trend	
COSE = COSEWIC status	8
POD2 = Population decreasing (significant)	6
PODN = Population decreasing (non-significant)	4
POST = Population stable	2
POIN = Population increasing	0

Maximum possible score (77) and minimum score (3)

5.2.1 Marsh Dependency

Marsh birds were grouped according to marsh nesting dependency. Marsh nesting obligate birds were defined as those birds that exclusively rely on marshes for nesting. These birds were ranked most vulnerable to projected hydrological changes because of their dependency on coastal wetland habitat. Facultative marsh nesting birds received a score of zero due to their nesting flexibility.

5.2.2 Nest Habitat Availability

Nest habitat availability was divided into seven categories based on vegetation composition and presence of water (Table 5.1). Wetland birds requiring Open Emergent Marsh (OEM) were considered most vulnerable to hydrological changes, due to climate change, because lower water levels would likely affect interspersion of aquatic plants and open water. Emergent Marsh standing Water required (EMW), and Meadow Marsh standing Water required (MMW), were weighted second and third because of the dependency of marsh birds using these habitats for standing water. EMW was weighted higher than MMW because lower water levels would likely result in the expansion of meadow marsh and a possible decline of emergent plants. Consequently, birds nesting in dry Meadow Marsh (MMD) and transitional Meadow Marsh (MMT) would likely be less vulnerable to hydrological changes due to habitat expansion. Transitional Meadow Marsh and Treed Swamp (TRS) were included as coastal wetland habitats because some marsh foraging birds, such as waterfowl and herons, nest in these habitats and rely on coastal wetlands for brood rearing. These habitats, however, were ranked lowest due to their low reliance on lake water levels.

5.2.3 Nest Location

A third vulnerability of marsh breeding birds to altered hydrology was nest flooding, dislodgement, or stranding. Marsh breeding birds were categorized into eight levels based on nest location (Tables 5.1, 5.2). Over water nesters and birds nesting closer to the substrate were ranked more vulnerable to flooding than tree nesters. Over water, Platform nesters (OPL) were ranked highest because nest flooding, or nest break-up and dislodgment due to flooding, were common reasons for nest failure in these birds. Although some marsh birds build slightly elevated nests above the water (e.g. on vegetation mounds, or hummocks), these birds were considered platform nesters because the nest was not directly interwoven with, or supported by surrounding vegetation. Other ground nesting marsh birds (e.g. Canada Goose) were considered elevated nesters because of their “typical” high nest location above the water (e.g. on muskrat lodges). Over water, slightly elevated nests (< 15 cm) (OE<15) were ranked higher than No standing water, Ground nesters (DGR) because of their closer proximity to the lake. DGR nesters were more vulnerable than over water, elevated 15-60 cm (OE15-60) nesters because of the potential for nest destruction from flash floods. All birds nesting 60 cm or more above substrates were considered the least vulnerable to hydrological changes due to climate change.

5.2.4 Foraging Habitat Availability

Some marsh birds depend on coastal wetland habitat for brood rearing but not nesting (Bellrose 1980; Poole and Gill 1992 - ongoing). Adults and foraging broods were also more likely to depend on spatial and temporal patterns in insect emergence than on wetland vegetation (Strehl and White 1986; Whittingham and Robertson 1994). Therefore, foraging habitat availability was weighted lower than nesting habitat requirements (e.g. marsh dependency, nest habitat availability, and nest suitability). Wetland birds requiring open emergent marsh and emergent marsh with standing water for foraging habitat were considered more vulnerable to hydrological changes than those birds requiring other habitats because of the presence of water and their proximity to the lake. Consequently, altered hydrology may influence the emergence of aquatic insects and plants in these habitats.

5.2.5 Nest Exposure

Hydrological vulnerability was also related to total exposure time of eggs and/or chicks while in the nest. This exposure period was determined by clutch size and length of incubation and nestling periods. Total nest exposure to flooding was calculated by summing the number of days required to lay an average clutch size and maximum range values for incubation and nestling periods. Hydrological rankings were then based on length of nest exposure time (i.e. birds with larger clutches, longer incubation and nestling periods were ranked with the highest vulnerability). Those birds nesting more than 60 cm from the substrate (either water or ground) were scored zero because these nests are not affected by hydrological changes.

Table 5.2 Breeding habitat requirements, life history traits, and population size in relation to hydrological rankings for selected marsh birds breeding in lower Great Lakes coastal wetlands (see Table 5.1 for codes and scoring)

Common Name	Scientific Name	Marsh Dependency	Nesting Habitat		Nest Location		Foraging Habitat		Nest Exposure						Reproductive Strategy								Population Trend		Hydrological Vulnerability
		Rank	Code	Rank	Code	Rank	Code	Rank	Average clutch size	Days required for egg-laying	Days required for incubation	Days required for fledging	Total Nest Exposure	Rank	Number of broods per yr	Annual Reproduction (# of broods × average clutch)	Age at First Reproduction	Reproduction in 3 yrs	Lifespan Code	Lifespan Rank	Lifetime "r"	Rank	Code	Rank	
Forster's Tern	*Sterna forsteri*	8	OEM	15	OPL	15	OEM	5	3	3	25.3	7	35	12	1	3	2	6	short	1	6	10	POIN	2	**67**
Pied-billed Grebe	*Podilymbus podiceps*	8	OEM	15	OPL	15	OEM	5	6.5	8	27	1	36	12	1	6.5	1	19.5	short	1	19.5	5	PODE	6	**66**
Black Tern	*Chlidonias niger*	8	OEM	15	OPL	15	OEM	5	3	4	21	1	26	4	1	3	2	6	short	1	6	10	PODE	6	**63**
King Rail	*Rallus elegans*	8	EMW	13	OPL	15	EMW	4	9	9	22	1	32	8	1	9	1	27	short	1	27	5	COSE	8	**61**
American Coot	*Fulica americana*	8	OEM	15	OPL	15	OEM	5	7	7	24	2	33	8	1	7	1	21	short	1	21	5	PODN	4	**60**
Common Moorhen	*Gallinula chloropus*	8	OEM	15	OPL	15	OEM	5	7.5	7.5	22	1	31	8	2	15	1	45	short	1	45	0	PODE	6	**57**
Sora	*Porzana carolina*	8	EMW	13	OPL	15	OEM	5	9	9	20	4	33	8	2	18	1	54	short	1	54	0	PODE	6	**55**
Least Bittern	*Ixobrychus exilis*	8	EMW	13	OE15-60	8	OEM	5	4.5	6	20	6	32	8	1	4.5	2	9	long	3	27	5	COSE	8	**55**
Virginia Rail	*Rallus limicola*	8	EMW	13	OPL	15	EMW	4	8.5	8.5	20	4	33	8	2	17	1	51	short	1	51	0	PODN	4	**52**
American Bittern	*Botaurus lentiginosus*	8	EMW	13	OE15-60	8	OEM	5	4	4	23	7	34	8	1	4	2	8	long	3	24	5	PODN	4	**51**
Swamp Sparrow	*Melospiza georgiana*	8	MMW	11	OE<15	13	MM	2	4.5	4.5	14	11	30	8	2	9	1	27	short	1	27	5	PODN	2	**49**
Marsh Wren	*Cistothorus palustris*	8	EMW	13	OE>60	2	EMW	4	5.5	5.5	15	15	36	12	2	11	1	33	short	1	33	5	PODN	4	**48**
Canada Goose	*Branta canadensis*	0	OEM	15	OE15-60	8	OEM	5	5	7.5	28	1	37	12	1	5	2	10	long	3	30	5	POIN	2	**47**
Sandhill Crane	*Grus canadensis*	0	MMT	4	DGR	11	OEM	5	2	2	28		30	8	1	2	2	4	long	3	12	10	PODN	4	**42**
Mallard	*Anas platyrhynchos*	0	MMD	6	DGR	11	OEM	5	9	9	28	1	38	12	1	9	1	27	short	1	27	5	POIN	2	**41**
Killdeer	*Charadrius vociferus*	0	MMD	6	DGR	11	MM	2	4	5.5	30	1	37	12	1	4	1	12	short	1	12	10		0	**41**
Northern Harrier	*Circus cyaneus*	0	MMD	6	DGR	11	MMT	1	4.5	9	32	14	55	16	1	4.5	2	9	moderate	2	18	5		0	**39**
Northern Shoveler	*Anas clypeata*	0	MMD	6	DGR	11	OEM	5	10.5	10.5	25	1	37	12	1	10.5	1	31.5	short	1	31.5	5		0	**39**
Gadwall	*Anas strepera*	0	MMD	6	DGR	11	OEM	5	9	9	26	1.5	37	12	1	9	1	27	short	1	27	5		0	**39**
American Black Duck	*Anas rubripes*	0	MMD	6	DGR	11	OEM	5	9	9	26	1	36	12	1	9	1	27	short	1	27	5		0	**39**
Common Snipe	*Gallinago gallinago*	0	MMD	6	DGR	11	MM	2	4	4	20	1	25	4	1	4	1	12	short	1	12	10	PODN	4	**37**
Le Conte's Sparrow	*Ammodramus leconteii*	0	MMD	6	DGR	11	MM	2	4.5	4.5	13	7	25	4	1	4.5	1	13.5	short	1	13.5	10		0	**33**
Spotted Sandpiper	*Actitis macularia*	0	MMD	6	DGR	11	MM	2	4	4	20	1	25	4	1	4	1	12	short	1	12	10		0	**33**
Wilson's Phalarope	*Phalaropus tricolor*	0	MMD	6	DGR	11	MM	2	4	4	27	1	32	8	2	8	1	24	short	1	24	5		0	**32**
Red-winged Blackbird	*Agelaius phoeniceus*	0	EMD	8	DE>60	1	EMW	4	4	4	13	11	28	0	2	8	1	24	short	1	24	5	PODN	4	**22**
Tree Swallow	*Tachycineta bicolor*	0	TRS	1	DE>60	1	OEM	5	5.5	5.5	14	22	42	0	1	5.5	1	16.5	short	1	16.5	10	PODN	4	**21**
Common Grackle	*Quiscalus quiscula*	0	MMT	4	DE>60	1	EMD	3	4.5	4.5	14	15	34	0	1	4.5	1	13.5	short	1	13.5	10	POIN	2	**20**
Alder Flycatcher	*Empidonax alnorum*	0	MMT	4	DE>60	1	MMT	1	3.5	3.5	11	14	29	0	1	3.5	1	10.5	short	1	10.5	10	POIN	2	**18**
Great Blue Heron	*Ardea herodius*	0	TRS	1	DE>60	1	OEM	5	4	8	27	81	116	0	1	4	2	8	long	3	24	5		0	**12**
Black-crowned Night-Heron	*Nycticorax nycticorax*	0	TRS	1	DE>60	1	OEM	5	3.5	7	23	14	44	0	1	3.5	2	7	long	3	21	5		0	**12**
Wood Duck	*Aix sponsa*	0	TRS	1	DE>60	1	OEM	5	11.5	11.5	29	1	42	0	1	11.5	1	34.5	short	1	34.5	5		0	**12**
Green Heron	*Butorides virescens*	0	TRS	1	DE>60	1	OEM	5	4	6	22	8	36	0	1	4	2	8	long	3	24	5		0	**12**

(+) Marsh breeding bird community only includes marsh birds breeding in Great Lakes coastal wetlands (based on Peck and James 1983, 1987)

5.2.6 Reproductive Strategy

The ability of a species to respond to, or recover from, environmental change was related to its intrinsic rate of population growth (or recruitment rate). Therefore, the hydrological vulnerability of a marsh bird species was related to life history traits, such as clutch size, number of broods per year, age at first reproduction, and lifespan (longevity). Although most northern temperate birds only raise one brood during a breeding season, some birds (e.g. Marsh Wren, Red-winged Blackbird, Swamp Sparrow) can raise two broods during a season. Consequently, to standardize maximum reproductive output for these birds, average clutch size was multiplied by number of broods per year. Reproductive strategies were then categorized into three hydrological vulnerabilities based upon maximum reproductive output in relation to age at first reproduction and lifespan. Birds that had the potential to produce many offspring at a young age were considered least vulnerable to hydrological change. Those birds requiring more than one year to mature and/or laying small clutches that were single brooded and short-lived were considered most vulnerable to water level changes.

5.2.7 Population Trend

Marsh birds identified as COSEWIC species were considered most vulnerable to hydrological changes. Bird populations that were documented within the Marsh Monitoring Program (Timmermans 2001) as declining significantly both annually and over a five-year period (1995-2000) were ranked second most vulnerable. Bird species with populations showing a non-significant declining trend were ranked third. Finally, those populations documented as either stable, or increasing, were ranked less vulnerable and not vulnerable to projected hydrological changes, respectively.

From the species-specific scores for these factors, an overall hydrological vulnerability index was calculated and plotted for each marsh bird species (Table 5.2; Figure 5.2). Risk categories were assigned based on defined plateaus and inflection points. Birds with scores between 49 and 65 were identified as “High Risk” species most vulnerable to projected hydrologic alterations due to climate change. “Moderate Risk” species had scores between 30 and 48, and “Low Risk” species were between 12 and 29.

5.2.8 Summary

Obligate wetland breeding bird species, with nesting and foraging preferences that require specific hydrologic conditions, were identified as most vulnerable to changing hydrology due to climate change. Species such as Forster’s and Black Tern, Pied-billed Grebe, rails, and bitterns nest on or near the water surface, and have extended nest exposure periods making these species particularly sensitive to increased hydrologic variability during the breeding season. The requirement for prolonged, relatively stable water levels during the breeding season was identified as a factor most vulnerable to changing hydrology due to climate change. Many of these species were also COSEWIC-listed or had declining population trends, indicating that these species were already under stress within the Great Lakes basin. Stabilizing these regional breeding populations would be especially difficult with the potential for added stress to emergent marsh habitats, and increased variability in long-term habitat supply due to climate change.

5.3 Bird Community Modelling

As is evident from the vulnerability assessment, there is a strong relationship between “High Risk” bird species and the requirement for flooded emergent marsh habitat. In particular, all marsh nesting obligates, with the exception of Swamp Sparrow, require emergent marsh habitat with standing water. In addition, DesGranges *et al.* (2005) documented that breeding densities of several emergent marsh bird species were correlated with breeding season water depth in Lake Ontario and St. Lawrence River wetlands. Consequently, development of models to predict potential impacts of climate change on wetland bird communities focused on marsh birds that nest within emergent marsh wetland habitat.

5.3.1 Model Development

Bird modelling incorporated bird survey data collected on Lakes Ontario, Erie, and St. Clair for the dyked/undyked wetland comparison (see Chapter 8) which primarily investigated the effects of habitat change, due to climate, on marsh birds. In addition, bird data from the IJC LOSLR Study (DeGranges *et al.*

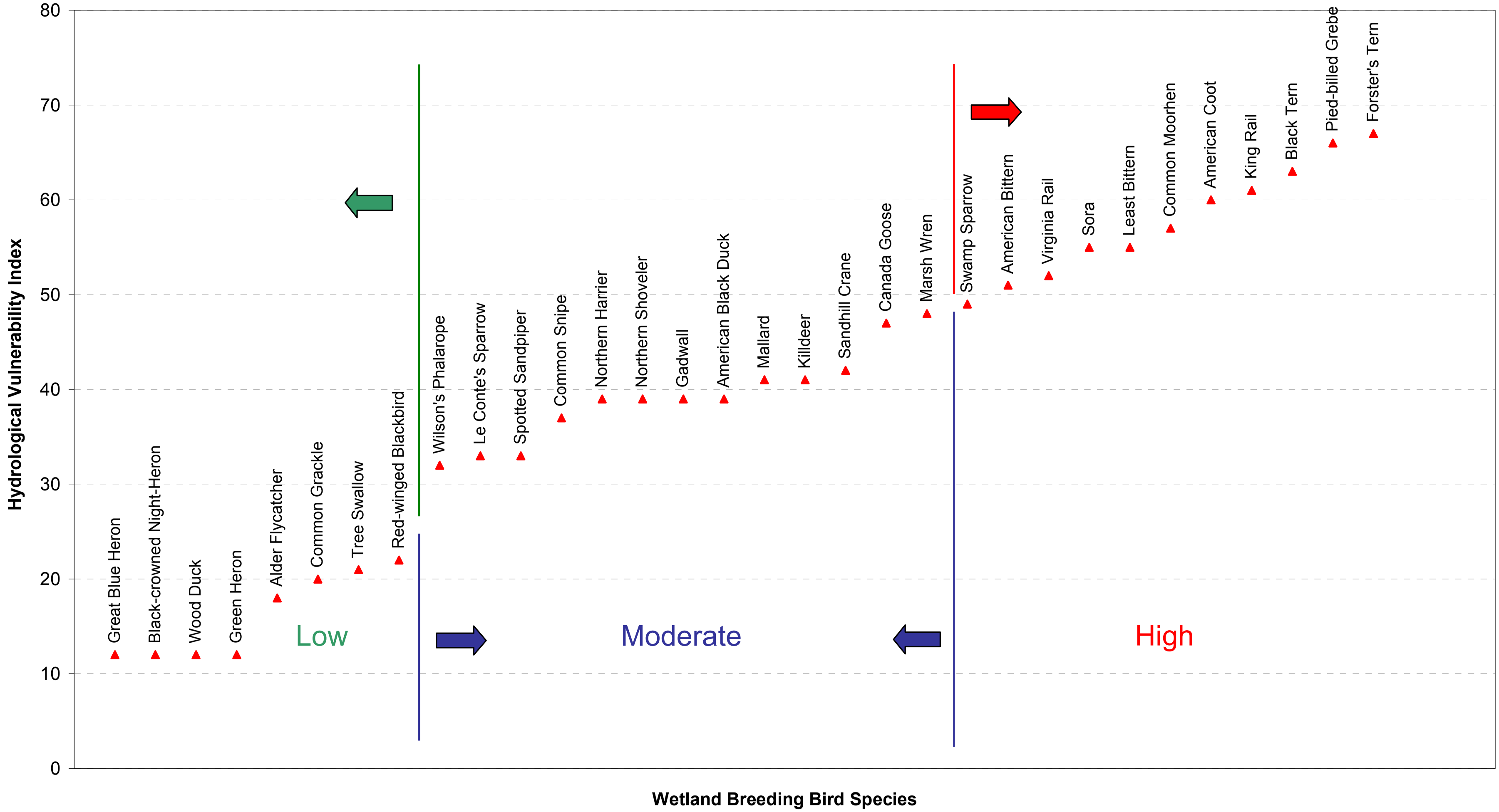

Figure 5.2 Hydrological vulnerability indices for selected marsh birds breeding in coastal wetlands on the lower Great Lakes (based on scores from Table 5.2)

2005) was used to supplement the emergent marsh data and evaluate potential effects of climate change on meadow marsh and treed/shrub bird species. Within emergent marsh habitat, breeding bird species were split into two sub-groups, marsh nesting obligates and marsh nesting generalists. Abundance indices were developed for each wetland breeding bird guild, using density estimates of representative species within each guild (Table 5.3). To standardize values, an index of abundance was calculated by using the maximum observed density for each bird sample station for each marsh user nesting category.

Table 5.3 List of bird species by wetland nesting habitat

Emergent Marsh Nesting Obligates	Emergent Marsh Nesting Generalists	Meadow Marsh Nesters	Treed/Shrub Nesters
Pied-billed Grebe	Red-winged Blackbird	Swamp Sparrow	Yellow Warbler
American Coot	Yellow-headed Blackbird	Common Snipe	Song Sparrow
Common Moorhen	Common Yellowthroat*	Savannah Sparrow	Common Yellowthroat*
Black Tern	Common Grackle	Sedge Wren	Great Crested Flycatcher
Forster's Tern	Mute Swan	Common Yellowthroat*	Willow Flycatcher
Least Bittern	Sand Hill Crane		Least Flycatcher
American Bittern			Alder Flycatcher
Virginia Rail			
Sora			

* Common Yellowthroat was included in three nesting guilds because IJC LOSLR Study data indicated that this bird species nested at similar densities in these wetland habitats

DesGranges *et al.* (2005) also reported a significant correlation between water depth within emergent marsh habitat and breeding bird densities for several emergent marsh bird species. In order to account for this interaction within the climate change scenario modelling, breeding bird abundance index values based upon observed data were plotted against average water depth over the survey period at each sample station. Polynomial regression equations were applied to fit observed data. The regression equations were projected beyond surveyed water depth to the point of crossing zero on the y-axis in order to predict abundance indices for water depths outside of those surveyed (Figure 5.3).

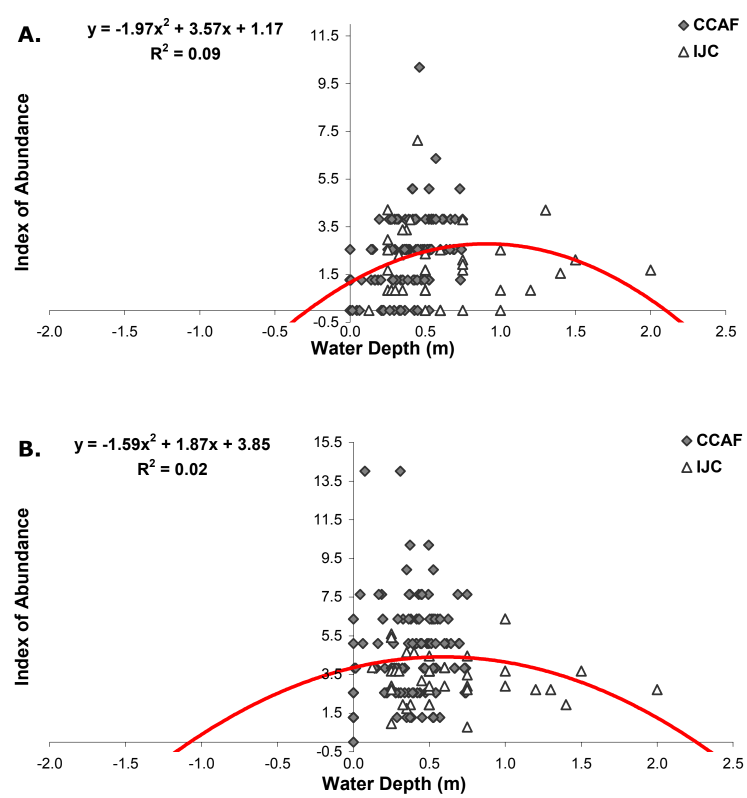

Figure 5.3 Regression equations for emergent marsh nesting obligates (A) and generalists (B) generated from indices of abundance and water depth data collected for the CCIAP and the IJC LOSLR study

Peak abundance indices for marsh nesting obligates occurred at a deeper water depth relative to the generalists, with indices peaking at approximately 1.0 and 0.5 m water depths respectively (Figure 5.3). Conversely, unflooded emergent habitat (i.e. water depth ≤ 0) was estimated to provide limited suitable habitat to obligates compared to the generalist bird guild. The narrower regression equation range between the upper and lower water depth limits for marsh nesting obligates also indicates a stronger sensitivity to water depths relative to the generalists. The water depth regression equations for marsh nesting obligates and generalists explained only a small portion of the total variability associated with the index scores (low R^2, Figure 5.3). Incorporation of additional habitat variables within multiple regression equations increased the associated R^2 values, however, the bird models were limited to incorporating input variables that could be predicted under various climate change scenarios (i.e. vegetation community type and water depth). The abundance-water depth association, as depicted by the regression equations, make biological sense and are supported by published literature. For these reasons, the regression equations were used to predict emergent marsh nesting obligate and generalist abundance indices on a cell basis and complete relative comparisons of bird guild abundance under various climate change scenarios.

DesGranges *et al.* (2005) did not observe a correlation between breeding bird densities and water depths for meadow marsh and tree/shrub bird species, so fixed indices of abundance based upon IJC Lake Ontario bird data were used to evaluate climate change scenarios for meadow marsh (3.7 birds per hectare (ha)) and treed/shrub bird species (5.6 birds per ha).

Wetland nesting bird guilds were modelled using the regression equations or fixed densities generated for each wetland habitat nesting guild in conjunction with predicted area of wetland habitat availability and breeding season water depths for the emergent marsh nesting guilds. The bird model was automated using the *Model Builder* in *ArcGIS 9.0* (ESRI 2004) to define the parameters and model progression. Preliminary water depth and vegetation grids were created from the site DEMs and historic water level data, as outlined in Section 4.3, for each modelled year and scenario. Secondary water depth grids were also created specifically for the emergent marsh bird models with consideration for breeding season influence (Figure 5.4A). The breeding season water level, defined as a spring/summer three-month mean (May, June, and July), was converted to cell-specific breeding season water depth by subtracting the substrate elevation based on the DEM, to create a water depth value for input into the regression equations (Figure 5.3). The regression equations were applied separately to the water depth grid as a cell-based function where emergent marsh was predicted to occur and then corrected for raster cell area (Figure 5.4B). Total estimated abundance index for each nesting community at a wetland study site was calculated by summing the regression outputs for all emergent marsh cells, and multiplying meadow marsh and tree/shrub habitat area predictions with the fixed density bird community values.

In Chapter 7, the wetland bird response model described here was used for an integrated assessment of coastal wetland community response to projected climate change and water level change scenarios in specific wetlands on Lakes Ontario and Erie

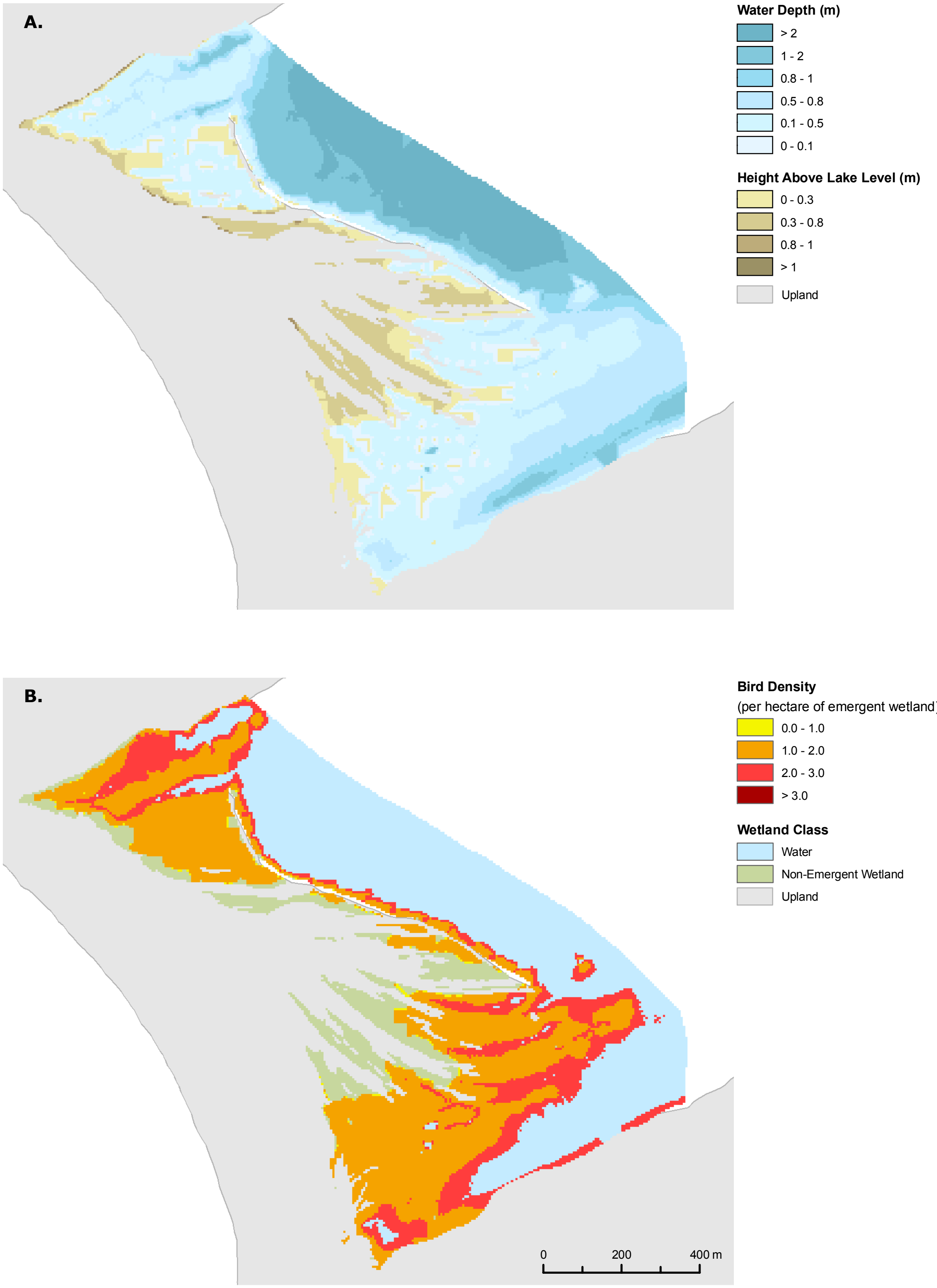

Figure 5.4 Example of secondary breeding season water depth model input grid (A) and modelled index of abundance density grid (B) for marsh nesting obligates for Presqu'ile, 1965

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6.0 Coastal Wetland Fish Community Assessment of Climate Change in the Lower Great Lakes

Susan Doka, Carolyn Bakelaar, and Lynn Bouvier

A literature review, a vulnerability metric, and an assessment model were used to examine the potential vulnerability of fishes to the integrated effects of fish habitat changes in selected coastal wetlands in the lower Great Lakes (Figure 6.1). Climate-induced changes in the Great Lakes region have large-scale implications for fishes that use the coastal zone. Changes in water quantity and quality resulting from altered thermal and precipitation regimes impact nearshore fish habitats through decreases in water levels, vegetation changes, continued land use encroachment in newly dry areas, and potential species invasions (Meyer *et al.* 1999). The thermal changes anticipated in aquatic environments have been well studied but the hydrologic changes, in the context of fish and fish habitat, have not. This assessment focuses on the latter but incorporates thermal changes in the vulnerability assessment of lacustrine fishes.

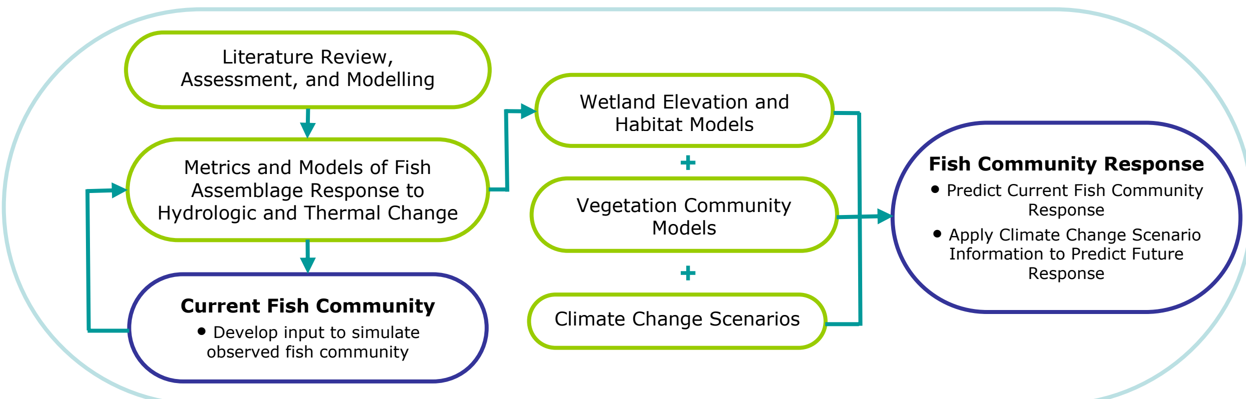


Figure 6.1 Flow diagram of the approach used to evaluate fish vulnerabilities and potential response to climate-induced hydrological and temperature change

6.1 Projected Impacts of Climate Change on Great Lakes Fishes

In the future, warmer temperatures and changes in seasonality may alter lake mixing regimes and the availability of fish thermal habitat. In smaller, mid-latitude lakes and reservoirs, climate warming would result in reduced habitat for many cool and coldwater species (Stefan *et al.* 1996). Thermal habitat area is predicted to increase for warm and coolwater species but decrease for coldwater species, with greater changes in more productive lakes (Stefan *et al.* 1996). Therefore, the invasion of warmwater fishes and extirpations of coldwater species are predicted to increase (Mandrak 1989). Shuter and Post (1990) found that overwinter temperatures restrict the northern distributions of selected fishes and this restriction will be relaxed with climate change. In deep, thermally stratified lakes, including the Great Lakes, winter survival, growth rates, and thermal habitat for most fish generally increase under a double-CO_2 climate scenario (Magnuson and Destasio 1997), but dissolved oxygen (DO) concentrations below the thermocline are also predicted to decrease (Magnuson *et al.* 1990) potentially reducing habitat availability for coldwater species in addition to thermal changes.

Concomitant changes in precipitation patterns (e.g. the magnitude and seasonality of runoff regimes) can alter nutrient loading and change productive capacity, as well as limit habitat availability under low flow conditions. Within drier catchments, dissolved organic carbon (DOC) concentrations are expected to decrease because of reduced runoff resulting in an increase in water clarity, thermocline depth, and a loss in nearshore productivity (Meyer *et al.* 1999). In the Prairies, the loss of pothole wetlands is predicted (Johnson *et al.* 2005) and similar shifts in other wetland and riparian vegetation in the Great Lakes can also be expected due to changing hydrology (Mitsch *et al.* 2001). The hydrologic regime is critical to maintaining coastal habitat diversity (Chapter 4) that in turn dictates nearshore fish habitat usage (Aboul Hosn and Downing 1994; Persson and Crowder 1998; Minns *et al.* 1999) and ultimately potential fish production (Randall and Minns 2002).

Climate is one of the major large-scale influences on fish community structure. Holmgren and Appelberg (2000) stated that benthic fish communities are affected by geographic gradients that are correlated with climatic factors and overall productivity is the most important factor for the classification of species. Regional lake species richness was significantly correlated with postglacial dispersal and climate factors (Mandrak 1995; Minns and Moore 1995).

Most assessments of the response of fishes to climate change consider the influence of alterations to thermal and hydrologic regimes of lakes and rivers. Mohseni *et al.* (2003) projected that hot, dry climate conditions would alter the spatial distribution of lake fishes along temperature gradients within streams, the reproductive success and synchrony of development in lake and stream fishes, and the eventual growth of predatory lake species. The response of fishes to climatic variation appeared to be strongly influenced by the fish's position in the 'hydrologic landscape', especially those dependent on groundwater, which would be most sensitive in their analysis.

King *et al.* (1999) projected that earlier stratification of the water column would change growth patterns of coldwater species (e.g. lake trout (*Salvelinus namaycush*)) and may result in a trade-off in production between epilimnetic and hypolimnetic species but that the interaction was modified by differences in lake morphometry. The timing of stratification and interactions between thermal and feeding habitats spatially accounted for, on average, 44% of variation in fish growth. Climate warming could increase good growth habitat areas for medium and deep lakes by 50-115% for coolwater and warmwater fish guilds but coldwater fish species could experience loss of growth habitat in weakly stratified, medium depth lakes, but a small gain in deep, strongly stratified lakes (Fang *et al.* 1999).

Casselman (2002) correlated historical July and August temperatures to changes in year-class strength for several sport fishes. Smallmouth bass (*Micropterus dolomieu*), a warmwater species, increased in year-class strength with warmer summer temperatures during the first year's growing season, while for northern pike (*Esox lucius*), a coolwater species, year-class strength was curvilinear and negatively related to mid-summer temperatures. Temperatures at spawning were inversely related to year-class strength of two coldwater species; spring temperatures were related to alewife abundance in future generations while fall temperatures were associated with lake trout reproductive success.

Shifts in mortality rates and growth effects are predicted to change due to temperature-related processes. Van Nes *et al.* (2002) showed that climate effects on fisheries were enough to mask fishing mortality effects and would largely be manifested through recruitment variability, which is largely the result of mortality differences in pre-recruitment stages. One contributor to mortality in young fish is winter severity. Fang *et al.* (1999) hypothesized that winterkill will no longer occur in shallow inland lakes because of the decreases in ice cover that are predicted in some regions. This would apply to some shallow coastal areas of the lower Great Lakes.

6.2 Vulnerability Assessment of Nearshore Fishes to Thermal and Coastal Landscape Changes

Fishes are poikilothermic, or cold-blooded, animals. Since temperature plays such an important role in species-specific physiological rates and distributions, fishes are usually assigned to warm, cool, and coldwater thermal guilds based on their adult preferred temperatures. However, of the 119 lacustrine species that are currently found in the lower Great Lakes watersheds (Portt *et al.* 1999; Coker *et al.* 2001; Baldwin *et al.* 2005), most use the warmer nearshore or coastal zone during some part of their life cycle although the adults belong to different thermal guilds. Therefore, it is important to consider both thermal requirements and ontogenetic niche shifts (life-stage specific habitat usage) in an assessment of climate change effects.

Most climate change impacts projected for fish communities are related to changes in the thermal regimes of lakes and river. Yet a neglected area of climate impact assessment is the hydrologic effects on nearshore and coastal vegetation communities and processes due to lowered water levels. The fishes that are most sensitive to changing water levels are species that use the nearshore or coastal zone for all or some part of their life cycle. For example, projections for Great Lakes fishes indicated that changes in optimal thermal space could result in increased competition between coldwater species. However, rapid changes in water levels could adversely affect the littoral zone potentially reducing their efficacy as spawning and nursery areas and resulting in changes in preferred fisheries yields (Meisner *et al.* 1987). This assessment addresses this shortcoming and evaluates the potential effects of decreased water levels, and the concomitant vegetation and fish habitat changes, on fishes in the lower Great Lakes.

6.2.1 Coastal Landscape Factors Affecting Fishes

Landscape in the context of this chapter includes both terrestrial and aquatic "landscapes", especially transitional areas like wetlands. In these areas, water temperature, water levels, water drainage, and circulation patterns could be altered to varying degrees by climate change. The magnitude and direction of change in these variables will differ depending on the climate change scenarios and resultant water level changes (see Chapter 2 for climate model projections) combined with local hydrogeomorphic differences between wetlands. Modification of physical characteristics in nearshore areas directly or indirectly affects fishes that use these habitats. Indirect effects include hydrologically induced changes to vegetation and coastal sediment processes that affect fish cover provided by macrophytes, larger substrates and turbidity.

6.2.1.1 Vegetation

Substantial differences in wetland vegetation can occur with changes in water levels (hydrologic variability; see Chapter 4 for emergent vegetation effects). The cover provided by vegetation, as well as its density, affects reproduction, growth, and mortality through different mechanisms and therefore influences population dynamics by altering these vital rates (Aboul Hosn and Downing 1994; Randall *et al.* 1996; Jeppesen *et al.* 1998). The distribution of submergent vegetation is influenced by water clarity, water depth (or rather the depth of light penetration), the exposure of a site to local wave energy, and substrate type (Wilcox *et al.* 2002).

6.2.1.2 Coastal Sediment Transport (Substrate and Turbidity)

Hydrology-induced changes in water levels and flows can affect coastal processes by changing current patterns due to modifications in water velocity and local basin morphometry and shorelines. Local sediment or substrate types, which are affected by coastal processes, can influence spawning site selection, food availability, and cover from predation (Bergman 1991; Paterson and Whitfield 2000). Storm events and associated coastal sediment transport can also affect local turbidity, which in turn influences egg survival and predator-prey interactions (Miner and Stein 1993).

6.2.1.3 Water Depth

Water levels and flows affect the distribution of fishes and the physical structure of lakes and rivers (Lamouroux *et al.* 1996; Riis and Hawes 2002). These dynamic factors, plus others, interact to create fish habitats of differing suitability depending on the constraints and preferences of individual species (McMahon and Holanov 1995; Minns *et al.* 1999). For example, water levels change the accessibility of different habitats and their suitability because of depth preferences. Changes in accessibility and depth suitability result from an interaction between water levels and local basin morphometry. The area of essential habitat available to different fishes and the overall fish density within those habitats affect population dynamics and the overall productive capacity of a system (Jones *et al.* 1996; Minns *et al.* 1996).

6.2.1.4 Water Temperature

Biologic processes of fishes are regulated by the temperature of their surroundings. Documented thermal effects on fishes are both physiological and behavioural. Examples include movement to preferred temperatures at different life stages (active thermoregulation), thermal cues for spawning (behavioural), and thermal effects on development, growth, and mortality rates (physiological). Many fishes' distributions are thought to be partially determined by upper and lower lethal thermal limits as well as optimal growth and thermal preferenda (Ferguson 1958; Brandt *et al.* 1980). Therefore, knowledge of thermodynamics in different systems is important as temperature can play a key role in structuring fish populations.

Coastal water temperatures are locally regulated by air temperature, hydrologic processes, and the heat budgets of surrounding water bodies (Schertzer 1987). Riverine and groundwater inputs also moderate local temperatures in a coastal area (Mellina *et al.* 2002). Solar radiation and light penetration influence vertical temperature profiles. Wind and waves, as well as current patterns, determine mixing and internal wave dynamics (i.e. seiches). Riparian, emergent, and submergent vegetation influence local temperatures through shading (affecting light penetration) and changing flow patterns. Several of these factors regulating local temperatures are likely to be affected by climate change (McCormick and Fahnenstiel 1999). Most importantly, ambient air and water temperatures will rise.

6.2.2 Aquatic Invasive Species and Species at Risk (Changing Distributions)

The lower Great Lakes are important areas for commercial and sport fisheries and are host to a diverse range of fishes in different habitats (Baldwin *et al*. 2005; Eakins 2005). Several of these species include species at risk (COSEWIC 2005), while some are introduced or invading species (Cudmore-Vokey and Crossman 2000). Non-native species mix with native fishes, common and rare, to form assemblages in different areas at different times depending on life histories. New invasions are predicted by climate change research, which has focused on the potential impact on fish distributions due to increases in temperature. Upper and lower thermal limits for fish populations, implied by geographic distributions and annual average temperatures have been used to predict theoretical future distributions of fishes. For the Great Lakes region, analyses indicate local extinctions of species because of losses in adult thermal habitat and invasions of new species because of thermal range expansions (Mandrak 1989). Changes in fish assemblages in the lower Great Lakes also affect extinction probabilities due to biotic interactions after invasion.

6.2.2.1 Diversity and Wetland Area, Habitat Heterogeneity, and Climate

In addition to temperature, other factors affect the distribution of fishes, their local diversity and richness. Wetlands and other areas of habitat heterogeneity contribute to the overall diversity of aquatic organisms (Benson and Magnuson 1992; Archambault and Bourget 1996). These coastal and nearshore areas are also likely to be affected dramatically by climate change because of the physical changes listed above. The changes in Great Lakes fish habitats in the coastal environment have not been addressed when considering climate change impacts on fish assemblages and diversity, although speculation on the potential effects have been made (Meisner *et al*. 1987).

6.3 Assessment of Hydrologic and Thermal Vulnerability of Fishes

A complete list of fish species found in the lower Great Lakes watersheds was used in a vulnerability assessment of fishes to potential climate-induced changes in the nearshore zone. Vulnerability in this sense implies that particular species would be more sensitive to changes in coastal wetlands; the habitat that would be most affected by water level changes as well as temperature increases. Sensitivity does not infer that changes would be detrimental. The assessment involved developing a metric of sensitivity based on geographic distributions, relative rarity, and habitat requirements at different life stages, including thermal requirements (Table 6.1). A subset of 99 of the 133 riverine and lacustrine fish species found in the lower Great Lakes was derived by excluding riverine and non-native species.

Separate terms in the vulnerability metric were based on characteristics that would make a species relatively more vulnerable to coastal wetland changes across the lower Great Lakes. Species distributions across secondary basins (i.e. individual lakes), as well as the species' rarity (i.e. SAR status), were included as an interactive term in the vulnerability metric. For example, a species at risk was considered more vulnerable to climate change because many SAR are sensitive to disturbance and have restrictive habitat requirements. SAR with more limited distributions were considered more vulnerable to climate change, especially when present in lakes that have greater anticipated water level and thermal changes, like Lake St. Clair.

In addition, thermal and physical habitat requirements by life stage component were included in the vulnerability score. Spawning and adult thermal and coastal habitat requirements were considered in a habitat usage term. All fishes were classified into adult thermal guilds of warm, cool and coldwater species for the assessment (Coker *et al*. 2001). Coldwater species are generally considered to be more susceptible to warmer temperatures under a changed climate (Casselman 2002). Species that spawn in months with greater predicted increases in temperature were considered more vulnerable. For example, these species would likely shift the timing of early life history events if there were thermal spawning cues, or require greater thermal tolerances during the egg stage if spawning is day-length induced.

Table 6.1 Species characteristics and vulnerability scores for lacustrine fishes using Great Lakes coastal wetlands

Species Characteristics	Basic Vulnerability Score
Hydrologic Association	
Lac = Lacustrine	1
Riv = Riverine	0
Species Origin	
Nat = Native	1
Non = Non-native	0
Species Status	
SAR = Species at risk	2
NAR = Not at risk	1
BASIN SCORE	
Great Lakes Basin Distribution	See Section 6.3.1.2 for calculations
LSC = Lake St. Clair only	4.5
LE = Lake Erie only	3.0
LO = Lake Ontario only	2.3
LSC + LE	1.8
LE + LO	1.5
LSC + LO	1.3
All Lakes	1.0
THERMAL SCORE	
Thermal Guild	
Warmwater species	1
Coolwater species	2
Coldwater species	3
Spawning Month	See Section 6.3.1.4
HABITAT SCORE	
Shallow Water Associations	Ratio of < 2m depth use across life stages
0-1 m depth	1 per life stage
1-2 m depth	1 per life stage
2-5 m depth	1 per life stage
5-10 m depth	1 per life stage
> 10 m depth	1 per life stage
Vegetation Associations	Ratio of vegetation use across life stages
Open water	1 per life stage
Submergent	1 per life stage
Emergent	1 per life stage

6.3.1 Vulnerability Score Calculation

The status by basin factor and the thermal and habitat requirement components were weighted equally in the calculation of the vulnerability metric. The individual scores for each of the components of the vulnerability metric are outlined below and in Table 6.1. The different elements of the score were combined in the following manner (Equation 6.1):

Equation 6.1 Vulnerability = (Species Status x Basin Score) + Σ (Thermal Score, Habitat Score)

Theoretical minima and maxima, possible scores for any one particular species, ranged from 4.8 to 36.2. In the calculation of vulnerabilities, the geographic distribution and status of the species were weighted equally with the physical habitat and thermal requirements of each life stage of the fish species.

Any completely riverine species received an automatic score of 0 and were not included in further assessment because they do not use coastal Great Lakes habitats (Lac = 1; Riv = 0). Also, only native species were evaluated in the assessment; any non-native species, whether intentionally introduced or not, was automatically given a score of 0 (Nat = 1; Non = 0) and not included in further assessments.

6.3.1.1 Species Status

Any species listed as a species at risk, either Endangered, Threatened, or Special Concern, received a score double that of more common fishes (SAR = 2; NAR = 1). A species at risk was considered more sensitive to climate change because many SAR are sensitive to disturbances and have restrictive habitat requirements. Also, they may currently be limited by annual temperatures.

6.3.1.2 Great Lakes Basin Distribution and Geography

If a species is only present in one Great Lake, their distribution is restricted, and therefore their vulnerability scores were assigned a higher value. This is especially true in smaller, shallower basins, like Lake St. Clair. Therefore, basin scores were weighted by the size of the basin as well as the number of basins where species were present. The equation used was 4.5/[sum of lake scores]; where Lake St. Clair = 1, Lake Erie = 1.5, and Lake Ontario = 2. Final basin scores are shown in Table 6.1.

6.3.1.3 Combined Status by Basin Score

A species' status was multiplied by its basin score to obtain an overall distributional/geographic vulnerability score. Logically, SAR found in areas more vulnerable to climate change would be more susceptible to change than species that were more common or that cover a larger geographic range in the lower Great Lakes. Effectively, the basin score (a measure of distribution and local geography) doubled for SAR to obtain a combined status by basin score in the overall sensitivity or vulnerability assessment of species. The maximum potential combined score was 9; a SAR found only in the Lake St. Clair basin.

6.3.1.4 Thermal Score

Both spawning times and adult temperature preferences were taken into consideration as part of a thermal vulnerability score for each species. Theoretically, the maximum score for thermal vulnerability was approximately 12 (i.e. achieved by a coldwater species that spawns only in May).

Spawning Temperatures

Based on the recorded spawning months for each species in the final list, a thermal spawning vulnerability score was developed. The average change in monthly water temperatures that was predicted under the warm & wet scenario was calculated using data provided by Croley (2003). The warm & wet scenario (HADCM3 A1FI) had the greatest changes in water temperatures of all the scenarios used in this assessment with the greatest temperature changes in the spring months (Figure 6.2). The range in monthly temperature deviations (ΔT) was 0.08-9.16 °C. The average ΔTs across the documented spawning months for each species were used as the measure of thermal vulnerability to climate change for this life stage.

Month of Year	Month Name	ΔAvgT (°C) BC → WW
1	January	0.11
2	February	0.08
3	March	0.46
4	April	6.26
5	May	9.16
6	June	2.99
7	July	2.87
8	August	4.26
9	September	4.02
10	October	3.58
11	November	3.02
12	December	1.43

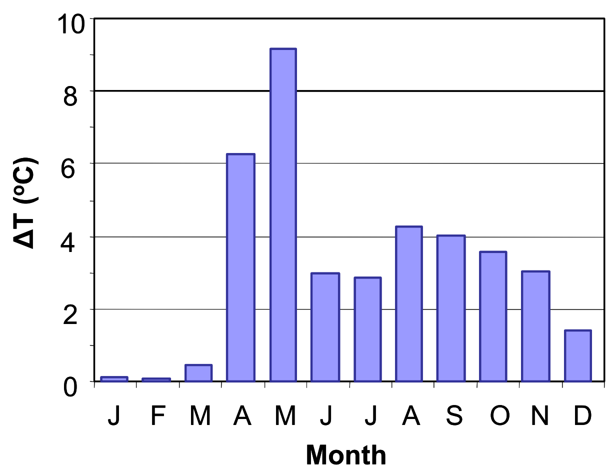


Figure 6.2 Average change in monthly temperatures between base case (BC) and warm & wet (WW) scenarios. Monthly water temperature deviations (ΔT or ΔAvgT; °C) were based on daily water temperatures provided by Croley (2003).

Adult Preferences

The thermal guild assignments of Great Lakes fishes were used to rank each species adult life stage's vulnerability to climate change. Thermal guild assignments were based on adult preferences and because temperatures are projected to increase, in all scenarios, species with lower thermal tolerances (e.g. coldwater species) will be most affected in growth and thermal habitat availability (Casselman 2002). Adult thermal scores were assigned as follows: warm species = 1; cool species = 2; and cold species = 3.

6.3.1.5 Nearshore Habitat Score

Nearshore shallow and vegetated habitat usage scores were added together to assign a nearshore habitat usage score to each species. The higher vulnerability of fish species that use coastal wetlands exclusively was reflected in this component of the overall scoring.

Shallow Water Habitat

Information on depth associations was collated for three life stages (spawning, nursery and juvenile/adult stages) of the 99 species being evaluated (Lane *et al.* 1996a,b,c). Depth associations were categorized into 0-1, 1-2, 2-5, 5-10, and >10 metre intervals by Minns *et al.* (2001). The proportion of shallow water habitat usage (< 2 m depth associations) across all three life stages was used as an indicator of shallow water preference for each species. A maximum score of 6 was assigned to species that only use the 0-1 and 1-2 m depth ranges exclusively. The water level and temperature changes anticipated under climate change will affect this area most directly.

Vegetated Habitat

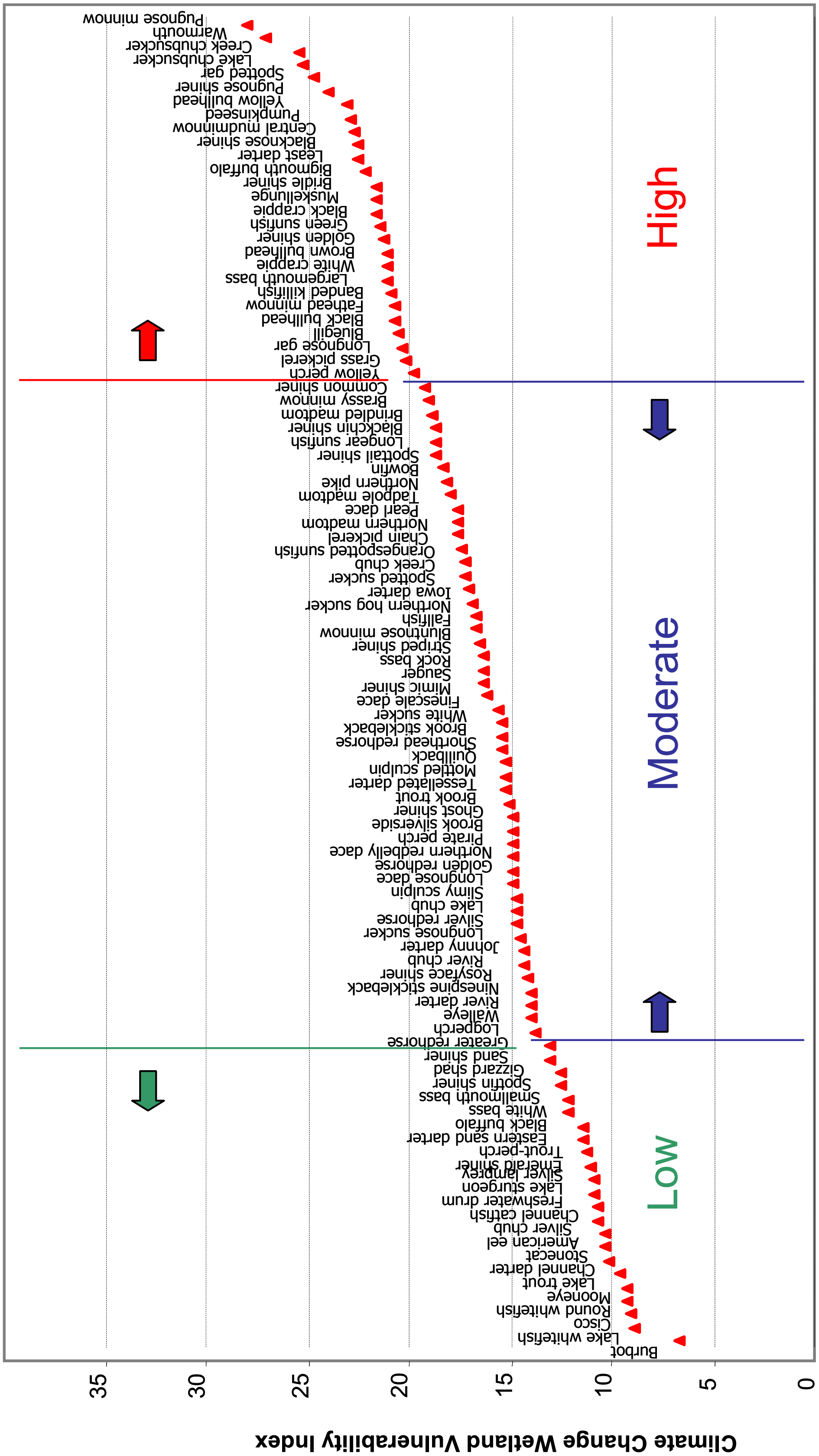
In a similar manner to shallow water habitat usage, a vegetated habitat usage score was developed. All species and their life stages were associated with emergent, submergent and open water habitats (Lane *et al.* 1996a,b,c; Minns *et al.* 2001). The proportion of emergent and submergent habitat usage across all vegetation types, including open water, for the three life stages, determined coastal wetland preference and therefore vulnerability to climate-induced change in these potentially vulnerable habitats (see Chapter 7 for wetland projections). The minimum and maximum vegetation scores were 0 and 10, respectively. A score of 10 was achieved if emergent and submergent vegetation were highly preferred and were the only habitats used by all life stages.

6.3.2 Vulnerability Score Results

The final coastal wetland vulnerability scores for all 99 fish species that used lacustrine habitat for some part of their life cycle ranged from 6.16 for burbot (*Lota lota*) to 27.82 for pugnose minnow (*Opsopoedus emiliae*) (Figure 6.3). The theoretical minimum and maximum vulnerability scores were 4.80 and 36.16, respectively. The theoretical minimum vulnerability score would be achieved by a common, coldwater species present in all the lower Great Lakes basins that spawns in late winter and does not require shallow or vegetated habitat for any of its life stages. The maximum vulnerability score could be obtained by a coolwater SAR found only in the Lake St. Clair basin that spawns in mid-spring and uses shallow, vegetated habitat exclusively for all of its life stages. The distribution of vulnerability scores across the 99 species was primarily linear, with curved tails at the end of the distribution range, especially at the upper end (Figure 6.3). Small natural breaks occurred in the distribution and were used to assign the list of ranked species to high, medium, and low vulnerability categories based on their final scores, as with vegetation and bird species (see Chapters 3 and 5 for results of other vulnerability analyses).

6.3.2.1 Species at Risk (SAR)

Of the 14 fish SAR included in the assessment (Table 6.2) from the original 99 species using lacustrine habitat, vulnerability rankings ranged from the species most sensitive to coastal wetland change, the pugnose minnow (rank 1), to one of the least sensitive species, the channel darter (*Percina copelandi*; rank 93). Based on the categorical assignments to high, medium, and low vulnerabilities, 7 of the 14 SAR ranked in the high risk category; roughly 26% of the high risk species (Table 6.2). Four SAR where assigned to the medium risk group and three species to the low risk category; approximately 8% and 13% of fishes within each respective category. High risk SAR had restricted distributions, but not necessarily restricted just to Lake St. Clair. High risk species were cool and warmwater species with late spring or early summer spawning, with mainly shallow and vegetated preferences. Low risk SAR were generally summer spawners of either cool or warmwater species, they used multiple depths across open and vegetated habitats in different life stages, and tended to be lacustrine species that also use riverine habitats.



Lacustrine Fish Species

Figure 6.3 Final vulnerability scores for 99 fish species that use lacustrine habitats. Vulnerabilities were based on an assessment of climate change risk associated with coastal wetland and thermal preferences for different life stages as well as species distributions.

6.3.2.2 Fisheries Species

Of the nine popular sport fish or commercial fisheries species included in the coastal wetland vulnerability assessment (Table 6.2), rankings ranged from 14 for muskellunge, *Esox masquinongy*, to 98 for lake whitefish, *Coregonus clupeaformis*. Three fisheries species were assigned to each of the risk categories based on final scores. High risk fisheries species were generally coolwater to warmwater and spring- and shallow-spawning species with a vegetated habitat preference. Low risk fisheries species were mainly coldwater, fall spawning species that were associated with deeper, open water habitats, with the exception of smallmouth bass.

Table 6.2 Vulnerability ranks and scores based on climate-induced changes in coastal wetlands and nearshore temperatures of fish species at risk (A) and select fisheries species (B) found in the lower Great Lakes. Maximum rank was 99, which indicates the least vulnerable species. The maximum theoretical vulnerability score was approximately 36. Categorical vulnerability assignments to risk groups were determined by natural breaks in the vulnerability score distribution (see Figure 6.3).

Common Name	Scientific Name	Ranking	Score	Category
A. Species at Risk				
pugnose minnow	*Opsopoeodus emiliae*	1	27.82	High
warmouth	*Lepomis gulosus*	2	26.96	High
lake chubsucker	*Erimyzon sucetta*	4	25.07	High
spotted gar	*Lepisosteus oculatus*	5	24.57	High
pugnose shiner	*Notropis anogenus*	6	23.93	High
bigmouth buffalo	*Ictiobus cyprinellus*	12	21.99	High
bridle shiner	*Notropis bifrenatus*	13	21.59	High
brindled madtom	*Noturus miurus*	30	18.72	Med
orangespotted sunfish	*Lepomis humilis*	40	17.29	Med
spotted sucker	*Minytrema melanops*	42	17.06	Med
eastern sand darter	*Ammocrypta pellucida*	83	11.38	Low
silver chub	*Macrhybopsis storeriana*	90	10.50	Low
channel darter	*Percina copelandi*	93	9.99	Low
B. Fisheries Species				
muskellunge	*Esox masquinongy*	14	21.52	High
largemouth bass	*Micropterus salmoides*	20	20.95	High
yellow perch	*Perca flavescens*	27	19.77	High
northern pike	*Esox lucius*	35	18.01	Med
brook trout	*Salvelinus fontinalis*	58	15.04	Med
walleye	*Sander vitreus*	74	13.93	Med
smallmouth bass	*Micropterus dolomieu*	80	12.33	Low
lake trout	*Salvelinus namaycush*	94	9.54	Low
lake whitefish	*Coregonus clupeaformis*	98	8.74	Low

6.4 Fish Habitat Supply Modelling

The Habitat Alteration Assessment Tool (HAAT, previously called Defensible Methods) was developed by Minns *et al.* (1995, 2001) and Minns and Nairn (1999). It uses an amalgamation of literature-based information compiled by Lane *et al.* (1996a,b,c), Portt *et al.* (1999), and Coker *et al.* (2001) that was input to a database detailing the habitat requirements of fishes during their spawning, young-of-the-year (YOY), and adult life stages. HAAT uses these life stage preferences to estimate habitat suitability values in defined areas for fishes grouped mainly by piscivory and thermal preferences (Minns *et al.* 2001; Minns *et al.* 2006). HAAT was used in this study to estimate suitability values for unique combinations of habitat characteristics and then to estimate the area of suitable habitat in a wetland for a maximum set of six fish guilds, each with three life stages. Guild membership varied between the evaluated sites based on the species sampled in selected coastal wetlands (see Chapter 8 for complete fish species lists across all wetlands).

6.4.1 Application of HAAT (Defensible Methods)

An assessment of suitable habitat supply was undertaken, initially using a theoretical wetland and then specifically chosen wetlands in the lower Great Lakes. The theoretical assessment contrasted the changes that might occur in a closed wetland under a changing climate. The assessment of actual wetlands involved site-specific changes that were modelled based on climate predictions for different Great Lakes. The use of HAAT in these assessments involved several steps that are outlined below.

6.4.1.1 Species Lists by Location and Assessment

A list of species that use the nearshore zone was compiled for the analysis. In an initial, theoretical assessment of climate effects on a generic, closed wetland, the entire fish list from the surveys of lower Great Lakes wetlands was used (see Chapter 8 for survey details).

In the assessment of actual wetlands, site-specific fish lists were used from both 2003 and 2004, spring and fall, surveys at selected locations. The species lists from the dyking study for Long Point, Lake Erie and Mitchell's Bay, Lake St. Clair were used (Chapter 8). Presqu'ile Bay, Lake Ontario, was not part of the dyking study but was situated close to wetlands that were sampled in the eastern outlet basin of the lake. In this case, combined species lists for Amherst Island and Parrott's Bay wetlands were used in the assessment of Presqu'ile Bay.

6.4.1.2 Assignment to Fish Guilds

Fish species on each list were assigned to different guilds. Guild assignments were based on adult thermal preferences (cold, cool, and warmwater species) and adult feeding habits; specifically whether the species was mainly a piscivore or not. Suitabilities and the calculation of suitable areas were guild-based (i.e. aggregate measures of individual species requirements and suitability-weighted areas, respectively). All species within a guild were weighted evenly in the assessments.

6.4.1.3 Fish Habitat Requirements and Suitability Assignments

A defined set of categories were used to describe three physical aspects in each study area:

- water depth (0, 1, 2, 5, 10, 10+ metres);
- substrate type (bedrock, boulder, cobble, rubble, gravel, sand, silt, clay, hardpan clay); and
- vegetative cover (no cover, emergent vegetation, submerged vegetation).

The percent composition, or coverage, of each subcategory for a particular physical characteristic (e.g. vegetative cover) within an area must add to 100 (e.g. a specific area within a wetland can have 25% emergent, 25% submergent and 50% open water, or no cover, equalling 100%). Suitabilities were assigned to each unique area within a wetland based on the physical characteristics of a site and the habitat requirements of each life stage of each species within the guilds (see Appendix 6.1 for an example of input and output data into HAAT).

The computation of composite suitabilities for individual species across life stages, a particular life stage across species within a guild, a specific guild across all life stages, or all guilds across all life stages, is described in detail in Minns *et al.* (2001). Weighted suitable areas (WSAs) were used as aggregate measures of overall suitability for a whole area or wetland. WSA estimates for any of the composite indices were obtained by summing the products of each subarea (patch) weighted by its suitability (Equation 6.2).

Equation 6.2 $WSA_{G,\,LS} = \Sigma\ (A \times S_{G,\,LS})$

Where WSA = weighted suitable area, G = Guild, LS = Life Stage, A = Area, S = Suitability based on physical attributes of the area

It should be noted that equivalent areas of weighted suitable habitat can result from calculations for areas with quite different suitabilities (e.g. a large area with low suitability and a small area with high suitability can have the same WSA). The productive capacity of those habitats should theoretically be equivalent.

6.4.2 Theoretical Assessment of Climate Change Effects Using HAAT

Based on the projected, theoretical changes in isolated coastal wetlands, an assessment of how fish habitat supply would change was conducted under basic hypothetical scenarios. This analysis was preliminary and conducted in early 2004, therefore only 2003 species lists from the wetland surveys were available at the time.

6.4.2.1 Scenarios

The HAAT model was applied to two conditions for a 100-ha theoretical wetland: a present-day baseline condition, and a future condition under a climate change scenario. Under baseline conditions, the wetland was 2 m at maximum depth with the total area divided evenly across two depth categories (0-1 and 1-2 m). The baseline wetland had 50% emergent coverage and 25% submergent cover with a 45% sand, 45% silt, and 10% clay substrate composition throughout. This is typical for most sheltered, small coastal wetlands (Wilcox *et al.* 2002). The depth decreased by 1 m under a climate change scenario but vegetation was assumed to migrate downslope gradually, therefore percent areal coverage in the final 1 m deep, 50-ha wetland was 50% emergent and 50% submergent cover (i.e. no open water). Substrate composition remained the same under the climate scenario conditions.

6.4.2.2 Species Evaluated

The species list used in the preliminary, theoretical assessment of climate-induced changes in wetlands was based on the fish species collected in 2003 coastal wetlands surveys only (Table 6.3). Species were assigned to six thermal and feeding guilds. Very few coldwater species were found, only three species, two of which were introduced. Approximately 20% of the 45 warm and coolwater species were piscivores in their adult stage. Each species within the guilds evaluated were treated equally in the calculation of suitabilities and WSAs, regardless of the number of species within each guild or their vulnerability scores. The percentage of highly vulnerable species, identified in Section 6.3, did vary between guilds. Warmwater non-piscivores and coolwater piscivores had the highest percentage of vulnerable species at 50% and 52%, respectively (i.e. 2 out of 4 and 13 out of 25 spp.). Warmwater piscivores and coolwater non-piscivores in the assessment were comprised of 25% and 36% high vulnerability species (i.e. 1 out of 4 and 4 out of 11 spp., respectively).

Table 6.3 Fish species used in an initial HAAT assessment of climate change effects on an isolated theoretical wetland. Fish species list was compiled from all 2003 surveys in lower Great Lakes as part of the wetland adaptation strategy assessment (Section 8.2.5). Fishes were assigned to guilds based on thermal and piscivory preferences.

Guild	Cold	Cool	Warm
Non-Piscivore (N)	trout-perch	white sucker, banded killifish, brook silverside, greater redhorse, shorthead redhorse, golden shiner, pugnose shiner, emerald shiner, blackchin shiner, spottail shiner, yellow perch, logperch	rock bass, black bullhead, yellow bullhead, brown bullhead, freshwater drum, goldfish, spotfin shiner, common carp, gizzard shad, channel catfish, bigmouth buffalo, green sunfish, pumpkinseed, warmouth, orangespotted sunfish, bluegill, pugnose minnow, white perch, mimic shiner, tadpole madtom, white crappie, bluntnose minnow, fathead minnow, black crappie, central mudminnow
Piscivore (P)	Chinook salmon, brown trout	longnose gar, northern pike, spotted gar, walleye	bowfin, smallmouth bass, largemouth bass, white bass

6.4.2.3 Theoretical Assessment Results

Weighted suitable areas for the baseline and the climate change scenarios of the 100-ha theoretical wetland are shown in Figure 6.4 for the three life stages (spawning, YOY, adult stages) and the six guilds evaluated. The baseline (simulated current conditions) WSAs for spawning habitat availability ranged from 0 to almost 100 ha of the wetland for coldwater piscivores and non-piscivores, respectively. Suitable spawning habitat for warmwater species comprised roughly 60% of the wetland area, regardless of feeding group. Coolwater species had the smallest suitable area at roughly 50 ha under baseline conditions, with equivalent areas available to both feeding groups. All suitable spawning area decreased for all guilds to varying degrees with loss of wetland volume under climate change. Coolwater spawning habitat decreased by 20%, warmwater by 25%, and spawning habitat for trout-perch, the only coldwater non-piscivore, would decrease by roughly 55% under the climate change scenario. The bulk of the species affected by decreases in spawning area would be in the warmwater group.

Suitable nursery or YOY habitat under baseline conditions varied between the guilds but was often 60% or less of the total wetland area. Nursery habitat for all guilds decreased under the lower water levels and habitat changes of the climate scenario. Very little YOY habitat was predicted for coldwater species under baseline conditions. This available nursery area was entirely lost under the climate change scenario, indicating that although the suitability of the wetland was initially low for coldwater species, wetlands that experienced the hypothetical changes tested here would no longer be viable habitat for coldwater YOY. Other guilds lost approximately one-third of the weighted suitable area for nursery habitat between scenarios.

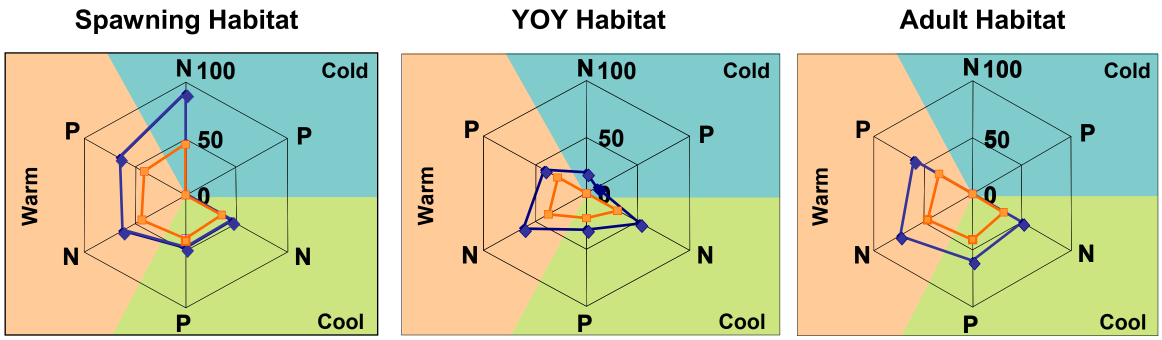


Figure 6.4 Habitat availability for different life stages of fish guilds (see Table 6.1 for list of fishes) in a theoretical, unconnected wetland under baseline (blue line) and climate change (orange line) conditions (see Section 6.4.2.1 for a description of the scenario). Coloured sections correspond to thermal guilds, while N = non-piscivore and P = piscivore guilds.

Suitable adult habitat was only present in the wetland for warm and coolwater species. It ranged from 75% of the area for non-piscivore (NP) warmwater species to 50% of the area for NPs in the coolwater guild. Adult piscivore habitat for both thermal guilds was equivalent under baseline conditions; roughly 60 ha of WSA. Generally, one-third of WSA for warm and cool guilds was lost under the climate change scenario in this isolated wetland habitat. Because one-half of the wetland area was lost between the base case and climate scenarios, this indicated that the suitability of the remaining habitat was higher but the overall loss in area still resulted in a reduction in productive capacity for these fish guilds.

6.4.3 Specific Assessment of Climate Change Effects on Fishes and Fish Habitat in Selected Lower Great Lakes Wetlands

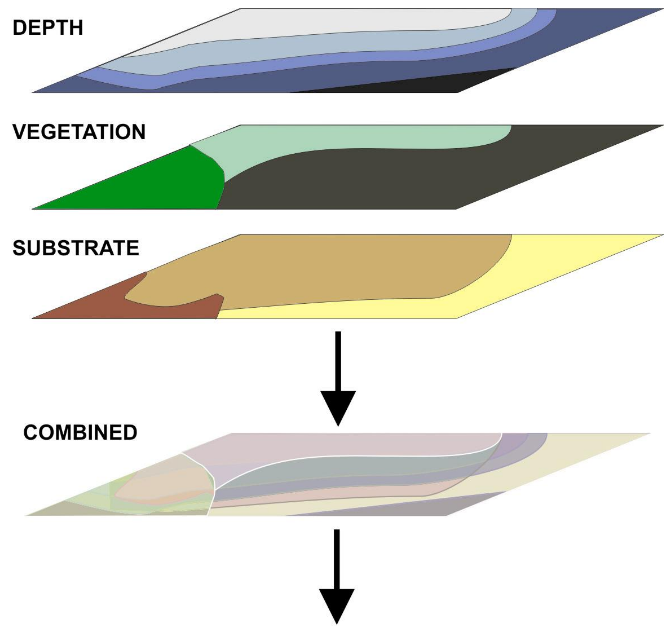
A combination of habitat surveys, habitat models, and suitability modelling based on fish life histories was used to evaluate changes to habitat availability for different fish guilds in selected wetlands in the lower Great Lakes. Fish habitat was evaluated in three study areas: Presqu'ile, Lake Ontario; Long Point, Lake Erie; and Mitchell's Bay, Lake St. Clair. Some habitat surveys were performed in conjunction with CWS, which supplemented existing elevation and substrate data for particular areas, and were also used for validating emergent and submergent vegetation models generated specifically for the study. The scenarios used in fish modelling were restricted to base case (low and high initial conditions) and warm & dry climate scenarios, which differed from those presented in the wetland vegetation results and bird assessment (Chapter 7) because of the additional spatial analysis required for fish habitat modelling (i.e. larger extents and additional variables, like substrate, fetch, and submergent vegetation). The chosen scenarios represent the extremes of water levels in the study.

6.4.3.1 Mapping and Quantifying Fish Habitat Features

A GIS was used to develop three physical attribute layers for each wetland assessed. Depth (metres below water level), substrate type, and vegetation polygons were used as input to the fish habitat model (HAAT; Section 6.4.1). Each layer was generated separately using various data sources and spatial processing before being spatially joined to create a single input file. Files were created for each climate change scenario, to be processed in HAAT, for calculation of suitabilities and habitat areas (Figure 6.5).

The extent of study area for each wetland was determined using elevation boundaries and natural features. At the upper extent, the high water level elevation from the historic or base case scenario was used. An offshore boundary was subjectively chosen, often a natural point or landscape break, which delineated each embayment. Study areas for the fish habitat modelling were larger than the vegetation and bird modelling wetland study areas. The extent of air photographs used in vegetation modelling determined the boundaries for the latter models, which is not necessarily an appropriate boundary for assessing fish habitat because it does not include all the important habitat features of an embayment for the assessment of different life stages.

Habitat layers for each study area were compiled as consistently as possible across sites from various data sources. All layers were standardized to UTM coordinates, with a NAD83 projection. Raster files were



DM_ZVEGSUB_UNQ	SUM_AREA(m2)	DepthRange	VegDensity_DM	Substrate_DM
0,0,0,100,0_0,0,100_0,0,0,0,0,0,54,46,0	101133.0	-5--10	0,0,100	0,0,0,0,0,0,54,46,0
0,0,100,0,0_0,0,100_0,0,0,0,0,0,3,5,0	400086.0	-2--5	0,0,100	0,0,0,0,0,0,3,5,0
0,100,0,0,0_0,0,100_0,0,0,0,0,10,52,38,0	460098.0	-1--2	0,0,100	0,0,0,0,0,10,52,38,0
100,0,0,0,0_0,0,100_0,0,0,0,0,100,0,0,0	647622.0	0--1	0,0,100	0,0,0,0,0,100,0,0,0

Figure 6.5 Schematic of three habitat layers (water depth, vegetation cover, and substrate type) spatially joined to create the input table required for HAAT

generated, using various methodologies, at a resolution of 10-m cell size for each layer and then converted to vector coverages before the final spatial join between layers. Cells (polygons) with similar attributes were lumped into larger polygons when adjacent. (A detailed description of the data layers and the methods used to construct them follows).

Depth

Depth layers were generated for each study area (Presqu'ile, Long Point, and Mitchell's Bay) using a combination of data sources. Elevation raster files were provided for some areas by study partners (CWS and AIRD). Land elevation data were acquired from the OMNR (OMNR 2003). Additional bathymetric data were extracted from NOAA's 1-m depth contour files for Lakes Ontario, Erie, and St. Clair (NOAA 1999, 2001). Depth contour values were converted to elevations in metres above sea level (m asl) using IGLD85. New depth contours were created for each of the water level scenarios based on adjustments to the IGLD85 elevations.

Each water level scenario used March to November (growing season) means for the scenario year being tested by using either historic water levels or adjusted levels based on climate scenarios (Table 6.4). Elevations above the mean water level were considered dry land and are not included in the fish modelling extents for that scenario. Elevations below mean water level were converted to water depth in metres below 0; the current scenario's water level elevation. Figures 6.6, 6.7, and 6.8 show the contours of the shorelines generated for each wetland from the elevation maps for each scenario. Depth maps that have been generalized to depth categories used for input to the HAAT model (e.g. 0-1, 1-2, 2-5, 5-10, 10+ depth intervals in metres) are also shown.

Table 6.4 Average March-November water levels for study wetlands in metres above sea level (m asl) for the different scenarios used in climate change evaluations (base case = historic; warm & dry scenario = CGCM2 A21). Adjustments were applied to both low and high base case conditions. The years used for low and high water level scenarios are shown.

Study Area	Base Case Low (m asl)	Base Case High (m asl)	Warm & Dry Adjustment (m)
Presqu'ile	74.50 (1965)	74.94 (1978)	-0.75
Long Point	173.66 (1964)	174.42 (1978)	-0.80
Mitchell's Bay	174.61 (1964)	175.03 (1978)	-1.0

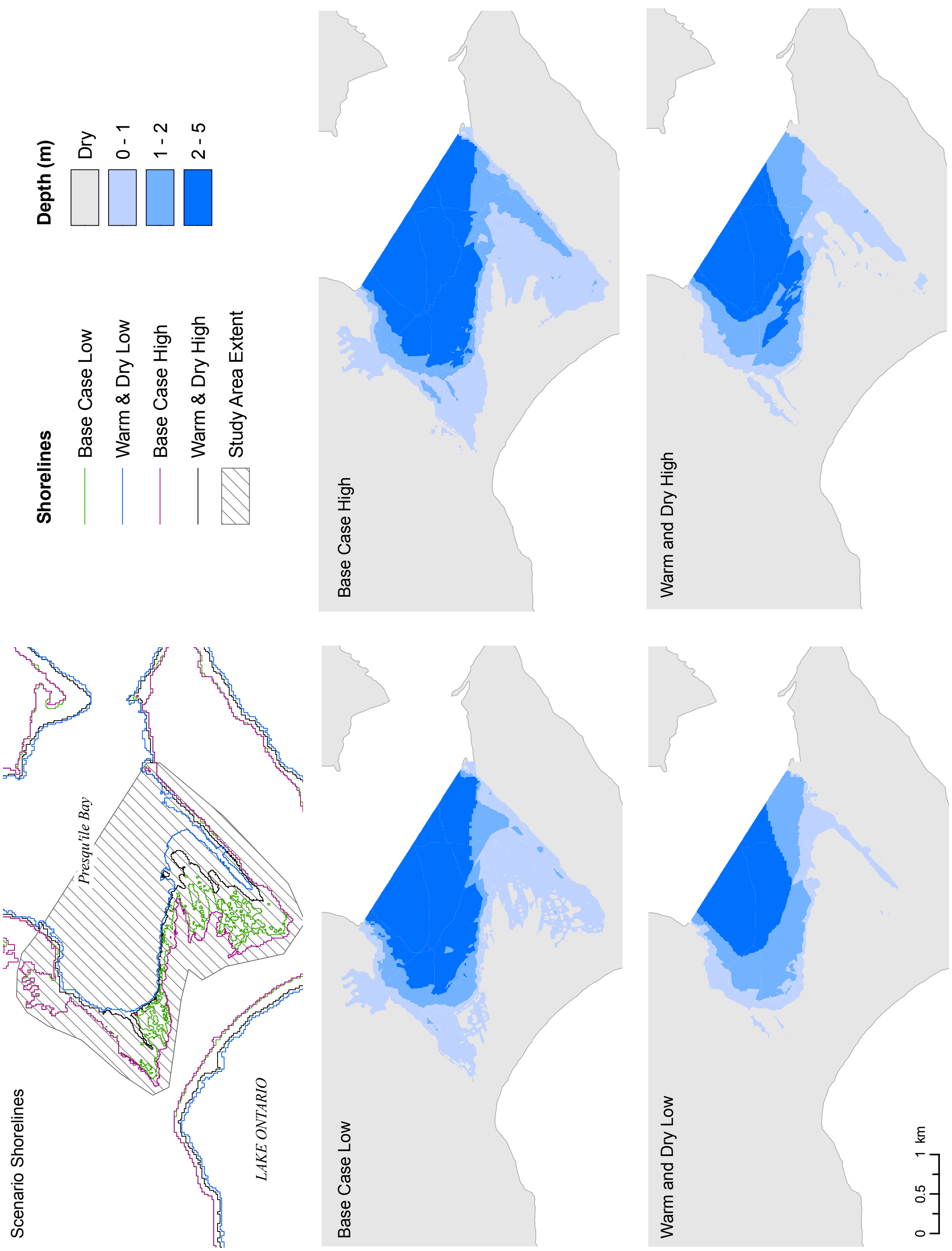


Figure 6.6 Presqu'ile Bay, Lake Ontario model extent and shoreline contours for base case low, base case high and warm & dry climate scenarios (low and high years). Figures show the depth contour areas used as input to the HAAT model for fish habitat evaluations.

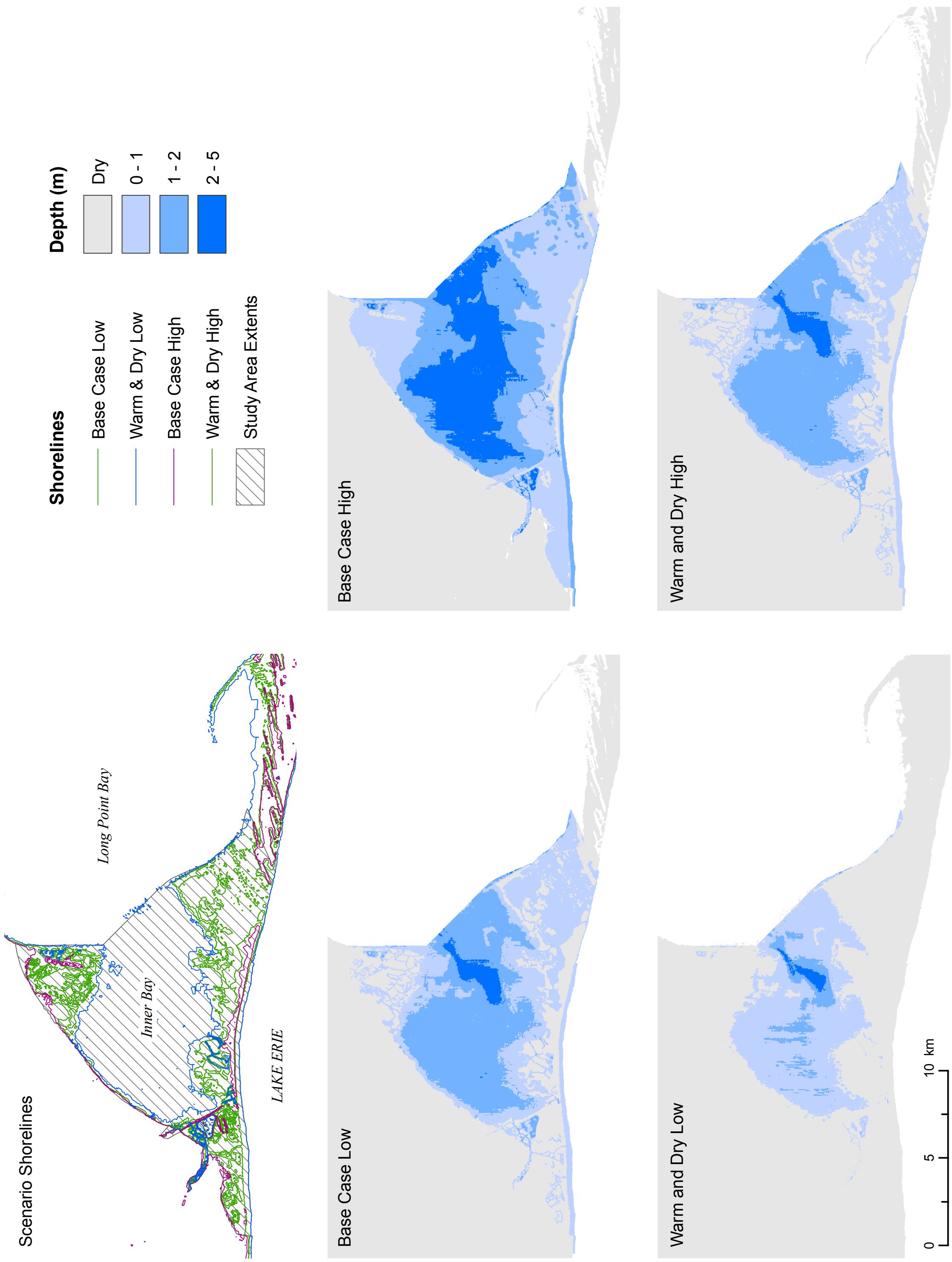


Figure 6.7 Long Point, Lake Erie model extent and shoreline contours for base case low, base case high and warm & dry climate scenarios (low and high years). Figures also show depth contours (areas) used as input for the HAAT model evaluations of fish habitat.

Figure 6.8 Mitchell's Bay, Lake St. Clair model extent and shoreline contours for base case low, base case high and warm & dry climate scenarios (low and high years). Lower figures show depth category areas that were used as HAAT input to fish habitat evaluations.

Vegetation

HAAT requires that vegetation within each unique polygon be described by percent cover of submergents, emergents, and no cover. A number of steps were taken to obtain a complete picture of vegetation density in each embayment from various sources, including spatial datasets of emergent vegetation (Chapter 7), and the use of submergent aquatic vegetation (SAV) modelling to fill data gaps in deeper waters.

Parts of an emergent vegetation coverage (layer) generated from the wetland vegetation model developed in Chapter 4 were used. The emergent classifications employed in the vegetation modelling were not consistent with the percent cover input requirements of HAAT. Therefore, emergent vegetation model output was reclassified into percent cover categories (Table 6.5). Any areas classified as 'Open Water' by the wetland vegetation model were substituted with submergent vegetation model output. A separate submergent vegetation model was developed as part of the fish habitat study and is outlined in the following section. A separate model was used because the modelled wetland vegetation area only included low water depths of up to 1 m, but submergent vegetation can exist to the extent of the euphotic zone, the depth of light penetration. Therefore, it was necessary to fill this data gap as SAV is important fish habitat.

Table 6.5 Wetland vegetation classes used in the study (Chapter 4) and the corresponding reclassifications into HAAT vegetation classes used in fish habitat evaluations

Wetland Vegetation Model Class	Class	HAAT Vegetation Classes (% Cover)		
		NO COVER	EMERGENT	SUBMERGENT
Open Water	1	Defined by submergent vegetation model		
Exposed Substrate	2	100	0	0
Emergent/Floating Mixed	3	5	15	80
Emergent	4	0	80	20
Meadow Marsh *	5	5	20	75
Treed/Shrub *	6	10	45	45

* These normally "dry" areas may be flooded under certain water level conditions and if used in fish habitat area calculations needed equivalent aquatic vegetation classifications based on plant structure

Submergent vegetation

Submergent vegetation densities were calculated for areas classified by the wetland vegetation community model as "Open Water", in addition to areas beyond the extent of the emergent modelling. Submergent vegetation establishes in areas that are relatively protected, at depths that are shallow enough for light penetration, and at sites that have the proper substrate type for the establishment of plants. Therefore, submergent densities were predicted using an empirical model based on a hierarchical application of algorithms using site-specific characteristics of fetch (a measure of wind- and wave-based exposure), water depth, and percent sand.

Fetch is a linear measure of distance between shoreline features and a point in a water body. A point coverage was generated using a GIS routine that measured the fetch between a series of points to the shoreline in 16 compass directions. Points were distributed at regular intervals that varied in resolution depending on the extent of the study area: Presqu'ile - 100 m, Long Point - 500 m, and Mitchell's Bay - 150 m. Average fetch over the 16 distances was calculated for each water level scenario (high and low water level base case and climate change warm & dry) using the appropriate shorelines (see Figures 6.6, 6.7, 6.8 for different shorelines). When the average fetch was less than 7 km, submergent vegetation was determined to be present; this threshold was based on previous analysis for vegetation distributions in Lake Ontario (Randall 2005; Bakelaar *et al.* 2006) and calibrated against known submergent vegetation distributions in the modelled wetlands (Doka *et al.* 2004).

The relationship between depth and submergent vegetation coverage is well established (Gasith and Hoyer 1998). Given adequate light penetration, submergents can exist in water up to depths of 10 m or more. The algorithm used for this part of the model was developed for the IJC LOSLR study and predicted submergent vegetation cover from water depth that was statistically determined by using submergent vegetation data collected in Lake Ontario at various depths (Bakelaar *et al.* 2006).

In addition to fetch (exposure) and depth (light penetration), substrate composition can affect the establishment of submergents. High sand content indicates depositional areas with little organic matter because they are high energy areas (Wetzel 1983). Therefore, when the dominant substrate was sand (i.e. ≥75%) and the depth model predicted moderate to high submergent densities (> 33% cover), the density values were then adjusted, *a priori,* to sparse (17%). This rule-based submergent model was validated using field data from the model embayments (Doka *et al.* 2006).

Fetch, depth and substrate, were used to assign a submergent vegetation percent cover to each polygon. The logical flow of equations and statements in the hierarchical submergent model is outlined in Box 6.1.

Box 6.1 Submergent vegetation model rules
If fetch is <7km then submergent vegetation (subveg) is present else no cover

If subveg present then subveg density = 15.463 * z + 79.019, where z = depth in metres below water level (a negative value)

If subveg density > 33% and sand ≥ 75% then subveg density = 17% (midpoint for low % cover category)

The submergent vegetation model was implemented using raster-based approaches in a GIS and then categorized by the percentage of the predicted submergent cover. Categorical assignments were more practical across large wetlands with a high resolution because similar areas could be lumped, reducing the number of calculations necessary. The midpoint of each of the vegetation classes was used as numerical input to the HAAT model for submergents. The remainder of the percent cover that was not assigned to either emergent or submergent coverages was assigned to the 'No Cover' class (Table 6.5). Emergent vegetation in 'Open Water' areas or outside the wetland community model emergent area extent was assumed to be nil. Emergent and submergent vegetation layers were merged to produce final vegetation maps for each base case and climate change scenario (Figures 6.9, 6.10, 6.11). Attributes for each layer included general vegetation descriptions (see legends Figures 6.9, 6.10, 6.11) and percent cover values (Tables 6.5, 6.6).

Table 6.6 Submergent vegetation percent cover values generalized for HAAT model input into three vegetation classifications

Submergent Vegetation Model Classes (% cover)	HAAT Vegetation Classes (% Cover) NO COVER	EMERGENT	SUBMERGENT
0	100	0	0
1-33 (Sparse)	83	0	17
33-67 (Moderate)	50	0	50
67-100 (Dense)	17	0	83

Substrate

Available information for substrate types in the modelled wetland areas was very limited. Sources included existing substrate survey data, shoreline type, and nearshore geology, as well as point samples from surveys collected as part of this study (Doka *et al.* 2004). A method for interpolating and extrapolating bottom types was developed for areas where point habitat surveys had been conducted in Long Point Bay and Mitchell's Bay (methods for habitat surveys are explained in Section 8.2.5). Data from each point sample (Long Point: N=52 and Mitchell's Bay: N=62) were used to generate Theissen polygons, some of which extended to the shoreline (Minns and Bakelaar 1999). The level of uncertainty in classification would be high due to extrapolation, especially in areas of < 1 m water depth and between actual sampled sites. In Presqu'ile Bay, a method for extrapolating the shoreline classification and nearshore geology (data available for Lake Ontario) to bottom composition was used (Bakelaar *et al.* 2006).

Percent composition in predetermined substrate classes were used for input to HAAT in all scenarios. Substrate categories required by HAAT include bedrock, boulder, cobble, rubble, gravel, sand, silt, clay, hardpan clay, and pelagic. The pelagic class, which was not used in the analysis, is normally used in deep waters when composition is unknown. For habitat modelling purposes, substrate composition was considered static through time and did not change for any polygon between climate scenarios. However, the overall substrate composition within a wetland changed depending on the water levels and the number of polygons flooded (Figures 6.8, 6.9, 6.10).

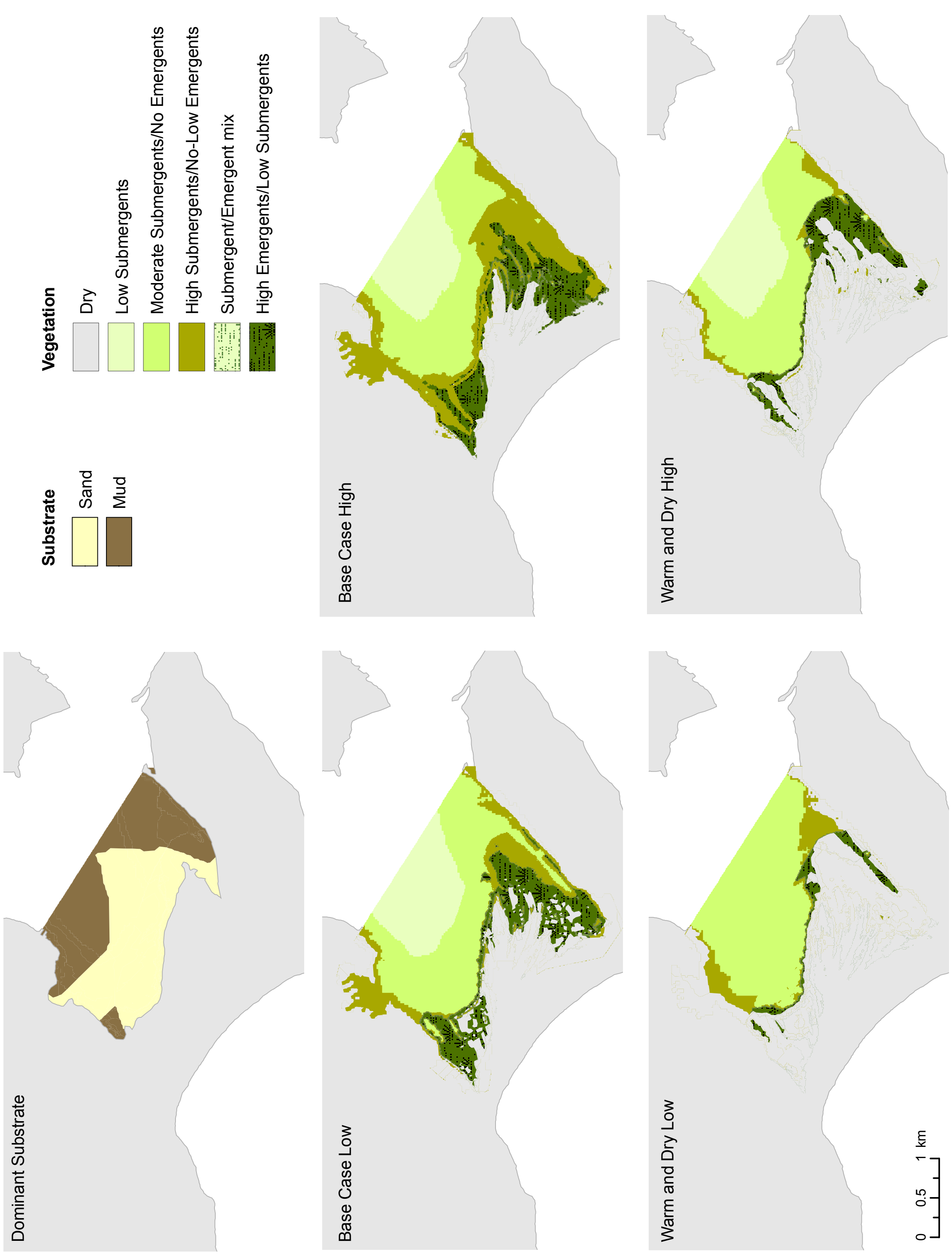


Figure 6.9 Substrate and vegetation (emergent and submergent) habitat classifications used in different climate scenarios and water level (base case low and high, warm & dry low and high) for Presqu'ile, Lake Ontario

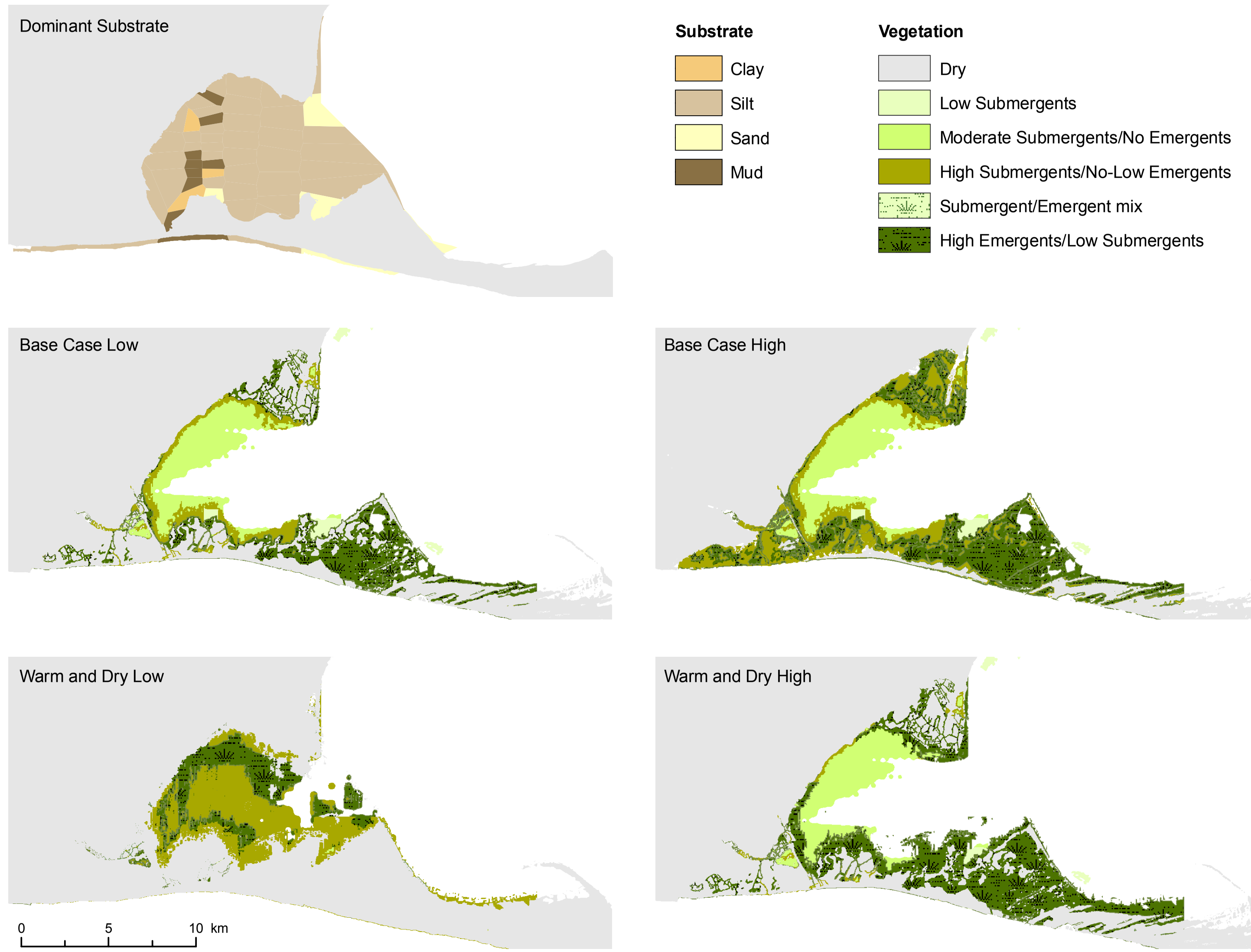


Figure 6.10 Substrate and vegetation (emergent and submergent) habitat classifications for climate and water level scenarios (base case low and high, warm & dry low and high) in Long Point Bay, Lake Erie

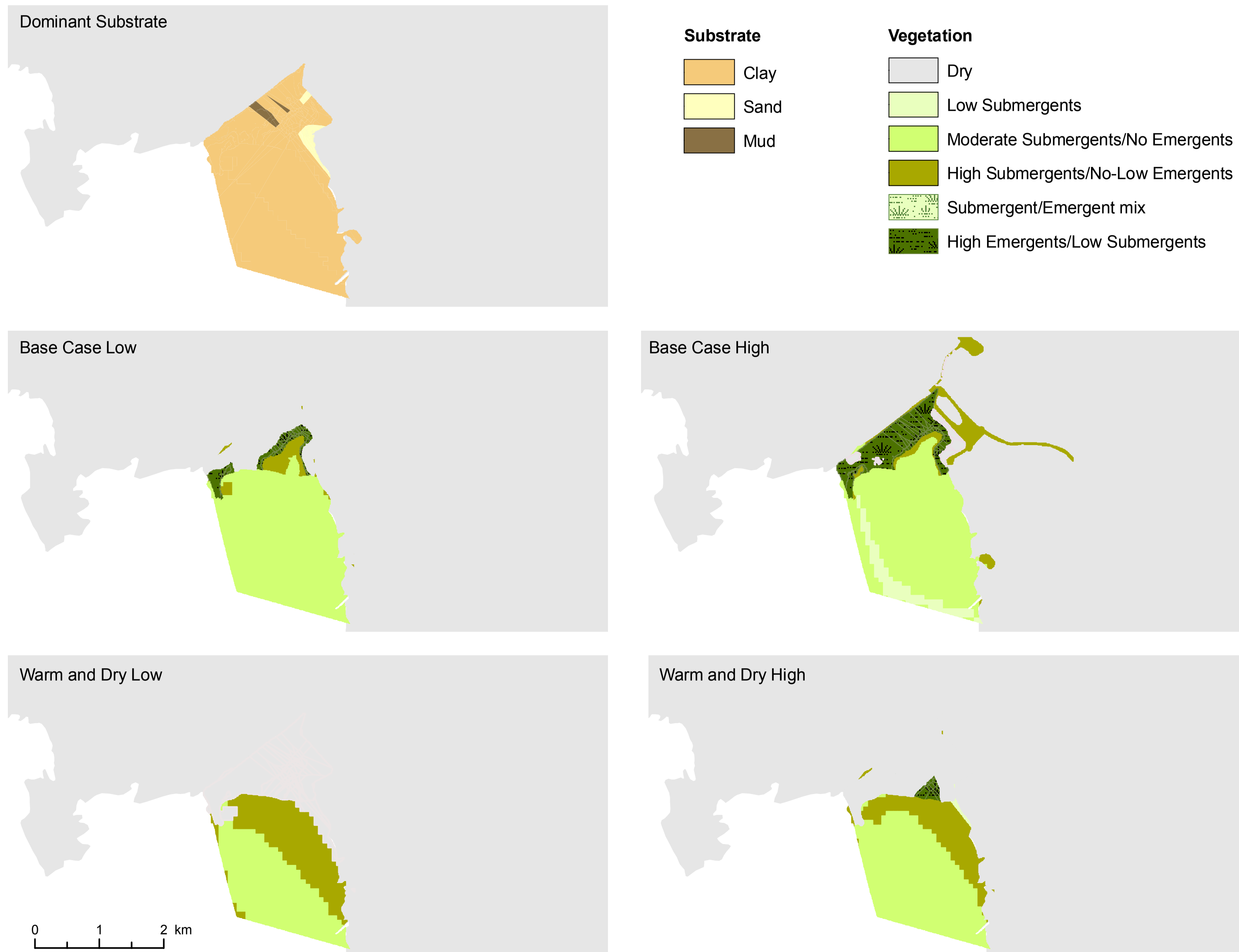


Figure 6.11 Substrate and vegetation (submergent and emergent) habitat classifications for different climate and water level scenarios (base case low and high, warm & dry low and high) in Mitchell's Bay, Lake St. Clair

Unique Habitat Combinations

A final input table of "unique" habitat areas (i.e. a sum of the areas across polygons with the same habitat features) was created by a topological overlay of depth, vegetation, and substrate layers. The result was a new GIS layer that preserved the features of all the percent compositions by category stored in the original coverages (see flow chart in Figure 6.5). Polygon areas for each unique habitat combination (i.e. similar depth, substrate composition, vegetative cover) were summed and used as input to HAAT for each study area and water level scenario (see Appendix 6.1 for an example input table). Aggregating the habitat features in this manner reduced the number of records in the HAAT input table and provided a simple way for the suitability output values to be re-mapped to polygons with similar attributes.

6.4.3.2 Assessment of Fish Habitat Suitability and Availability

A general application of the HAAT model for calculating fish habitat suitability and availability requires, at a minimum, that a region be described by the percent areal coverage of specific depth, substrate, and vegetation categories (see Section 6.4.1 and Appendix 6.1). Based on the life stage requirements of the species found in an area, wetlands can be classified according to the relative suitability of each habitat patch. Habitat suitabilities can be assigned for any species, life stage, or a grouping of species that are usually a functional group or guild. Suitabilities represent the relative importance of a habitat patch, values range between 0 and 1, where 0 represents no suitability and 1 represents the highest suitability. Suitability rankings are based on the requirements or preferences of the species, life stage, or group, for the physical characteristics being evaluated in a system.

In this assessment, fish species were assigned to adult thermal and feeding guilds based on temperature preferences and piscivory (Coker *et al.* 2001). A maximum of six guilds were assessed per wetland, depending on the local species composition. These six guilds were warm, cool, and coldwater fishes assigned as piscivores and non-piscivores within each thermal guild. Each sub-guild was further subdivided into three life stages: spawning, YOY, and juvenile/adult stages, as each stage generally has niche shifts with their own habitat requirements (Persson and Crowder 1998). Therefore, up to a maximum of 18 subgroups (life stage by thermal guild by feeding guild combination) were evaluated depending on the site. Although coldwater species were found in wetland surveys, none were present in the wetlands that were evaluated in this assessment.

The species present over two years of spring and fall sampling in northshore Lake Ontario wetlands belonged to coolwater (11 spp.) and warmwater guilds (17 spp; Table 6.7). Six species were piscivores while 22 fishes belonged to lower trophic levels. No species at risk were found but non-native common carp (*Cyprinella spiloptera*) was present in the warmwater non-piscivore guild. The combined species list for coastal fishes sampled as part of a wetland study (Chapter 8) was used in lieu of site-specific information on Presqu'ile Bay fish assemblages. This wetland was not part of the dyked wetlands study design.

Table 6.7 Fish species by thermal and feeding guilds used in habitat assessments for Presqu'ile Bay, Lake Ontario. The coastal species used in the assessment were species sampled in Parrott's Bay and Amherst Bay as part of a coastal wetland study.

Guild	Cool	Warm
Non-Piscivore (N)	white sucker, banded killifish, golden shiner, spottail shiner, yellow perch, johnny darter, emerald shiner, brook stickleback	rock bass, black bullhead, brown bullhead, common carp, brook silverside, pumpkinseed, bluegill, white perch, black crappie, central mudminnow, freshwater drum, bluntnose minnow, spotfin shiner, fathead minnow
Piscivore (P)	northern pike, longnose gar, walleye	bowfin, largemouth bass, grass pickerel

The species caught during two years of spring and fall sampling in the Inner Bay and Big Creek Marsh areas of Long Point belonged to coolwater (16 spp.) and warmwater guilds (20 spp.; Table 6.8). Five species were piscivores while 31 fishes belonged to lower trophic levels. Several species at risk were found, including pugnose shiner (*Notropis anogenus*), spotted gar (*Lepisosteus oculatus*), warmouth (*Lepomis gulosus*), and pugnose minnow, representing several guilds. Non-native common carp, goldfish (*Carassius auratus*), and round goby (*Neogobius melanostomus*) were present in the non-piscivore guilds.

Table 6.8 Fish species by thermal and feeding guilds used in habitat assessments for Long Point Bay, Lake Erie (* indicates that the species was sampled but there was not enough information on habitat requirements for evaluation). The coastal species used in the assessment were actual species sampled in the inner Long Point Bay and Big Creek NWA as part of a coastal wetland study (see Section 8.2.5 for details).

Guild	Cool	Warm
Non-Piscivore (N)	white sucker, banded killifish, round goby, golden shiner, spottail shiner, yellow perch, quillback, pugnose shiner, emerald shiner, blackchin shiner, blacknose shiner, logperch, johnny darter	rock bass, black bullhead, brown bullhead, common carp, brook silverside, pumpkinseed, bluegill, yellow bullhead, freshwater drum, goldfish, gizzard shad, tadpole madtom, bluntnose minnow, warmouth, mimic shiner, pugnose minnow, black crappie, central mudminnow
Piscivore (P)	northern pike, longnose gar, spotted gar*	bowfin, largemouth bass

The species surveyed during two years of spring and fall field surveys in the Lake St. Clair area (both Mitchell's Bay and National Wildlife Areas) belonged to coolwater (15 spp.) and warmwater guilds (22 spp; Table 6.9). Seven species were piscivores while 30 fishes belonged to lower trophic levels. Several species at risk were found in surveys, including pugnose shiner, spotted sucker, and lake chubsucker from the non-piscivore guilds. Non-native common carp, goldfish, white perch, and round goby also contributed to the same guilds.

Table 6.9 Fish species by thermal and feeding guilds used in habitat assessments for Mitchell's Bay, Lake St. Clair (* indicates that the species was sampled but there was not enough information on habitat requirements for evaluation). The coastal species used in the assessment were actual species sampled in Mitchell's Bay and the nearby St. Clair NWA as part of a coastal wetland study (see Section 8.2.5 for details).

Guild	Cool	Warm
Non-Piscivore (N)	quillback, white sucker, banded killifish shorthead redhorse, round goby*, golden shiner, pugnose shiner, emerald shiner, blackchin shiner, spottail shiner, yellow perch, Iowa darter	rock bass, black bullhead, yellow bullhead, brown bullhead, freshwater drum, goldfish, common carp, gizzard shad, brook silverside, pumpkinseed, bluegill, spotted sucker, white perch, tadpole madtom, bluntnose minnow, black crappie, lake chubsucker, central mudminnow
Piscivore (P)	northern pike, longnose gar, walleye	bowfin, smallmouth bass, largemouth bass, muskellunge

Suitabilities were assigned to each habitat patch based on the requirements of each species within a guild at a particular life stage and based on the physical characteristics of each patch (i.e. its depth, substrate, and vegetation). An overall suitability for each subgroup was determined based on equal weighting of the species within the group. Therefore, common species, species at risk, and invasive species were all treated equally. Composite suitabilities across life stages within a guild, or across all guilds, were also calculated for local species lists, again equally weighted across species. Details of how the composite suitabilities were calculated can be found in Minns *et al.* (2001, 2006).

6.4.4 Conclusion

An evaluation of the vulnerability results for fishes, in light of the vegetation and bird community effects observed in the lower Great Lakes, is presented in Chapter 7. The implications of the other climate scenario effects that were not tested in the fish response modelling and the overall effects of climate change on the Great Lakes ecosystem are also discussed in a synthesis (Chapter 9). The implications for management of our natural resources in the Great Lakes coastal area and waters are presented there.

It should be noted that temperature changes in nearshore areas were not considered in the assessment of fish habitat supply, only coastal habitat changes. Therefore, the responses of guilds based on their thermal requirements, like egg development and survival needs, as well as YOY tolerances, could produce different responses under climate change but likely not change trajectories projected for guild responses. These thermal requirements were taken into consideration in the vulnerability assessment of Great Lakes fish species earlier and the proportion of high risk species in the habitat assessment was also high. Twelve species of the 28 fishes evaluated in Presqu'ile Bay, 17 of 36 species in Long Point Bay, and 15 of 37 species in Mitchell's Bay, were high risk. Future work will factor thermal changes in the effects of water level reductions on coastal habitat availability by incorporating temperature into HAAT evaluations.

The physical characteristics across sites and scenarios that were evaluated varied in percent composition. The percent composition of depths in different scenarios was different between wetlands. The relative percentages of depths in 0-1, 1-2 and 2-5 m categories remained relatively similar no matter which water level or scenario was tested in Presqu'ile Bay. The 2-5 m depths were dramatically reduced in Long Point Bay under mid-range water level changes and 1-2 m depths were reduced under low climate conditions. Deeper depths were also dramatically reduced under climate change conditions in Mitchell's Bay but 1-2 m depths remained a high percentage of the area. Substrate compositions were relatively consistent across scenarios but different between wetlands; Presqu'ile had roughly 50% sand and mud, Long Point has the most diverse substrate but was mainly composed of silt with some clay, sand, and mud areas, Mitchell's Bay has largely clay-based soils. Percentage of emergent vegetation declined in all wetlands with the exception of Long Point Bay where climate scenario has the greatest range in vegetation communities, especially submergent densities. These physical differences are the basis for the fish habitat comparisons and evaluations presented in Chapter 7.

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7.0 Integrated Assessment: Vulnerability of Great Lakes Coastal Wetland Communities to Climate Change

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In this section the models described in Chapters 4, 5, and 6 were used to assess the potential effects of climate change on wetland vegetation, birds, and fishes through the application of various water level scenarios. First, the future distribution and abundance of wetland vegetation classes were determined by using water levels perturbed by climate change effects in a rule-based vegetation model (Chapter 4). Then, the resultant digital maps of wetland class distributions, as well as the average water level depth during the breeding season, were combined to predict effects on wetland-dependent breeding bird abundance and density using nonlinear regression models (Chapter 5). Lastly, fish habitat suitability was assessed using the digital vegetation maps (areal extent of emergent marsh vegetation), in combination with estimated submergent vegetation distribution and density, substrate type, and water depth (Chapter 6). Eight wetlands on Lakes Ontario and Erie were assessed for wetland vegetation class, six wetlands for bird responses and two wetlands for fish; fish habitat was also modelled for one wetland on Lake St. Clair. Tabular and mapped results for wetland vegetation, birds, and fish were organized so that findings could be compared for low and high water level conditions between base case and climate change scenarios.

The modelling results were integrated for an assessment of the potential impacts of water level changes in coastal wetlands – in terms of their vegetation, bird, and fish communities. Assessment questions included:

- Were there different responses between regulated Lake Ontario and unregulated Lake Erie?
- Does wetland hydrogeomorphology affect response to water level change (i.e. lacustrine or riverine conditions)?
- Were there any identifiable water level thresholds in biotic responses?
- Does the overall quality of wetlands and habitats change?
- Were there any communities that were affected more than others?

7.1 Climate Change Water Level Scenarios and Ecosystem Modelling

The impacts of climate change on Great Lakes coastal ecosystems were assessed using water level changes as the surrogate for changes in climate. To explore the widest range of conditions for the climate change assessment, two cases – a high and a low historical water level state – were used for initial simulations (base case comparisons) in the modelling. Different antecedent conditions, creating low and high water level conditions, might reveal different responses under climate change scenarios. Ideally, the high and low water levels chosen would be the lowest or highest observed water levels recorded to assess the maximum range of effects. However, it was necessary to use low or high water level conditions for years where there was historical air photo coverage that could be used to create wetland vegetation mapping (see Chapter 4). These high and low water years are summarized in Table 7.1.

Table 7.1 Low and high water level years selected to initialize climate change simulation runs in wetland ecosystem response modelling

Lake and Wetland	Low Water Level		High Water Level	
	Year	Annual average (m) IGLD	Year	Annual average (m) IGLD
Lake Ontario				
Hay Bay	1962	74.64	1978	74.92
Lynde Creek	1959	74.48	1978	74.92
Presqu'ile Bay	1965	74.40	1978	74.92
South Bay	1962	74.64	1978	74.92
Lake Erie				
Dunnville	1965	173.71	1978	174.37
Long Point	1964	173.61	1978	174.37
Turkey Point	1964	173.61	1978	174.37
Rondeau Bay	1962	173.91	1978	174.37
Lake St. Clair				
Mitchell's Bay	1964	174.37	1978	175.27

For wetland vegetation modelling, an assumption was that current vegetation communities reflected the influence of water levels for the current year as well as the previous 40 years (see Chapter 2 and historical analysis and modelling in Chapter 4). To model vegetation, annual mean water levels were chosen as adequate input because they integrate the pattern of flooding and dewatering that influences wetland vegetation abundance and distributions. Starting from the representative low or high water level year specific to each wetland (e.g. low 1964 and high 1978 for Long Point), a time series of 41 years of annual historical water levels was compiled (e.g. starting in year 1923 for low and 1937 for the high water year at Long Point). These time series were input to the rule-based vegetation model to predict the base case wetland community distributions for low or high water level conditions for each wetland.

The effects of climate change were imposed by applying annual water level change fields to the 41-year historical water level time series for four separate climate change scenarios (Chapter 2). The water level change fields, derived from the LOSLR study hydrologic modelling, are listed in Table 7.2. The climate change, 41-year time series for low or high conditions was input to the rule-based vegetation model to predict vegetation effects. The modelled climate change wetland classes were compared to their respective high or low modelled base case classes to calculate the change between the two conditions and thereby assess the climate change effects under the different scenarios.

Both the bird and fish modelling incorporated the results of the wetland vegetation community conditions (current and climate change) and also applied water level change fields specific to bird and fish habitat responses for water depth (see Table 7.2). These change fields were only applied to the current low or high water level year to simulate climate change; these models were not dependent on a 41-year time series as the vegetation response modelling.

Table 7.2 Summary of water level change fields used in modelling wetland ecosystem responses to climate change (based on 100-year hydrologic modelling run)

	Water Level Change Fields (m)		
Lake and Scenario	**Wetland Vegetation** Annual Average	**Birds** 3-Month Breeding Season (May to July)	**Fish** Growing Season (April to November)
Lake Ontario			
Not as Warm & Wet	+0.06*	-	-
Not as Warm & Dry	-0.46*	-	-
Warm & Wet	-0.60*	-0.70	-
Warm & Dry	-0.75*	-0.88	-0.75
Lake Erie			
Not as Warm & Wet	-0.15	-	-
Not as Warm & Dry	-0.55	-	-
Warm & Wet	-0.67	-0.63	-
Warm & Dry	-0.81	-0.81	-0.80
Lake St. Clair			
Not as Warm & Wet	-0.21	-	-
Not as Warm & Dry	-0.63	-	-
Warm & Wet	-0.81	-0.78	-
Warm & Dry	-0.99	-1.00	-1.00

* based on pre-project conditions and 29-year modelling run

The methods outlined in Chapter 5 for modelling and mapping wetland breeding bird response to changes in the quantity and quality of habitat were used to compare potential changes in wetland breeding bird communities under climate change scenarios (warm & dry and warm & wet) during high and low water level conditions. Abundance indices were developed for each wetland breeding bird guild, using density estimates of representative species within each predicted wetland habitat type. Emergent marsh nesting bird guild estimates also incorporated regression equations to adjust density estimates based upon water depth within emergent marsh during the breeding season.

The methods outlined in Chapter 6 for mapping and predicting physical variables and assessing fish habitat quantity and quality were used in the analysis of fish habitat supply. The composite suitability values across all life stages and guilds were mapped to illustrate trends in habitat suitability and availability for the local fish assemblages between historic (base case) and climate change (warm & dry) highs and lows. The WSA for each guild and life stage was plotted separately for comparison across scenarios. Details of how the composite suitabilities were calculated can be found in Minns *et al.* (2001, 2006). From this comparison, the most sensitive life stage and guild were selected for each modelled wetland and the suitabilities mapped for each of the scenarios. The spatial extent that was modelled was determined by the availability of spatial information on vegetation (i.e. actual vegetation information or input variables used in modelling submergents), substrate, and bathymetry. This was important for a fuller assessment of fish habitat at the appropriate scale (i.e. not just coastal margins and uplands in emergent marsh areas).

7.2 Lake Ontario Climate Change Assessment

Modelling the potential effects of climate change by using water level scenarios included assessing effects on vegetation communities, bird abundance, and fish habitat supply on the Presqu'ile, Hay Bay, Lynde Creek, and South Bay wetlands. The results are summarized in Tables 7.3 (vegetation), 7.4 (bird), and 7.5 (fish), as well as other associated figures. Results are mainly separated into low and high water level year comparisons for each wetland modelled.

7.2.1 Presqu'ile Bay

7.2.1.1 Vegetation Response

Low Water Year (1965)

In the low water year base case, 60% of the Presqu'ile wetland was vegetated. Emergent marsh vegetation composed 50% of the total wetland area, meadow marsh accounted for another 9% and treed/shrub was less than 1% of the area. The remaining 40% of the wetland was open water (Figure 7.1).

As water levels increased minimally (+0.06 m) under the not as warm & wet scenario, wetland vegetation communities remained much the same as under base case conditions. The amount of open water increased slightly (+6.9 ha, +6%) from base case, flooding emergent vegetation in deeper sections of the wetland (Table 7.3). There was a slight shift of wetland communities landward, up the moisture gradient, as water levels increased. Emergent vegetation migrated landward into areas of meadow marsh while meadow moved upslope into areas of treed/shrub (Figure 7.1). As a result, the total amount of vegetated area within the wetland marginally decreased by 2% under the not as warm & wet scenario.

There were notable changes in the Presqu'ile wetland under the three other scenarios, which all projected water levels declines with climate change. The area of open water decreased so that under the most extreme warm & dry scenario, where water levels were projected to decline -0.75 m from base case, -54.4 ha (-49%) of open water was lost. The extent of emergent vegetation in the wetland also decreased under the three most extreme scenarios, and with the warm & dry scenario, emergent vegetation decreased a total of -90.3 ha (-66%) from base case. However, modelling the expansion of emergent vegetation may have been limited by an insufficient "clip" size of the wetland study area, constraining downslope movement beyond the extents of the study area. As water levels declined, meadow marsh expanded, migrating lakeward. Compared to base case, the area of meadow marsh actually increased the most under the not as warm & dry scenario (+89.3 ha, +352%), while increases under warm & wet and warm & dry were not as high. This was due to greater increases in the area of treed/shrub under these two scenarios, as the community progressively expanded lakeward along the moisture gradient as water levels continued to decline for each scenario. Under the warm & dry scenario, the area of treed/shrub increased +71.2 ha or +5,474% (Table 7.3). Due to the increases in meadow marsh and treed/shrub, the total vegetated area increased to 79% of the wetland under the warm & dry scenario (Figure 7.1).

High Water Year (1978)

In the high water base case, approximately 48% of the wetland was vegetated. Emergent vegetation comprised 47% of the wetland, while meadow marsh and treed/shrub totalled less than 1% of the wetland area. Open water dominated the wetland and accounted for 52% of the total wetland area (Figure 7.3; Table 7.3).

Similar to the low water year, the area of open water increased in the high water year, while the area of the vegetated wetland communities decreased under the not as warm & wet scenario, which projected a marginal

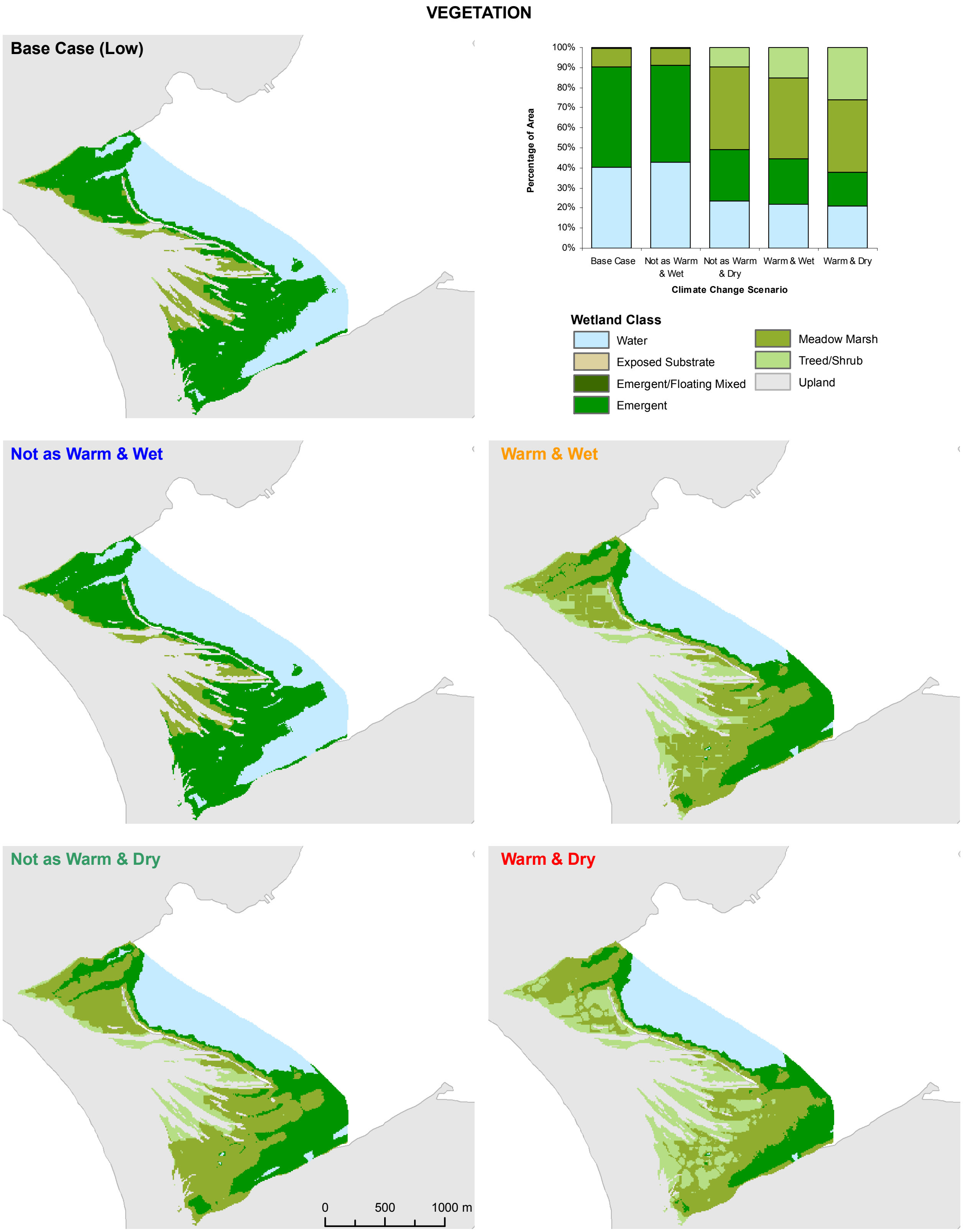

Figure 7.1 Predicted wetland class distribution under base case and climate change scenarios at Presqu'ile Bay during low water (1965) conditions

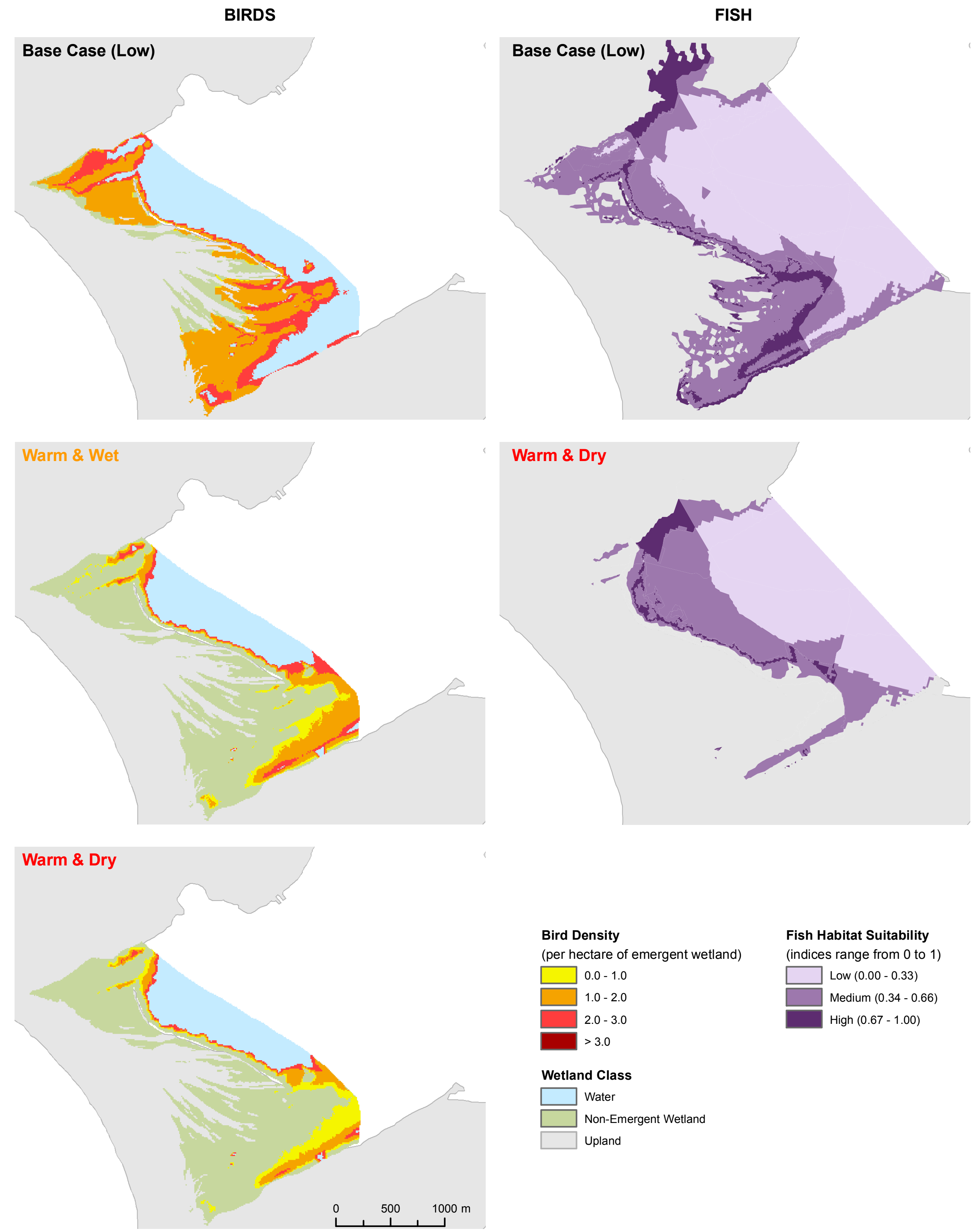

Figure 7.2 Predicted marsh nesting obligate bird distribution and overall fish habitat suitability under base case and climate change scenarios at Presqu'ile Bay during low water (1965) conditions (warm & wet scenario, birds only; warm & dry scenario, birds and fish). Note: fish suitability maps show wetted area only.

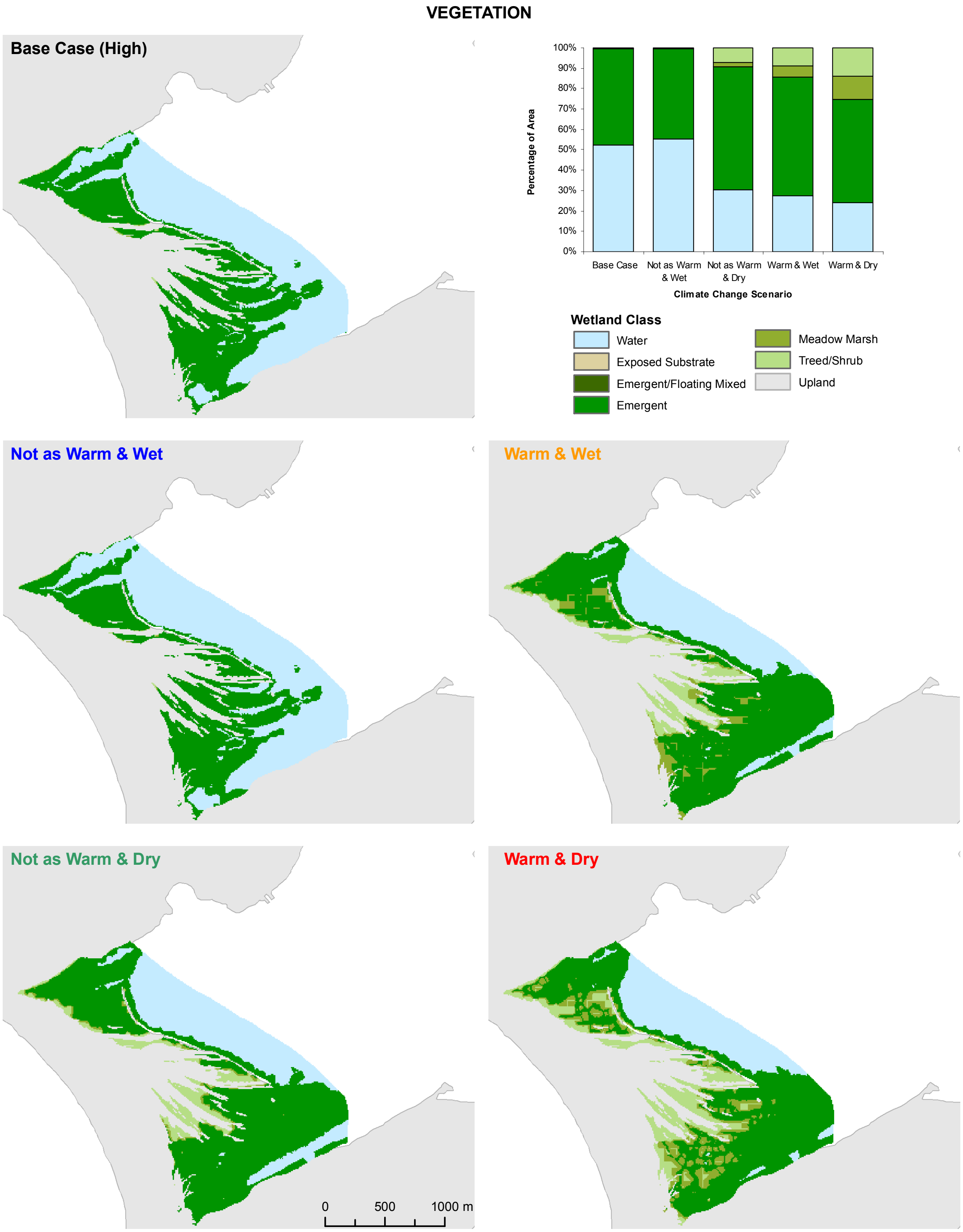

Figure 7.3 Predicted wetland class distribution under base case and climate change scenarios at Presqu'ile Bay during high water (1978) conditions

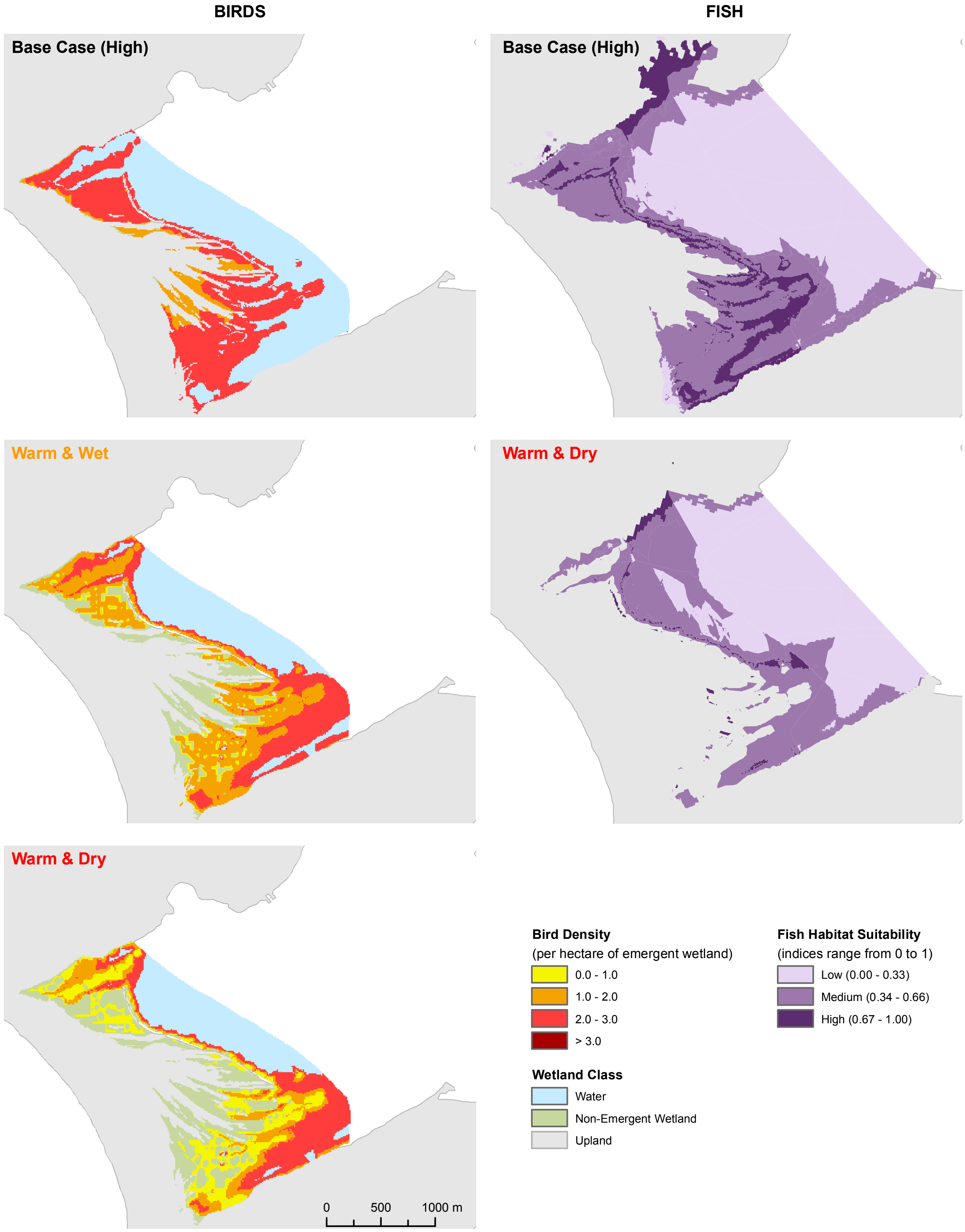

Figure 7.4 Predicted marsh nesting obligate bird densities and overall fish community habitat suitability under base case and climate change scenarios at Presqu'ile Bay during high water (1978) conditions (warm & wet, birds only; warm & dry, birds and fish). Note: fish suitability maps show wetted area only.

Table 7.3 Wetland class area (ha) projected under climate change scenarios, change in area (Δ ha), and percent change (%Δ) from base case for Lake Ontario

Wetland	Year	Climate Change Scenario	Water			Exposed Substrate			Emergent/Floating			Emergent			Meadow Marsh			Treed/ Shrub		
			ha	Δ ha	% Δ	ha	Δ ha	% Δ	ha	Δ ha	% Δ	ha	Δ ha	% Δ	ha	Δ ha	% Δ	ha	Δ ha	% Δ
Hay Bay	1962	Base Case	53.02			0.00			0.00			64.45			143.87			7.75		
	(low water)	NWW	61.43	8.41	15.86	0.00	-	-	0.00	-	-	59.71	-4.74	-7.35	141.13	-2.74	-1.90	6.79	-0.96	-12.39
		NWD	25.27	-27.75	-52.34	0.00	-	-	0.00	-	-	65.76	1.31	2.03	22.93	-120.94	-84.06	155.13	147.38	1901.68
		WW	19.68	-33.34	-62.88	0.00	-	-	0.00	-	-	65.78	1.33	2.06	22.62	-121.25	-84.28	161.01	153.26	1977.55
		WD	11.08	-41.94	-79.10	0.00	-	-	0.00	-	-	64.56	0.11	0.17	24.38	-119.49	-83.05	169.07	161.32	2081.55
	1978	Base Case	82.33			0.00			0.00			175.59			3.42			7.75		
	(high water)	NWW	85.92	3.59	4.36	0.00	-	-	0.00	-	-	173.24	-2.35	-1.34	3.14	-0.28	-8.19	6.79	-0.96	-12.39
		NWD	41.39	-40.94	-49.73	0.00	-	-	0.00	-	-	63.96	-111.63	-63.57	8.61	5.19	151.75	155.13	147.38	1901.68
		WW	34.72	-47.61	-57.83	0.00	-	-	0.00	-	-	64.80	-110.79	-63.10	8.56	5.14	150.29	161.01	153.26	1977.55
		WD	26.72	-55.61	-67.55	0.00	-	-	0.00	-	-	64.79	-110.80	-63.10	8.51	5.09	148.83	169.07	161.32	2081.55
Lynde Creek	1959	Base Case	0.23			0.00			0.00			24.74			30.27			0.99		
	(low water)	NWW	0.51	0.28	121.74	0.00	-	-	0.00	-	-	25.88	1.14	4.61	29.16	-1.11	-3.67	0.68	-0.31	-31.31
		NWD	0.02	-0.21	-91.30	0.00	-	-	0.00	-	-	4.86	-19.88	-80.36	22.37	-7.90	-26.10	28.98	27.99	2827.27
		WW	0.02	-0.21	-91.30	0.00	-	-	0.00	-	-	1.61	-23.13	-93.49	23.17	-7.10	-23.46	31.43	30.44	3074.75
		WD	0.02	-0.21	-91.30	0.00	-	-	0.00	-	-	0.31	-24.43	-98.75	20.86	-9.41	-31.09	35.04	34.05	3439.39
	1978	Base Case	4.88			0.00			0.00			49.19			1.17			0.99		
	(high water)	NWW	7.80	2.92	59.84	0.00	-	-	0.00	-	-	46.79	-2.40	-4.88	0.96	-0.21	-17.95	0.68	-0.31	-31.31
		NWD	0.06	-4.82	-98.77	0.00	-	-	0.00	-	-	23.68	-25.51	-51.86	3.51	2.34	200.00	28.98	27.99	2827.27
		WW	0.05	-4.83	-98.98	0.00	-	-	0.00	-	-	20.89	-28.30	-57.53	3.86	2.69	229.91	31.43	30.44	3074.75
		WD	0.02	-4.86	-99.59	0.00	-	-	0.00	-	-	11.74	-37.45	-76.13	9.43	8.26	705.98	35.04	34.05	3439.39
Presqu'ile Bay	1965	Base Case	111.94			0.00			0.00			137.79			25.33			1.30		
	(low water)	NWW	118.83	6.89	6.16	0.00	-	-	0.00	-	-	133.21	-4.58	-3.32	23.21	-2.12	-8.37	1.11	-0.19	-14.62
		NWD	65.09	-46.85	-41.85	0.00	-	-	0.00	-	-	70.41	-67.38	-48.90	114.59	89.26	352.39	26.27	24.97	1920.77
		WW	60.51	-51.43	-45.94	0.00	-	-	0.00	-	-	62.72	-75.07	-54.48	111.82	86.49	341.45	41.31	40.01	3077.69
		WD	57.53	-54.41	-48.61	0.00	-	-	0.00	-	-	47.45	-90.34	-65.56	98.92	73.59	290.53	72.46	71.16	5473.85
	1978	Base Case	144.84			0.00			0.00			130.28			0.41			0.83		
	(high water)	NWW	152.40	7.56	5.22	0.00	-	-	0.00	-	-	122.88	-7.40	-5.68	0.32	-0.09	-21.95	0.76	-0.07	-8.43
		NWD	84.09	-60.75	-41.94	0.00	-	-	0.00	-	-	166.39	36.11	27.72	5.49	5.08	1239.02	20.39	19.56	2356.63
		WW	75.48	-69.36	-47.89	0.00	-	-	0.00	-	-	160.84	30.56	23.46	16.04	15.63	3812.20	24.00	23.17	2791.57
		WD	66.61	-78.23	-54.01	0.00	-	-	0.00	-	-	140.20	9.92	7.61	31.29	30.88	7531.71	38.26	37.43	4509.64
South Bay	1962	Base Case	15.40			0.00			0.00			31.90			7.64			1.42		
	(low water)	NWW	17.45	2.05	13.31	0.00	-	-	0.00	-	-	32.11	0.21	0.66	5.56	-2.08	-27.23	1.24	-0.18	-12.68
		NWD	0.66	-14.74	-95.71	0.00	-	-	0.00	-	-	28.85	-3.05	-9.56	15.56	7.92	103.66	11.29	9.87	695.07
		WW	0.03	-15.37	-99.81	0.00	-	-	0.00	-	-	26.59	-5.31	-16.65	13.20	5.56	72.77	16.54	15.12	1064.79
		WD	0.00	-15.40	-100.00	0.00	-	-	0.00	-	-	21.78	-10.12	-31.72	11.88	4.24	55.50	22.70	21.28	1498.59
	1978	Base Case	25.23			0.00			0.00			28.38			1.33			1.42		
	(high water)	NWW	26.91	1.68	6.66	0.00	-	-	0.00	-	-	27.23	-1.15	-4.05	0.98	-0.35	-26.32	1.24	-0.18	-12.68
		NWD	7.99	-17.24	-68.33	0.00	-	-	0.00	-	-	28.05	-0.33	-1.16	9.03	7.70	578.95	11.29	9.87	695.07
		WW	3.94	-21.29	-84.38	0.00	-	-	0.00	-	-	29.54	1.16	4.09	6.34	5.01	376.69	16.54	15.12	1064.79
		WD	0.95	-24.28	-96.23	0.00	-	-	0.00	-	-	28.83	0.45	1.59	3.88	2.55	191.73	22.70	21.28	1498.59

Climate Change Scenario: Not as Warm & Wet (NWW), Not as Warm & Dry (NWD), Warm & Wet (WW), Warm & Dry (WD)

Table 7.4 Indices of abundance, average water depth in emergent marsh during the breeding season, and density per hectare a for wetland bird nesting guilds projected under climate change scenarios, change in abundance (% Δ) from base case

Wetland	Year	Climate Change Scenario	Emergent Marsh	Marsh Nesting Obligates			Marsh Nesting Generalists			Meadow Marsh Nesters		Treed/Shrub Nesters	
			Average Water Depth (m)	Index of Abundance	% Δ in Abundance	Density per ha	Index of Abundance	% Δ in Abundance	Density per ha	Index of Abundance	% Δ in Abundance	Index of Abundance	% Δ in Abundance
LAKE ONTARIO													
Lynde Creek	1964	Base Case	0.03	30.27		1.22	95.64		3.87	112.00		5.54	
	(low water)	Warm & Dry	-0.33	0.05	-99.85	0.15	0.94	-99.02	3.02	77.18	-31.09	196.22	3439.39
		Warm & Wet	-0.30	0.19	-99.39	0.12	5.05	-94.72	3.13	85.73	-23.46	176.01	3074.75
	1978	Base Case	0.39	103.92		2.11	207.89		4.23	4.33		5.54	
	(high water)	Warm & Dry	0.02	14.32	-86.22	1.22	45.46	-78.13	3.87	34.89	705.98	196.22	3439.39
		Warm & Wet	0.09	30.20	-70.94	1.45	83.17	-59.99	3.98	14.28	229.91	176.01	3074.75
Hay Bay	1964	Base Case	0.49	147.60		2.29	274.57		4.26	532.32		43.40	
	(low water)	Warm & Dry	0.21	108.86	-26.24	1.69	261.80	-4.65	4.06	90.21	-83.05	946.79	2081.55
		Warm & Wet	0.26	121.20	-17.88	1.84	271.81	-1.01	4.13	83.69	-84.28	901.66	1977.55
	1978	Base Case	0.24	323.92		1.84	729.65		4.16	12.65		43.40	
	(high water)	Warm & Dry	0.30	126.36	-60.99	1.95	270.97	-62.86	4.18	31.49	148.83	946.79	2081.55
		Warm & Wet	0.37	136.04	-58.00	2.10	274.46	-62.39	4.24	31.67	150.29	901.66	1977.55
Presqu'ile	1965	Base Case	0.22	248.16		1.80	570.86		4.14	93.72		7.28	
Bay	(low water)	Warm & Dry	0.01	43.01	-82.67	0.90	143.65	-74.84	2.99	363.92	288.30	384.65	5183.72
		Warm & Wet	0.09	88.19	-64.46	1.41	248.08	-56.54	3.96	413.73	341.45	231.34	3077.69
	1978	Base Case	0.62	331.19		2.54	564.38		4.33	1.52		4.65	
	(high water)	Warm & Dry	0.15	167.39	-49.46	1.20	436.90	-22.59	3.12	111.37	7241.76	200.84	4221.02
		Warm & Wet	0.24	286.77	-13.41	1.78	660.75	17.07	4.11	59.35	3812.20	134.40	2791.57
LAKE ERIE													
Dunnville	1964	Base Case	0.07	187.42		1.34	548.38		3.92	273.25		81.70	
	(low water)	Warm & Dry	0.12	61.47	-67.20	1.50	164.35	-70.03	4.00	272.43	-0.30	1004.25	1129.13
		Warm & Wet	0.15	80.47	-57.07	1.60	203.74	-62.85	4.05	286.27	4.77	891.07	990.61
	1978	Base Case	0.50	424.96		2.34	779.87		4.29	6.36		44.58	
	(high water)	Warm & Dry	0.36	261.30	-38.51	2.05	536.00	-31.27	4.20	40.89	542.44	811.83	1721.23
		Warm & Wet	0.43	306.15	-27.96	2.18	594.80	-23.73	4.24	200.61	3052.33	450.58	910.80
Long Point	1964	Base Case	0.24	8381.76		1.89	18594.90		4.19	4648.38		3973.26	
	(low water)	Warm & Dry	0.21	5546.02	-33.83	1.79	12812.00	-31.10	4.14	15313.93	229.45	18008.93	353.25
		Warm & Wet	0.22	4695.52	-43.98	1.82	10702.80	-42.44	4.15	15351.56	230.26	14693.84	269.82
	1978	Base Case	0.64	6707.94		2.55	11369.10		4.32	142.01		2800.50	
	(high water)	Warm & Dry	0.42	14513.90	116.37	2.16	28432.00	150.08	4.24	955.12	572.59	10601.19	278.55
		Warm & Wet	0.47	14337.00	113.73	2.31	26702.80	134.87	4.29	2411.07	1597.86	5570.38	98.91
Rondeau	1964	Base Case	0.32	271.69		2.06	560.26		4.25	5.88		15.29	
Bay	(low water)	Warm & Dry	-0.06	158.17	-41.78	0.91	628.11	12.11	3.63	329.49	5500.63	84.06	449.82
		Warm & Wet	-0.05	206.27	-24.08	0.89	847.96	51.35	3.65	52.76	796.86	56.34	268.50
	1978	Base Case	0.70	45.31		2.59	75.09		4.30	0.67		11.76	
	(high water)	Warm & Dry	0.47	529.45	1068.45	2.32	984.35	1210.83	4.31	6.73	911.11	44.97	282.38
		Warm & Wet	0.60	536.17	1083.30	2.53	921.82	1127.56	4.35	3.40	411.11	31.64	169.05

Table 7.5 Weighted suitable areas (WSAs) (ha) and average habitat suitabilities (A/Suit) (range from 0=poor to 1=excellent) for different fish guilds and life stages evaluated under base case and warm & dry climate scenarios during low and high water level conditions

Wetland	Year	Climate Change Scenario	Water Level	Metric	Coolwater Piscivore			Coolwater Non-Piscivore			Warmwater Piscivore			Warmwater Non-Piscivore		
					Spawn	YOY	Adult	Spawn	YOY	Adult	Spawn	YOY	Adult	Spawn	YOY	Adult
LAKE ONTARIO																
Presqu'ile	1965	Base Case	74.04	A/Suit	0.20	0.19	0.48	0.31	0.43	0.42	0.35	0.31	0.44	0.37	0.29	0.46
Bay	(low water)			WSA	119.06	84.81	260.80	171.22	224.89	209.35	181.51	144.48	244.03	198.73	143.97	244.67
		Warm & Dry	73.29	A/Suit	0.22	0.21	0.52	0.35	0.47	0.46	0.42	0.35	0.48	0.43	0.33	0.52
				WSA	37.30	60.80	160.96	71.76	141.96	135.06	89.80	99.81	157.05	99.47	78.95	139.96
	1978	Base Case	74.92	A/Suit	0.18	0.18	0.48	0.29	0.42	0.41	0.34	0.29	0.44	0.36	0.28	0.45
	(high water)			WSA	127.14	97.56	306.97	188.25	262.07	243.28	210.34	170.07	287.13	225.50	164.22	282.76
		Warm & Dry	74.17	A/Suit	0.19	0.19	0.46	0.30	0.42	0.42	0.36	0.31	0.43	0.37	0.30	0.47
				WSA	69.13	63.76	172.06	97.80	154.23	147.09	119.09	102.28	164.39	126.02	98.44	166.10
LAKE ERIE																
Long Point	1964	Base Case	173.61	A/Suit	0.26	0.38	0.41	0.24	0.42	0.36	0.39	0.41	0.44	0.42	0.45	0.50
Bay	(low water)			WSA	3834.99	4010.98	3877.11	3090.74	4398.32	3876.87	4262.24	4109.39	4464.69	4918.44	5354.81	5587.86
		Warm & Dry	172.81	A/Suit	0.38	0.46	0.58	0.30	0.53	0.45	0.55	0.49	0.55	0.54	0.56	0.62
				WSA	3007.19	3337.66	3636.17	1898.54	3562.21	2807.71	3473.12	3419.86	3611.27	3531.45	3892.06	4048.85
	1978	Base Case	174.37	A/Suit	0.22	0.28	0.29	0.19	0.30	0.26	0.29	0.28	0.34	0.32	0.36	0.42
	(high water)			WSA	3226.44	3166.36	2246.56	2418.93	3164.76	2775.81	2828.47	2895.83	3584.82	3711.15	4336.73	5137.11
		Warm & Dry	173.57	A/Suit	0.28	0.37	0.46	0.25	0.43	0.37	0.44	0.40	0.45	0.45	0.48	0.53
				WSA	4747.04	3703.76	4461.85	3473.25	4541.45	4004.03	5235.12	3998.55	4625.64	5597.67	6023.66	6227.68
LAKE ST. CLAIR																
Mitchell's	1964	Base Case	174.37	A/Suit	0.26	0.09	0.29	0.17	0.23	0.21	0.31	0.13	0.25	0.29	0.18	0.17
Bay	(low water)			WSA	36.44	8.81	58.17	12.68	33.28	25.73	40.86	10.98	43.14	48.18	20.39	13.63
		Warm & Dry	173.37	A/Suit	0.16	0.05	0.19	0.04	0.14	0.11	0.17	0.05	0.14	0.20	0.08	0.05
				WSA	46.80	6.17	55.03	6.36	35.22	26.81	48.97	6.26	37.13	56.27	16.45	6.01
	1978	Base Case	175.27	A/Suit	0.24	0.10	0.28	0.16	0.23	0.21	0.29	0.14	0.25	0.28	0.17	0.17
	(high water)			WSA	62.01	15.94	89.48	26.83	53.38	44.75	72.56	22.28	70.45	73.80	38.99	30.34
		Warm & Dry	174.27	A/Suit	0.21	0.07	0.24	0.16	0.20	0.18	0.25	0.11	0.21	0.25	0.14	0.14
				WSA	44.31	8.06	56.00	9.42	36.81	27.83	47.50	8.81	39.71	56.03	18.08	8.58

increase in water levels compared to base case. As the rising water levels flood marsh vegetation, the total amount of vegetated area in the wetland decreased slightly to 45% (Table 7.3).

Under the three other scenarios projecting water level declines the amount of open water in the wetland decreased, resulting in 70% of the wetland vegetated under the not as warm & dry scenario, 73% under the warm & wet scenario and 76% under the warm & dry scenario. Whereas the extent of emergent vegetation decreased in these three scenarios in the low water year, emergent vegetation actually increased and expanded lakeward in the high water year. The greatest increase in emergent vegetation occurred under the not as warm & dry scenario (+36.1 ha, +28%) where water levels only dropped -0.46 m from base case. As water levels continued to decline, however, the increase in emergent vegetation became less. By the warm & dry scenario where water levels declined -0.75 m, there was only a +9.9 ha (+8%) increase in the area of emergent vegetation compared to base case. Similarly, the amount of meadow marsh and treed/shrub also expanded as water levels declined. There were notable changes of emergent vegetation to treed/shrub along the upland extent of the wetland under the not as warm & dry scenario (Figure 7.3); during this period the area of treed/shrub increased +20.4 ha (+20%) from base case. Meadow marsh and treed/shrub also visibly increased under the warm & dry scenario as meadow marsh increased +30.1 ha (+7,532%) and treed/shrub +37.4 ha (+4,510%) from base case (Figure 7.3; Table 7.3).

7.2.1.2 Bird Response

Low Water Year (1965)

In the low water year base case, the Presqu'ile wetland supported primarily marsh nesting bird guilds (obligate and generalists combined), 89% of the total index of abundance (Table 7.4). This was primarily due to the abundance of emergent marsh habitat relative to meadow and treed/shrub habitats in this scenario (Figure 7.1).

For the low water year at Presqu'ile, there were significant decreases in the modelled index of abundance for marsh nesting obligates and generalists, and increases in meadow marsh and treed/shrub nesting guilds under the climate change scenarios (warm & wet, warm & dry) compared to base case (Table 7.4). The marsh nesting obligate and generalist indices decreased -83% and -75%, respectively, under the warm & dry climate change scenario (Figure 7.5). These predicted decreases in abundance were largely the result of losses in area of emergent vegetation under these scenarios (Table 7.3). However, shallower breeding-season water depths also detrimentally affected density per hectare, under the climate change scenarios (Table 7.4). The abundance index of meadow marsh and treed/shrub nesting bird guilds increased in response to expansions in the area of these habitat types under the climate change scenarios. The distribution of marsh nesting birds was predicted to shift lakeward as the emergent marsh vegetation that occurred in the base case was replaced by meadow and treed/shrub habitats in the higher wetland elevations with lower water level conditions (Figure 7.2).

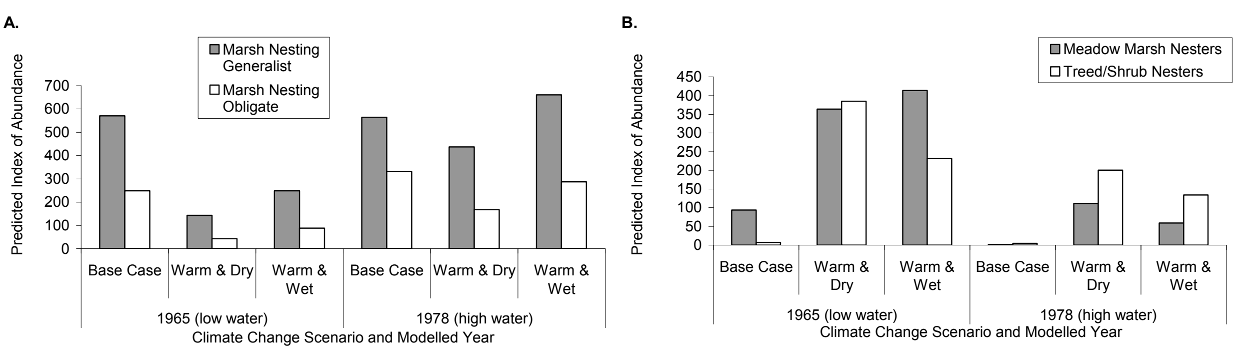

Figure 7.5 Modelled indices of abundance for marsh nesting (A), meadow marsh, and treed/shrub nesting (B) bird guilds under base case and climate change scenarios at Presqu'ile Bay during low water (1965) and high water (1978) conditions

High Water Year (1978)

In the high water year base case, the Presqu'ile wetland supported an even higher percentage of the total bird index of abundance as marsh nesting bird guilds, relative to the low water base case, 89% and 99%. This was primarily due to a reduction in the area of meadow marsh habitat between the low and high water level base cases (Table 7.4; Figures 7.1, 7.3).

During the high water year, there were also predicted decreases in the abundance index for the marsh nesting obligate (-49%) and marsh nesting generalist (-23%) guilds under the warm & dry scenario relative to base case (Table 7.4; Figure 7.5). Decreases in these marsh nesting guilds were due to a reduction in the densities per hectare because of shallower water depths in the wetland (Figure 7.4), as the area of emergent marsh actually increased under the warm & dry scenario (Table 7.3). The base case average breeding season water depth of 0.62 m was estimated to support approximately 2.5 birds per ha on average, compared to the warm & dry breeding season water depth at 0.15 m that supported an average of 1.20 birds per ha. Under the warm & wet scenario, the index of abundance for marsh nesting obligates (-13%) decreased and increased for marsh nesting generalists (+17%). The index of abundance for the marsh nesting generalist guild was less sensitive to breeding season water depth than the obligate guild, so the generalists responded positively to the larger area of emergent marsh despite the shallower average breeding season water depth. Both climate change scenarios supported higher abundance indices for the meadow marsh and treed/shrub habitat nesters (Figure 7.5) as a result of more area in these habitat types.

7.2.1.3 Fish

Low Water Year (1965) & High Water Year (1978)

Presqu'ile Bay, as part of Lake Ontario, is subject to water regulation. The effects of regulation were considered explicitly in water level scenarios provided from the IJC LOSLR Study. The total area flooded in low and high base case scenarios did not vary greatly, but inundated areas under climate change conditions were much smaller, and also varied more, between high and low scenarios than under base case conditions. When all fish species and their life stage requirements were evaluated under base case conditions using the composite scores, the area of high suitability habitat varied the most of the suitability categories between high and low regimes. High suitability habitat area was greatly reduced under climate scenarios, however, the proportion of medium suitability habitat increased under low water levels while total area decreased (Figures 7.2, 7.4). WSAs were reduced by roughly 90 ha during high periods and roughly 80 ha under low levels.

When WSAs were compared across guilds and life stages (Figure 7.6), a changed climate affected suitable habitat availability much more than high and low periods during the normal hydrocycle. This probably resulted from active regulation of the lake levels, which dampens natural variability.

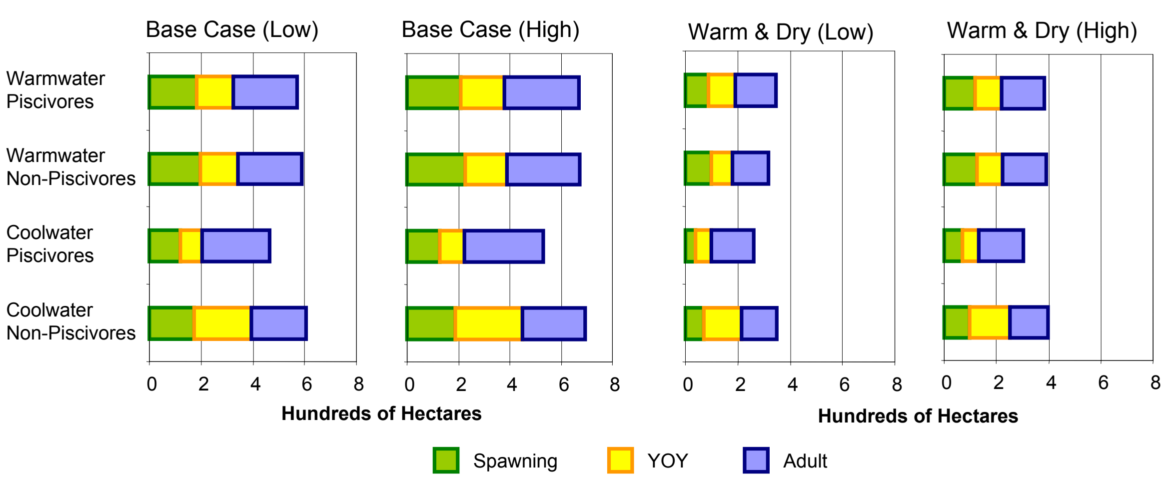

Figure 7.6 WSAs (in hundreds of hectares; areas x suitability rankings) for each of the guilds (feeding x thermal group) by life stage (spawning, YOY, or adult habitat). Fish guild membership is listed in Table 6.7 (Chapter 6). WSAs are shown for base case and warm & dry climate scenarios during low water (1965) and high water (1978) conditions and vegetation changes predicted for Presqu'ile Bay.

Differences between climate change and base case scenarios predicted the loss of guild-specific life stage habitats would range from 30 to 70% during comparable low water periods and 35 to 50% loss would occur under high water levels (Table 7.5). Coolwater spawning habitats decreased the most under a changing climate, although nursery habitat is generally considered to be the most limiting (Minns *et al.* 1999). The coolwater non-piscivore guild had the greatest proportional loss of YOY habitat and was considered the most sensitive to climate-induced habitat changes even though all life stage habitats were affected almost equally.

The changes in YOY habitat for coolwater non-piscivores (CL-NP) in Presqu'ile Bay are shown in Figure 7.7. Fringe emergent habitat was ranked as high suitability for this guild's life stage. Under historic fluctuations of water levels and habitat changes, the weighted suitable area for CL-NP YOY varied between 262 and 225 ha with subtle changes in high suitability habitat and new areas of water inundation. Under climate scenarios, high suitability habitat is reduced but medium suitability habitats expand, somewhat compensating for the loss of flooded area. In this case, weighted suitable habitat ranged from 140 and 155 ha during low and high water level cycles (Table 7.5). Because high suitability habitat probably contributes disproportionately to overall productivity, the trade-off is probably not equitable for the fishes using Presqu'ile Bay.

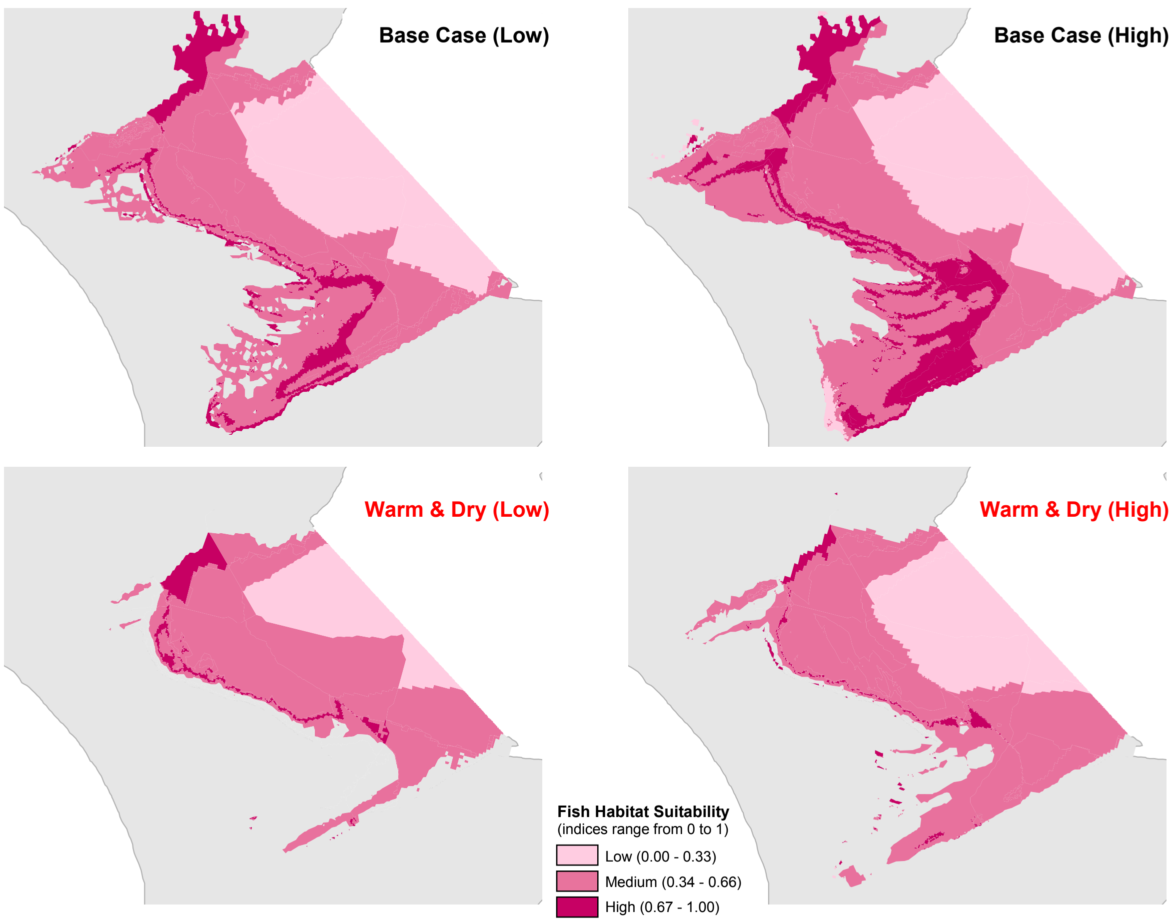

Figure 7.7 Habitat suitabilities (low, medium and high) for the YOY life stage of coolwater non-piscivore fish species listed in Table 6.7 (Chapter 6). This life stage and guild were the most sensitive of the fish groups assessed in Presqu'ile Bay. Habitat suitabilities were mapped for base case and warm & dry climate scenarios during low water (1965) and high water (1978) conditions and vegetation changes predicted for this area.

7.2.2 Hay Bay

7.2.2.1 Vegetation Response

Low Water Year (1962)

In the low water year base case, the wetland was dominated by meadow marsh (54%), a similar amount of water and emergent occurred (20% and 24% respectively), and very little treed/shrub (Table 7.3; Figure 7.8). The not as warm & wet scenario on Lake Ontario resulted in a small increase in water levels (+0.06 m). Vegetation distribution and area at Hay Bay essentially remained the same as the base case with a modest increase in open water (+8.4 ha, +16%) while areas of emergent, meadow and treed/shrub declined slightly (Table 7.3).

Under the three other climate change scenarios (not as warm & dry, warm & wet, warm & dry), open water was lost; by the warm & dry conditions, open water had decreased by -79% being replaced by wetland vegetation, predominantly treed/shrub. Due to the dry upland conditions, treed/shrub also replaced meadow marsh which decreased roughly -120.0 ha (-84%) in all three scenarios. The meadow marsh community passed a critical threshold between the base case and the not as warm & dry scenario where most of the meadow marsh area changed to treed/shrub in higher elevation areas of the wetland. During this period, the area of treed/shrub increased by +147.4 ha (+1,902%) from base case (Table 7.3). The area of emergent vegetation remained relatively consistent under the three drier climate scenarios but its occurrence moved from shallow areas along the river banks to the mouth of the river and into the lake (Figure 7.8).

High Water Year (1978)

Reflecting the wet conditions within the wetland in the high water year base case, emergents and open water dominated (65% and 31% respectively), with less than 5% of the area in meadow marsh and treed/shrub (Table 7.3; Figure 7.9).

Under the not as warm & wet scenario with a slight increase in water levels, the amount of open water increased slightly, and the amount of emergent, meadow and treed/shrub contracted slightly (Table 7.3; Figure 7.9).

With the three scenarios projecting lake level declines, the area of open water continually decreased and by the warm & dry scenario, the area had decreased -55.6 ha (-68%). Emergent vegetation decreased dramatically as well (approximately -111.0 ha or -63% under all three scenarios) and occurred in shallow areas along the river bank and in the river delta. As water levels declined between the base case and not as warm & dry scenario, emergent vegetation passed a critical threshold where most of the community changed to treed/shrub (Figure 7.9). Meadow marsh increased slightly (approximately +5.0 ha or +150%) under all three scenarios, though treed/shrub dramatically increased +147.4 ha (+1,902%) under the not as warm & dry scenario and +161.3 ha (+2,081%) under the warm & dry scenario compared to base case in higher areas in the wetland. As water levels declined in warm & dry conditions, there was a shift of wetland vegetation composition within the wetland. Treed/shrub was the most dominate community in the wetland (63%), followed by emergents at 24%, open water 10% and meadow marsh 3% (Table 7.3; Figure 7.9). It is interesting to note that the area of treed/shrub under the low and high water level conditions were similar for all four water level scenarios. This could have been due to the lack of detailed elevation information in upland areas of the wetland or because the lower elevation threshold for the existence of treed/shrub in the wetland was reached.

7.2.2.2 Bird Response

Low Water Year (1962)

In the low water year base case, the wetland breeding bird index of abundance was primarily composed of marsh birds. The index was similar for marsh and meadow marsh nesting guilds, representing 42% and 53% of the total abundance respectively (Table 7.4).

The area of emergent marsh habitat remained relatively unchanged under the base case and climate change scenarios during the low water year at Hay Bay. Decreases relative to base case in the abundance index for marsh nesting obligates under the warm & dry (-26%) and warm & wet (-5%) scenarios were the result of lower breeding season water depths (Table 7.4). Smaller decreases were also predicted for marsh nesting generalists, -18% and -1% for the warm & dry and warm & wet scenarios, respectively. Large decreases in meadow marsh nesters, and increases in treed/shrub nesters were also predicted (Figure 7.10) as a large area of the modelled wetland shifted from meadow marsh under the base case, to treed/shrub habitat under the climate change scenarios (Figure 7.8).

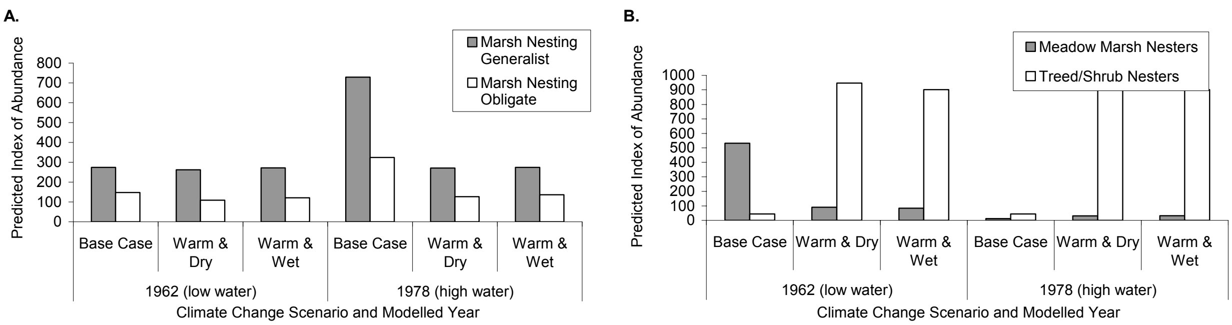

Figure 7.10 Predicted indices of abundance for marsh nesting (A), meadow marsh and treed/shrub nesting (B) bird guilds under base case and climate change scenarios at Hay Bay during low water (1962) and high water (1978) conditions

High Water Year (1978)

The wetland breeding bird community reflected a low availability of meadow marsh due to the wet conditions (Figure 7.10), within the wetland in the high water year base case, marsh nesting guilds made up 95% of the total wetland bird index of abundance for Hay Bay (Table 7.4).

The large shift in emergent marsh habitat area under base case, to treed/shrub habitat under the climate change scenarios results in large decreases in the marsh nesting guild indices of abundance (-58% to -63%). The distribution of marsh nesting obligate birds also becomes restricted to emergent marsh habitat associated with the river channel under the climate change scenarios (Figure 7.9). The index of abundance for treed/shrub nesters increases from 43.4 under base case to over 900 under the climate change scenarios. The meadow marsh nester index increases from 12.7 under base case to approximately 31 under the climate change scenarios (Table 7.4). These results indicate that within drowned river-mouth wetlands such as Hay Bay, large shifts in habitat and associated wildlife communities can be expected as the floodplain hydrologic conditions change.

7.2.3 Lynde Creek

7.2.3.1 Vegetation Response

Low Water Year (1959)

For the base case low water year, the modelled vegetation was predominantly emergent (44%) in the lower portion of the system and meadow marsh (54%) in the drier, upper portion of the wetland. There was little treed/shrub (1.0 ha) and a small amount of water (0.2 ha) (Figure 7.11; Table 7.3).

As water levels rose slightly under the not as warm & wet scenario (+0.06 m), there were marginal increases in open water and emergent vegetation in the wetland and small decreases in meadow marsh and treed/shrub vegetation (Table 7.3).

As water levels declined in the other three scenarios, open water virtually disappeared in the wetland as only 0.02 ha remained. The amount of emergent vegetation decreased in the wetland as the community transitioned to drier meadow marsh. In fact, there was a -99% (-24.4 ha) reduction in the area of emergent vegetation, particularly in the southern creek portion of the wetland, under the warm & dry scenario compared to base case. Although there were substantial increases in meadow marsh in the southern portion of the creek, there were net losses in meadow marsh in the three most extreme scenarios because meadow

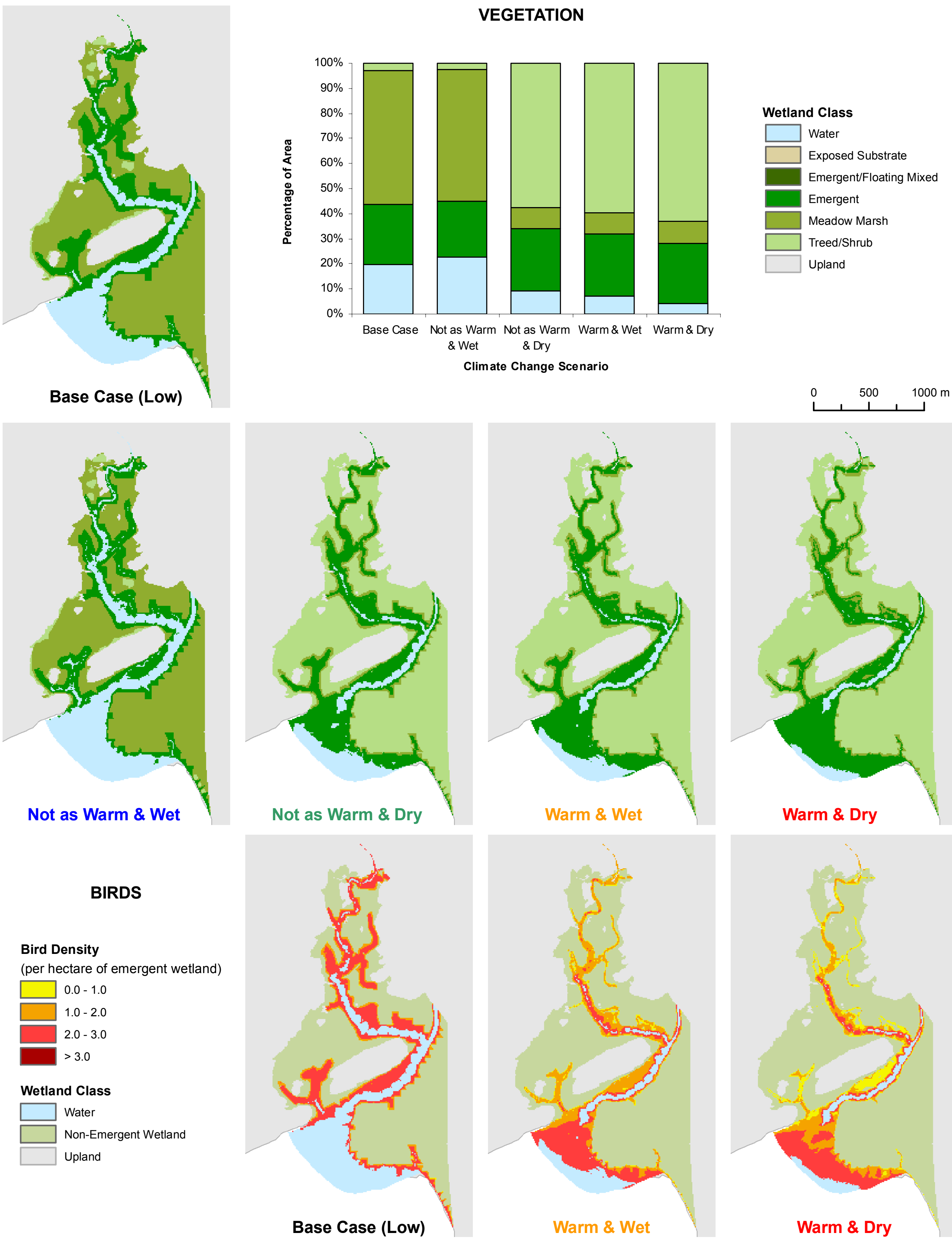

Figure 7.8 Predicted wetland class distribution and marsh nesting obligate bird distribution under base case and climate change scenarios at Hay Bay during low water (1962) conditions

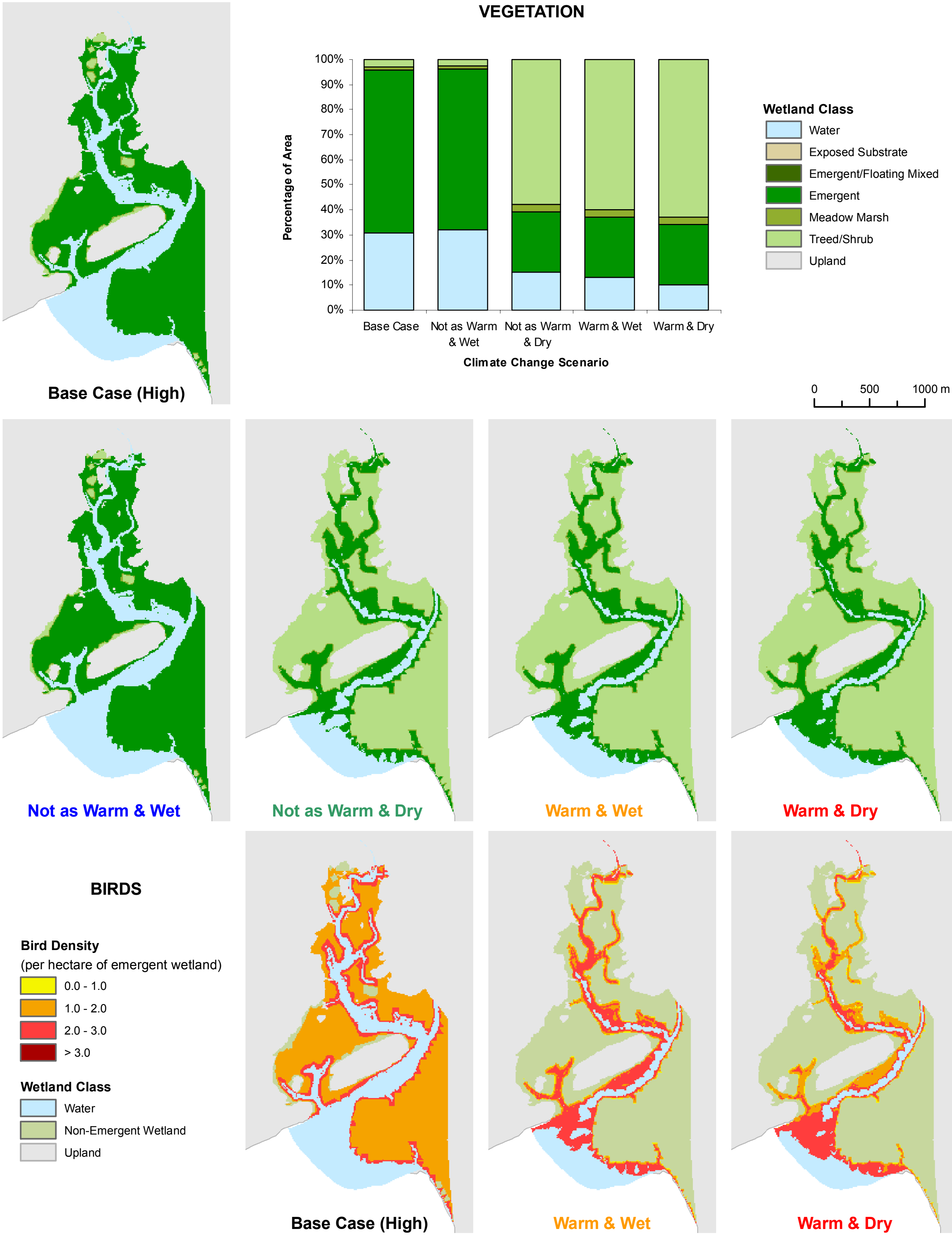

Figure 7.9 Predicted wetland class distribution and marsh nesting obligate bird distribution under base case and climate change scenarios at Hay Bay during high water (1978) conditions

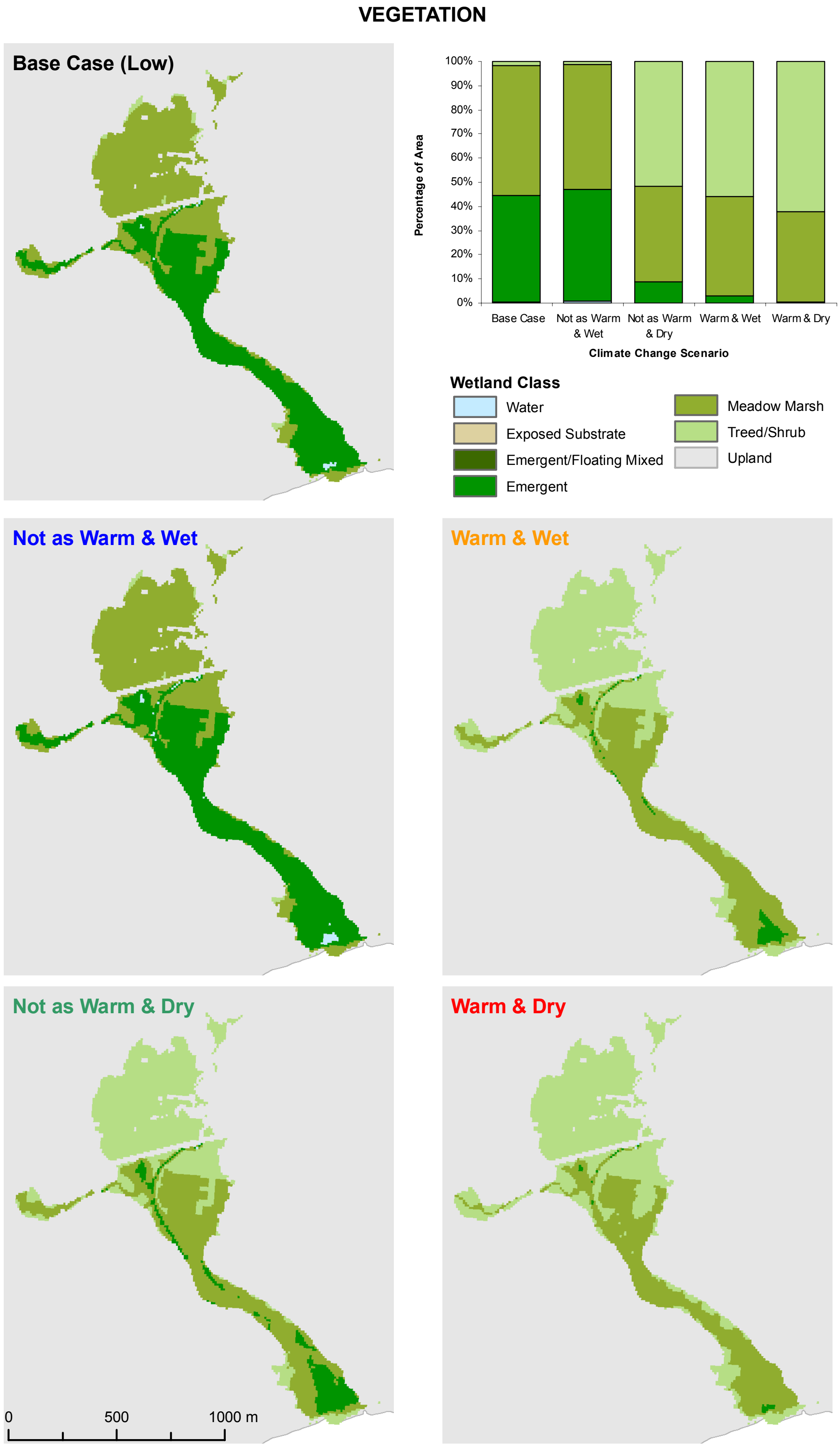

Figure 7.11 Predicted wetland class distribution under base case and climate change scenarios at Lynde Creek during low water (1959) conditions

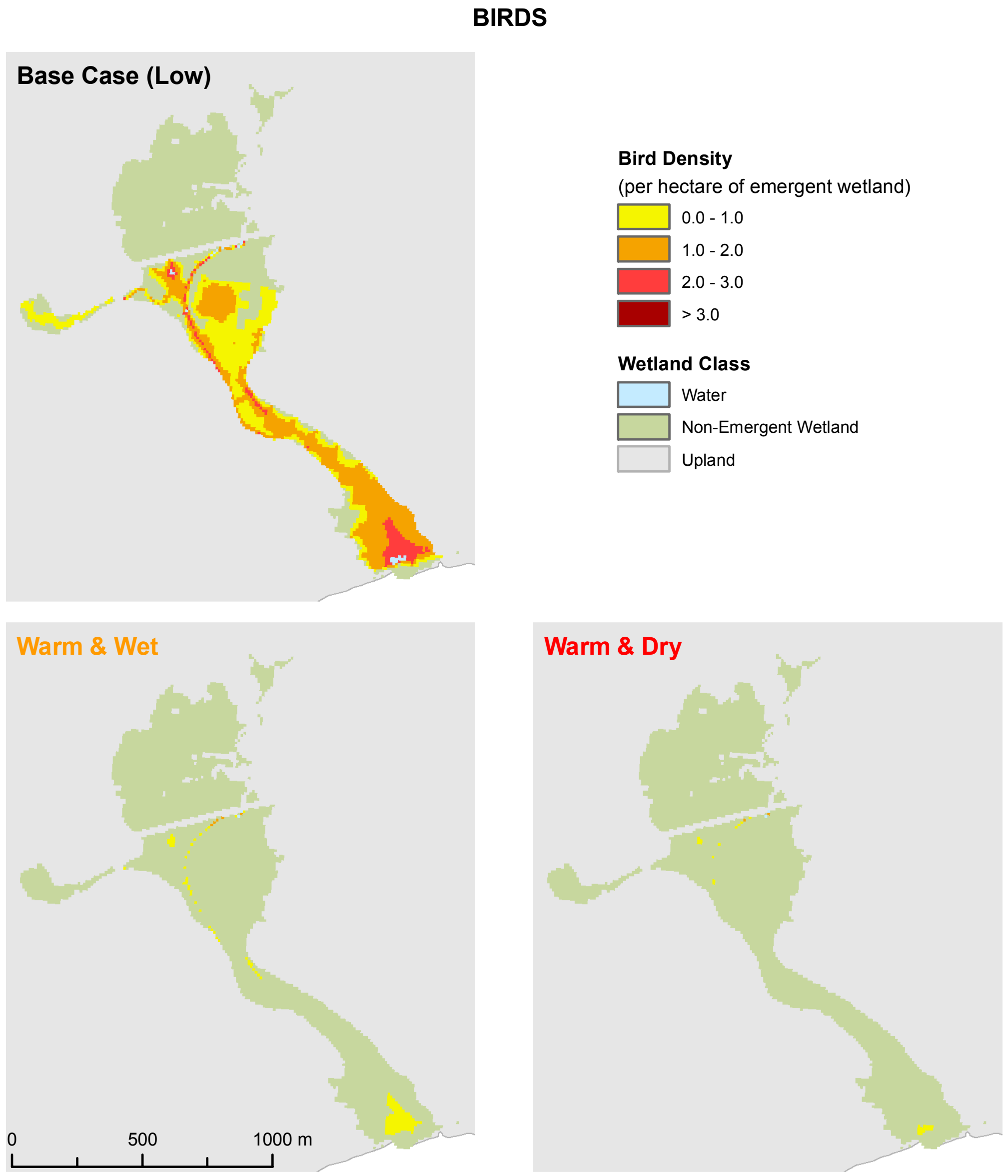

Figure 7.12 Predicted marsh nesting obligate bird distribution under base case and climate change scenarios at Lynde Creek during low water (1959) conditions

marsh in the northern portion of the wetland changed to treed/shrub. Meadow marsh had a net loss of -9.4 ha (-31%) under the most extreme scenario (warm & dry) compared to base case. Treed/shrub became dominant in the drier and higher elevation upper portion of the wetland and gradually expanded as water levels dropped in the not as warm & wet, warm & wet and warm & dry scenarios. Therefore, in the warm & dry scenario, the wetland primarily consisted of treed/shrub (62%) and meadow marsh (37%); there was little open water and emergent vegetation remaining.

High Water Year (1978)

The modelled vegetation communities for the high water year base case for Lynde Creek showed a wetland dominated throughout by emergent vegetation (87%) with small areas of treed/shrub and meadow marsh in the upper portion of the wetland accounting for another 4% of the total wetland area (Figure 7.13).

Under the not as warm & wet scenario projecting a slight rise in water levels, the area of open water increased (+2.9 ha, +60%). Conversely, the area of emergent, meadow marsh and treed/shrub all decreased marginally in this scenario.

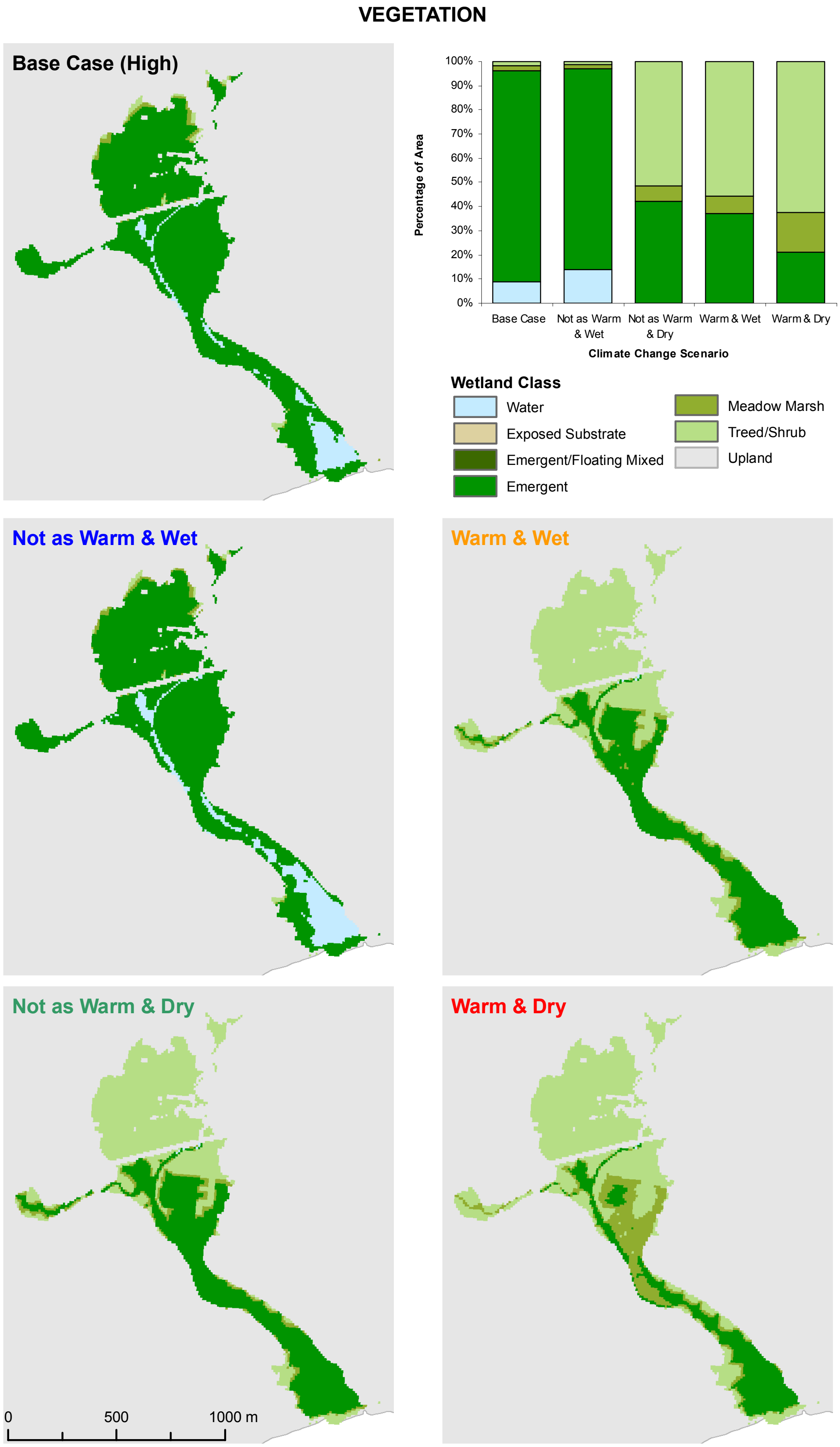

Figure 7.13 Predicted wetland class distribution under base case and climate change scenarios at Lynde Creek during high water (1978) conditions

BIRDS

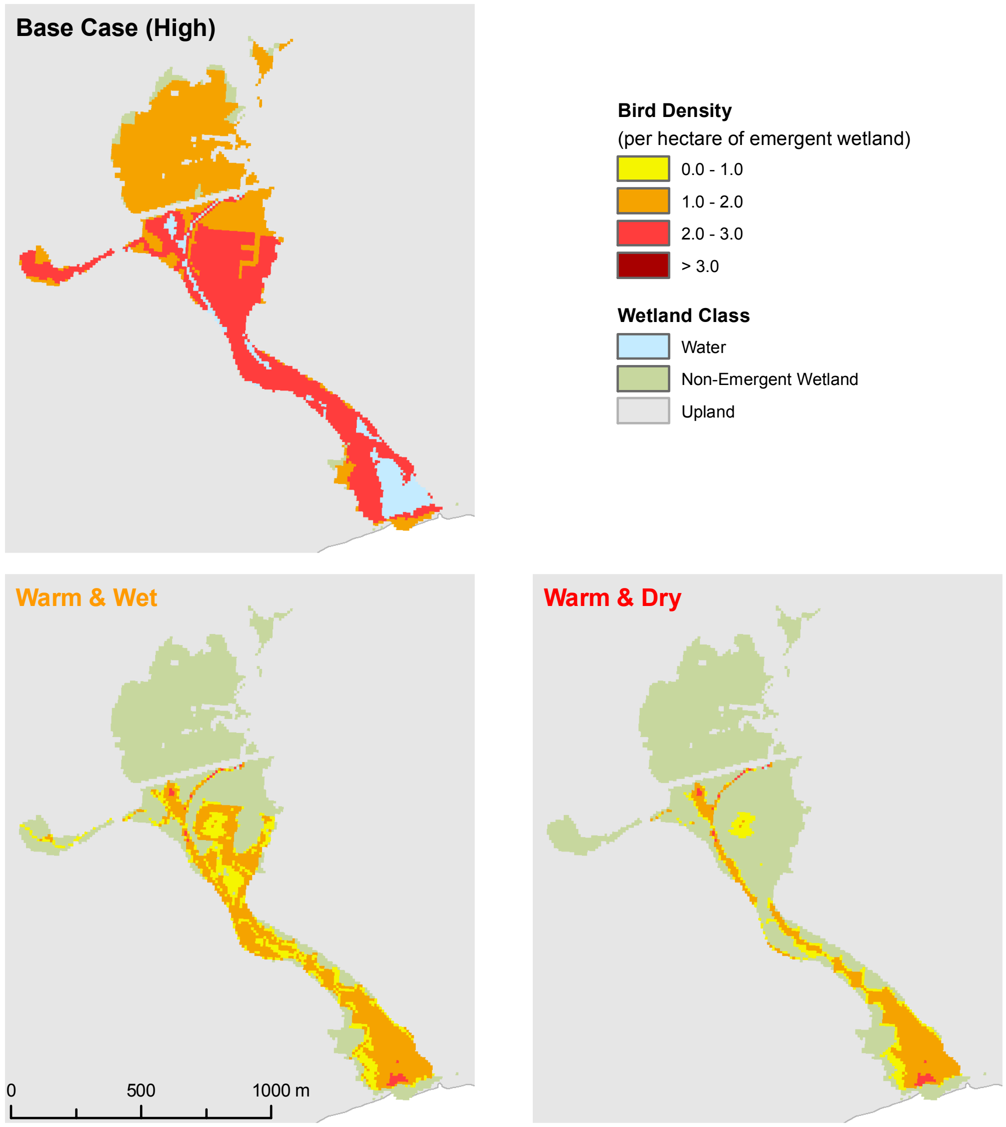

Figure 7.14 Predicted marsh nesting obligate bird distribution under base case and climate change scenarios at Lynde Creek under high water (1978) conditions

As water levels declined under the three other scenarios, there was a dramatic reduction in the amount of open water; by the warm & dry scenario only 0.02 ha of open water remained (a loss of -4.9 ha or -100%). There was also a reduction in emergent vegetation under the three drier scenarios. There was a significant change in emergent vegetation to treed/shrub in the northern portion of the wetland in the not as warm & dry scenario compared to base case (Figure 7.13). There were further reductions in emergent vegetation under the other two scenarios as meadow marsh and treed/shrub expanded. The largest increase in meadow marsh and treed/shrub was under the warm & dry scenario, where there were increases of +8.3 ha (+706%) and +34.0 ha (+3,439%) from base case, particularly in the centre of the creek basin (Table 7.3; Figure 7.13). Similar to Hay Bay, changes in treed/shrub were exactly the same as for the low water year. The wetland passed a critical threshold for the lower limit of treed/shrub vegetation. Again, this could have been due to the lack of detailed elevation information in upland areas.

7.2.3.2 Bird Response

Low Water Year (1959)

In the low water year base case, the wetland breed bird index of abundance was similar for marsh and meadow marsh nesting guilds, representing 52% and 46% of the total abundance respectively (Table 7.4; Figure 7.12).

Under the climate change scenarios, there were large decreases in the estimated indices of abundance for both marsh nesting bird guilds, and a large increase in the treed/shrub nesting guild (Figure 7.15) relative to base case. The 95% to 100% reduction in the marsh nesting guilds was a result of meadow marsh almost completely replacing the base case emergent marsh areas under the climate change scenarios. Increases in treed/shrub nester guild index from 5.5 under base case, to 196 under warm & dry, and 176 under warm & wet scenarios were the result of upper elevations within the wetland converting from meadow marsh to treed/shrub habitat. The meadow marsh bird guild index of abundance decreased under the climate change scenarios; however, the change was relatively small in comparison to the other habitats and bird guilds (Figure 7.15).

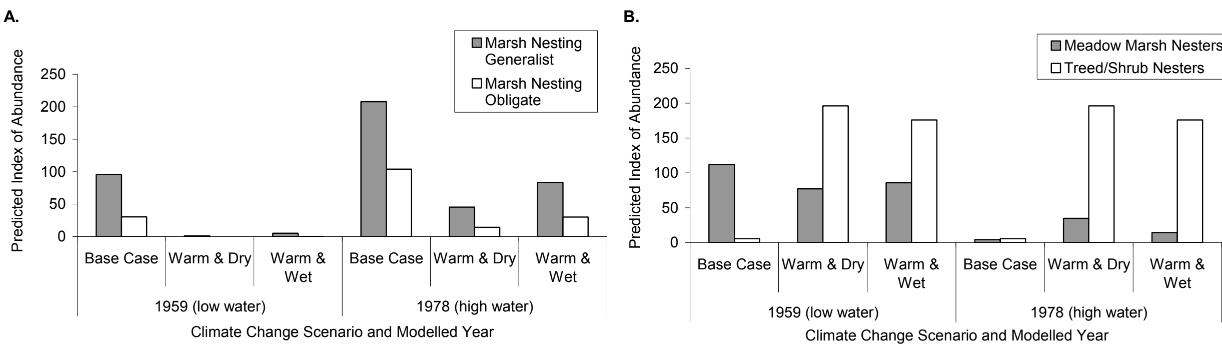

Figure 7.15 Predicted indices of abundance for marsh nesting (A), meadow marsh and treed/shrub nesting (B) bird guilds under base case and climate change scenarios at Lynde Creek during low water (1959) and high water (1978) conditions

High Water Year (1978)

The wetland breeding bird community reflected a low availability of meadow marsh due to the wet conditions (Figures 7.13, 7.14), within the wetland in the high water year base case, marsh nesting guilds made up 97% of the total wetland bird index of abundance for Lynde Creek (Table 7.4).

Under the climate change scenarios, predicted reductions in the abundance indices of marsh nesting guilds (-60% to -86%) also occurred at Lynde Creek during the high water year, as a result of decreases in emergent marsh area (Figure 7.13) and bird abundance index density per hectare (Table 7.4) relative to base case. In the high water year climate change scenarios, treed/shrub habitat and associated bird guild expanded into the same area as predicted under the base case low water year. The treed/shrub habitat replaced emergent marsh that occurred in upper elevations under the base case scenario (Figure 7.13). As a result, the distribution and density of marsh nesting obligates decreased under the climate change scenarios relative to base case (Figure 7.14). The meadow marsh nesting guild index of abundance increased from 4.3 under base case, to 34.9 and 14.3 under the warm & dry and warm & wet scenarios, respectively (Table 7.4).

7.2.4 South Bay

7.2.4.1 Vegetation Response

Low Water Year (1962)

The vegetation modelling results for the low water year base case showed a wetland dominated by emergent vegetation, comprising 57% of the wetland, with small areas of meadow marsh (14%) and treed/shrub (3%) in the upper, landward edges. The remaining wetland consisted of open water (Figure 7.16).

For the not as warm & wet scenario, the area of open water and emergent vegetation increased slightly, while meadow marsh and treed/shrub decreased with the projected lake level rise.

For the other three scenarios, the area of open water decreased substantially. There was approximately a -15 ha (-99%) decrease in open water under all three scenarios compared to base case. In fact, under the warm & dry scenario, no open water remained at South Bay. Open water areas progressively filled with emergent vegetation, but overall there was a net loss of emergents as the vegetation transitioned to meadow marsh and treed/shrub along the wetland/upland boundary. Emergent vegetation decreased by -3.0 ha (-10%) under the not as warm & dry scenario and -10.1 ha (-32%) under the warm & dry scenario from base case; the defined study area boundary in the GIS-based vegetation model may have precluded a lakeward expansion of emergent vegetation in the wetland. Meadow marsh increased under the three drier scenarios, experiencing the greatest increase under the not as warm & dry scenario compared to base case. Meadow marsh actually decreased in area under the two warm scenarios compared to the not as warm & dry scenario, likely due to the expansion of treed/shrub lakeward into meadow marsh habitat (Figure 7.16; Table 7.3).

High Water Year (1978)

For the base case high water year, the modelled vegetation was predominantly emergent (50%) and open water (45%). The remaining 5% of the wetland area was equally composed of meadow marsh and treed/shrub (Figure 7.17).

As water levels rose slightly under the not as warm & wet scenario (+0.06 m), there was a marginal increase in open water and small decreases in emergent, meadow marsh, and treed/shrub vegetation in the marsh (Table 7.3)

As water levels declined under the three drier scenarios, there was nearly a -25.0 ha (96%) decline in open water by the warm & dry scenario. The area of emergent vegetation decreased marginally (-1.1 ha, -4%) under the not as warm & wet scenario despite the decline in water levels by -0.46 m. With the warm & wet and warm & dry scenarios, emergent vegetation increased slightly compared to base case; however, the area of emergent vegetation actually decreased as water levels declined between the warm & wet and water & dry scenarios. Meadow marsh increased in the wetland as water levels declined. Notable changes occurred under the not as warm & dry scenario compared to base case as this community increased +7.7 ha (+579%). Under the warmer scenarios, the amount of meadow vegetation increased relative to base case, but the absolute area actually declined compared to the not as warm & dry scenario. As water levels declined, treed/shrub vegetation continually expanded (+21.3 ha, +1,499%) into areas previously vegetated with emergent and meadow marsh communities (Figure 7.17; Table 7.3). Similar to Hay Bay and Lynde Creek, changes in the treed/shrub were the same as in the low water year, likely due to the lack of detailed elevation information in the upland areas.

7.2.5 Lake Ontario Summary

For all Lake Ontario wetlands, there was a distinct difference in the wetland class responses to the not as warm & wet climate change scenario, which projected a minimal (+6 cm) increase in annual average water level, and the three other scenarios that projected a decrease in levels ranging from -46 to -75 cm. With the slight increase in water level, area of open water showed a small increase for both low and high initial conditions. Meadow marsh and treed/shrub decreased, but emergent vegetation showed very small increases and decreases under highs and lows.

The opposite responses occurred when water level declined. The area of open water decreased and treed/shrub area increased in both high and low initial conditions. As the water level decline became more severe in each scenario, the area of treed/shrub progressively expanded. Emergents, under high water conditions, decreased in riverine environments, and increased in lacustrine conditions, while in low water conditions no consistent response was apparent. While meadow marsh increased in all wetlands under high initial conditions, its response was dependent on hydrogeomorphic form under low initial conditions – i.e. meadow marsh increased in lacustrine sites and decreased in riverine sites.

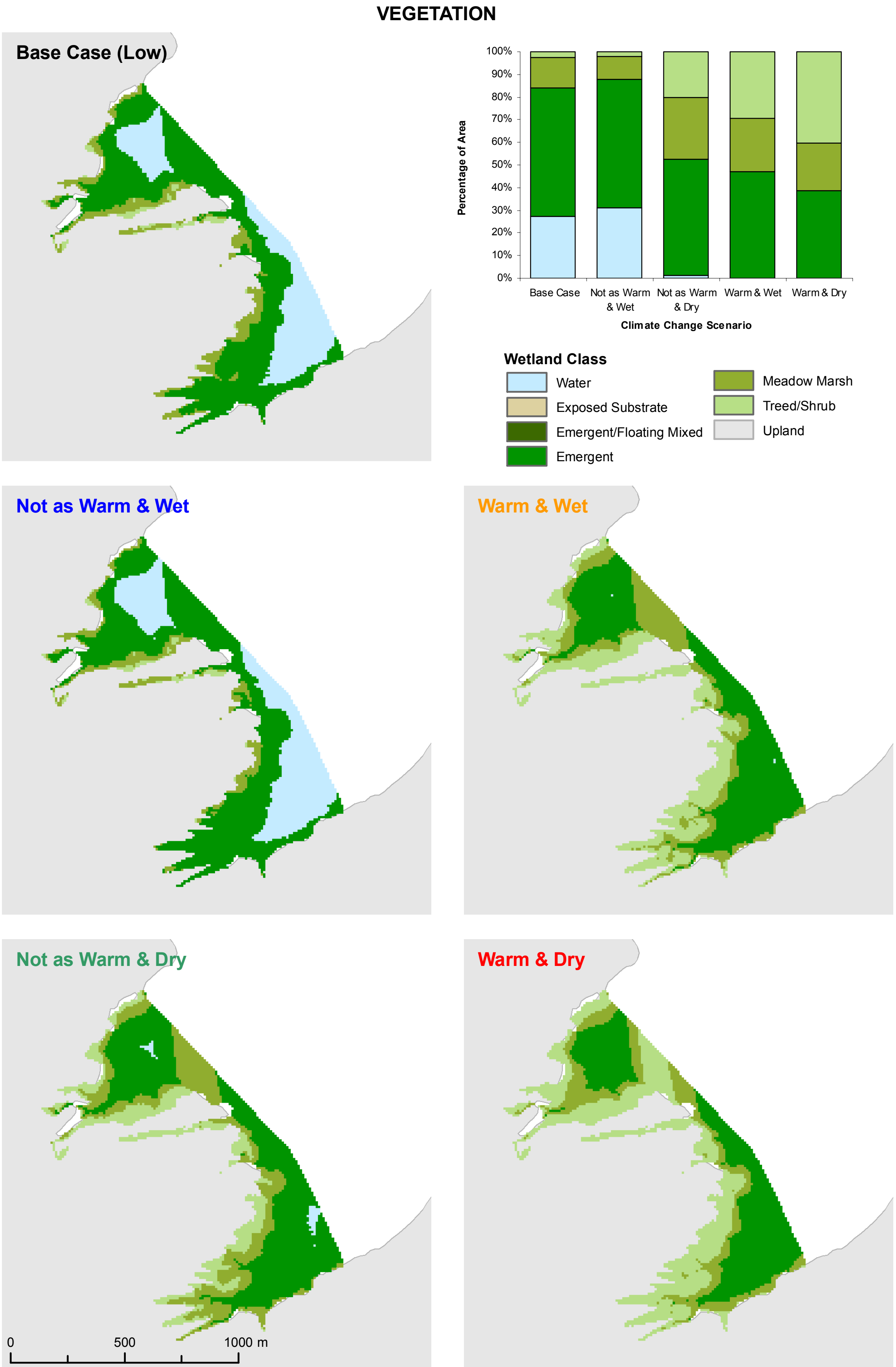

Figure 7.16 Predicted wetland class distribution under base case and climate change scenarios at South Bay during low water (1962) conditions

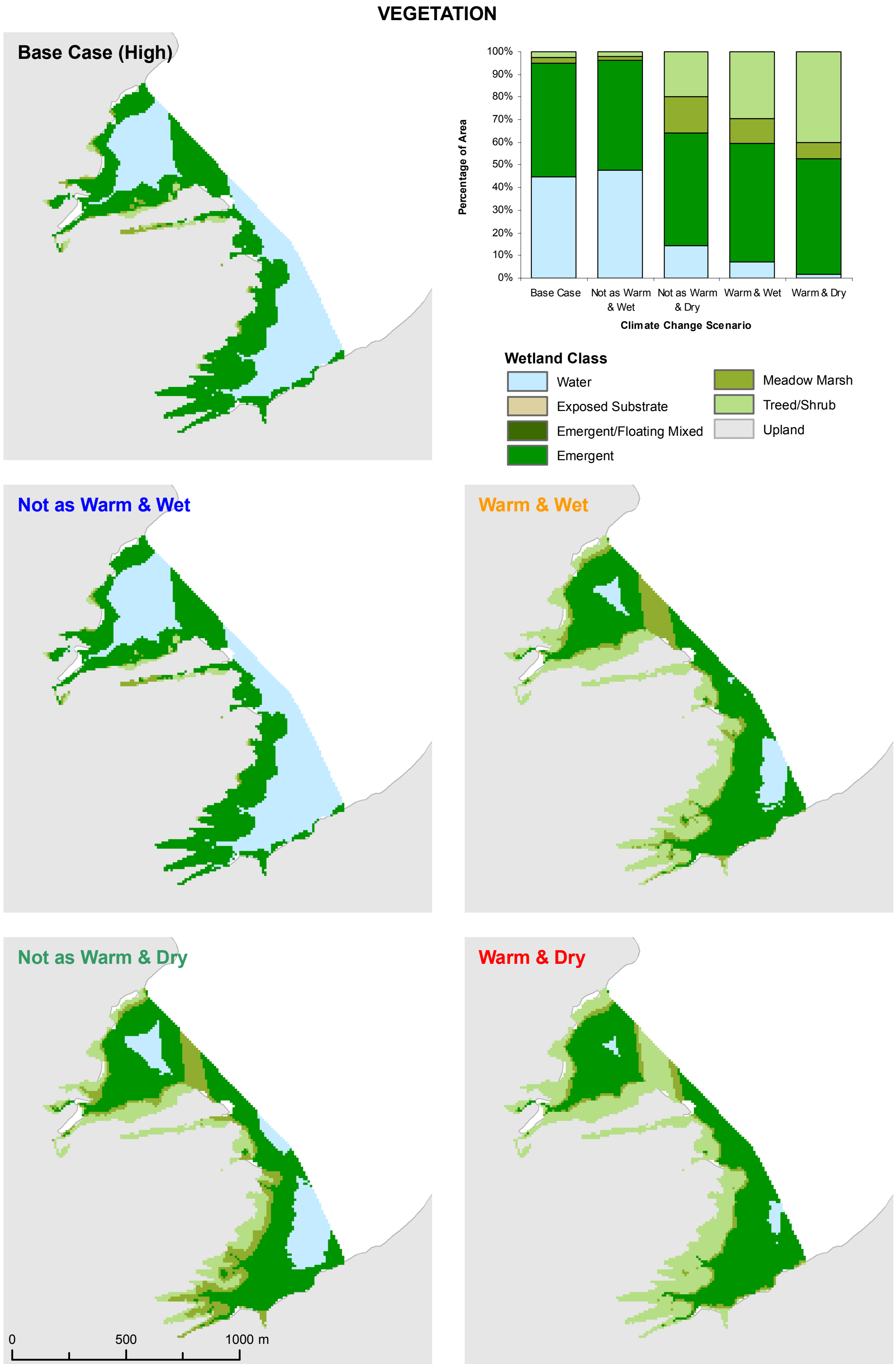

Figure 7.17 Predicted wetland class distribution under base case and climate change scenarios at South Bay during high water (1978) conditions

The bird modelling explored the two extreme temperature and precipitation scenarios – warm & wet and warm & dry. The abundance of marsh nesting obligate and generalist guilds decreased in the warm & dry scenario across all wetlands while treed/shrub nesting guilds increased. In the warm & wet scenario, abundance of marsh nesting obligates decreased and treed/shrub nesting guilds increased in all wetlands. However, the abundance of marsh nesting generalists and meadow marsh nesters did not exhibit a consistent pattern.

The water level scenarios (high and low years) were tested using fish modelling in Presqu'ile Bay only for the extreme warm & dry climate scenario. There were marked differences between historic (base case) and the climate scenario, regardless of whether water levels were low or high. Loss of habitat quantity under climate change affected the habitat supply for the coastal fish guilds and life stages evaluated, more so than the average habitat suitability in Presqu'ile. Although the loss of high quality habitat was apparent under climate scenarios, moderate suitability habitat expanded in area compensating for the loss. Because vegetated habitat changed to a greater extent in some of the other wetlands evaluated on Lake Ontario in the vegetation models (e.g. Lynde Creek), the responses predicted for the Presqu'ile fish assemblage could be more pronounced in Lynde Creek and Hay Bay drowned river-mouths. The responses predicted for South Bay vegetation were similar to Presqu'ile Bay, therefore similar fish responses might be expected in both areas. The loss of high suitability habitat would be detrimental to productivity in all cases.

7.3 Lake Erie Climate Change Assessment

Responses to climate change water level scenarios were modelled for Long Point, Turkey Point, Dunnville, and Rondeau wetlands and included an assessment of effects on vegetation communities, bird abundance, and fish habitat. The modelling results were stratified by low and high water level case studies.

7.3.1 Long Point and Turkey Point

7.3.1.1 Vegetation Response

There were notable changes in the Long Point and Turkey Point wetlands under the four climate change scenarios (refer to Table 7.6; Figures 7.18, 7.20). Key vegetation changes are summarized for the selected low and high water years.

Low Water Year (1964)

For the modelled low water base case year, Long Point was predominately open water (73%). The remaining wetland area consisted of emergent vegetation (19%), meadow marsh (5%) and treed/shrub (3%). The amount of open water at the Turkey Point marsh was notably lower, covering only 30% of the total area. Turkey Point was dominated by vegetated communities, including emergent vegetation (42%), meadow marsh (23%) and treed/shrub (5%).

In the low water year at Long Point, the modelled extent of open water gradually decreased under all four climate change scenarios. From the base case to the warm & dry climate change scenario, the area of open water decreased by -4,042.8 ha (-23%) but still accounted for 56% of the total wetland area. As water levels declined, drier wetland vegetation communities expanded, migrating lakeward. Under the not as warm & wet scenario, emergent vegetation increased in extent, expanding into shallow open water areas. There were further migrations of emergent vegetation lakeward as water levels declined under the three more extreme scenarios, but there was a net loss in the community as emergent areas in the Inner Bay, Company Marshes, and Outer Peninsula transitioned to meadow marsh during the not as warm & dry scenario; the area of meadow marsh increased by +2,854.7 ha (+227%) under this scenario compared to base case. There were also substantial increases in treed/shrub for the three drier scenarios, increasing +1,441.7 ha (203%) under the not as warm & dry scenario and +2,506.4 ha (+353%) under the warm & dry scenario (Table 7.6). The community migrated into areas of meadow marsh as moisture conditions declined (Figure 7.18).

Similar changes in wetland communities occurred at Turkey Point. There was a greater loss in open water compared to Long Point under the four climate change scenarios. Here, the amount of open water decreased by -496.3 ha (-91%) from base case conditions to the most extreme warm & dry scenario. Open water

Table 7.6 Wetland class area (ha) projected under climate change scenarios, change in area (Δ ha), and percent change (%Δ) from base case for Lake Erie

Wetland	Year	Climate Change Scenario	Water			Exposed Substrate			Emergent/Floating			Emergent			Meadow Marsh			Treed/ Shrub		
			ha	Δ ha	% Δ	ha	Δ ha	% Δ	ha	Δ ha	% Δ	ha	Δ ha	% Δ	ha	Δ ha	% Δ	ha	Δ ha	% Δ
Dunnville	1965	Base Case	239.73			0.00			9.22			140.01			73.85			14.59		
	(low water)	NWW	222.49	-17.24	-7.19	0.00	-	-	8.71	-0.51	-5.53	90.18	-49.83	-35.59	107.94	34.09	46.16	48.08	33.49	229.54
		NWD	192.71	-47.02	-19.61	0.00	-	-	4.89	-4.33	-46.96	56.53	-83.48	-59.62	83.13	9.28	12.57	140.14	125.55	860.52
		WW	186.34	-53.39	-22.27	0.00	-	-	4.28	-4.94	-53.58	50.29	-89.72	-64.08	77.37	3.52	4.77	159.12	144.53	990.61
		WD	179.88	-59.85	-24.97	0.00	-	-	3.48	-5.74	-62.26	41.08	-98.93	-70.66	73.63	-0.22	-0.30	179.33	164.74	1129.13
	1978	Base Case	285.77			0.00			0.00			181.95			1.72			7.96		
	(high water)	NWW	254.13	-31.64	-11.07	0.00	-	-	0.00	-	-	209.12	27.17	14.93	2.35	0.63	36.63	11.80	3.84	48.24
		NWD	210.98	-74.79	-26.17	0.00	-	-	0.00	-	-	191.78	9.83	5.40	8.37	6.65	386.63	66.27	58.31	732.54
		WW	202.42	-83.35	-29.17	0.00	-	-	0.00	-	-	140.30	-41.65	-22.89	54.22	52.50	3052.33	80.46	72.50	910.80
		WD	193.75	-92.02	-32.20	0.00	-	-	0.00	-	-	127.63	-54.32	-29.85	11.05	9.33	542.44	144.97	137.01	1721.23
Long Point	1964	Base Case	17294.48			0.00			0.00			4441.40			1256.32			709.51		
	(low water)	NWW	16604.49	-689.99	-3.99	0.00	-	-	0.00	-	-	4554.15	112.75	2.54	1717.23	460.91	36.69	825.84	116.33	16.40
		NWD	14769.04	-2525.44	-14.60	0.00	-	-	0.00	-	-	2670.48	-1770.92	-39.87	4110.98	2854.66	227.22	2151.21	1441.70	203.20
		WW	14352.49	-2941.99	-17.01	0.00	-	-	0.00	-	-	2576.25	-1865.15	-41.99	4149.07	2892.75	230.26	2623.90	1914.39	269.82
		WD	13251.65	-4042.83	-23.38	0.00	-	-	0.00	-	-	3095.28	-1346.12	-30.31	4138.90	2882.58	229.45	3215.88	2506.37	353.25
	1978	Base Case	20529.78			0.00			0.00			2633.46			38.38			500.09		
	(high water)	NWW	17957.15	-2572.63	-12.53	0.00	-	-	0.00	-	-	5111.60	2478.14	94.10	46.45	8.07	21.03	586.51	86.42	17.28
		NWD	16219.26	-4310.52	-21.00	0.00	-	-	0.00	-	-	6534.46	3901.00	148.13	63.21	24.83	64.70	884.78	384.69	76.92
		WW	15835.66	-4694.12	-22.86	0.00	-	-	0.00	-	-	6219.70	3586.24	136.18	651.64	613.26	1597.86	994.71	494.62	98.91
		WD	14838.72	-5691.06	-27.72	0.00	-	-	0.00	-	-	6711.78	4078.32	154.87	258.14	219.76	572.59	1893.07	1392.98	278.55
Turkey Point	1964	Base Case	546.95			0.00			0.00			763.07			425.59			91.31		
	(low water)	NWW	475.84	-71.11	-13.00	0.00	-	-	0.00	-	-	665.04	-98.03	-12.85	583.11	157.52	37.01	102.93	11.62	12.73
		NWD	247.59	-299.36	-54.73	0.00	-	-	0.00	-	-	310.49	-452.58	-59.31	696.72	271.13	63.71	572.12	480.81	526.57
		WW	200.07	-346.88	-63.42	0.00	-	-	0.00	-	-	312.96	-450.11	-58.99	604.68	179.09	42.08	709.21	617.90	676.71
		WD	50.63	-496.32	-90.74	0.00	-	-	0.00	-	-	403.30	-359.77	-47.15	494.31	68.72	16.15	878.68	787.37	862.30
	1978	Base Case	960.10			0.00			0.00			794.48			4.38			67.96		
	(high water)	NWW	595.66	-364.44	-37.96	0.00	-	-	0.00	-	-	1148.24	353.76	44.53	5.33	0.95	21.69	77.69	9.73	14.32
		NWD	443.16	-516.94	-53.84	0.00	-	-	0.00	-	-	1268.45	473.97	59.66	6.32	1.94	44.29	108.99	41.03	60.37
		WW	409.65	-550.45	-57.33	0.00	-	-	0.00	-	-	996.66	202.18	25.45	300.64	296.26	6763.93	119.97	52.01	76.53
		WD	256.30	-703.80	-73.30	0.00	-	-	0.00	-	-	998.50	204.02	25.68	76.82	72.44	1653.88	495.30	427.34	628.81
Rondeau Bay	1962	Base Case	170.12			0.14			0.00			131.68			1.59			2.73		
	(low water)	NWW	127.24	-42.88	-25.21	0.28	0.14	100.00	0.00	-	-	172.98	41.30	31.36	2.38	0.79	49.69	3.38	0.65	23.81
		NWD	64.77	-105.35	-61.93	0.79	0.65	464.29	0.00	-	-	223.56	91.88	69.78	10.05	8.46	532.08	7.09	4.36	159.71
		WW	44.88	-125.24	-73.62	4.63	4.49	3207.14	0.00	-	-	232.43	100.75	76.51	14.26	12.67	796.86	10.06	7.33	268.50
		WD	26.53	-143.59	-84.41	2.41	2.27	1621.43	0.00	-	-	173.26	41.58	31.58	89.05	87.46	5500.63	15.01	12.28	449.82
	1978	Base Case	286.50			0.00			0.00			17.48			0.18			2.10		
	(high water)	NWW	232.81	-53.69	-18.74	0.00	-	-	0.00	-	-	70.72	53.24	304.58	0.24	0.06	33.33	2.49	0.39	18.57
		NWD	106.29	-180.21	-62.90	0.00	-	-	0.00	-	-	194.86	177.38	1014.76	0.63	0.45	250.00	4.48	2.38	113.33
		WW	87.75	-198.75	-69.37	0.00	-	-	0.00	-	-	211.94	194.46	1112.47	0.92	0.74	411.11	5.65	3.55	169.05
		WD	67.87	-218.63	-76.31	0.00	-	-	0.00	-	-	228.54	211.06	1207.44	1.82	1.64	911.11	8.03	5.93	282.38

Climate Change Scenario: Not as Warm & Wet (NWW), Not as Warm & Dry (NWD), Warm & Wet (WW), Warm & Dry (WD)

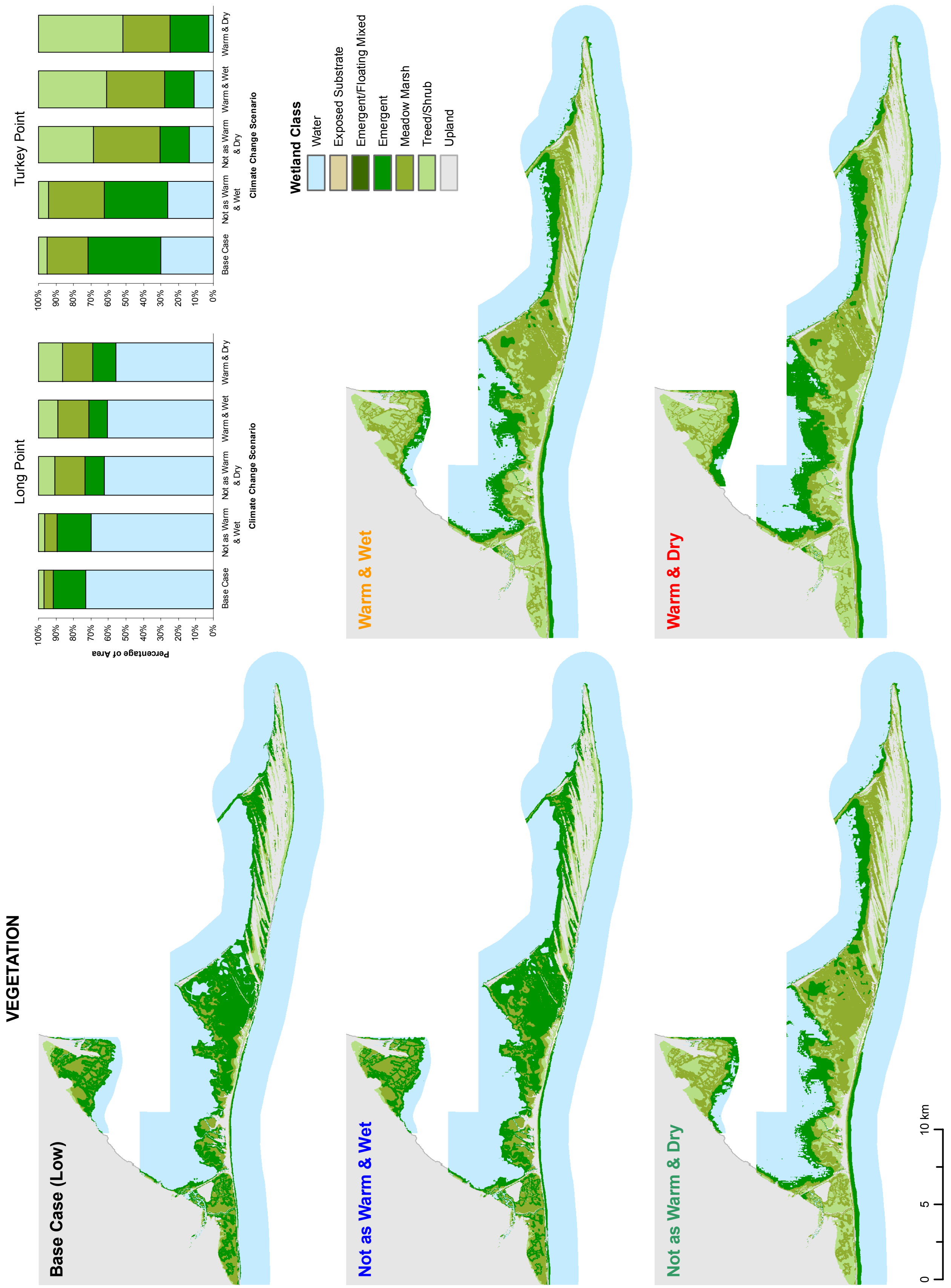

Figure 7.18 Predicted wetland class distribution under base case and climate change scenarios at Long Point and Turkey Point during low water (1964) conditions

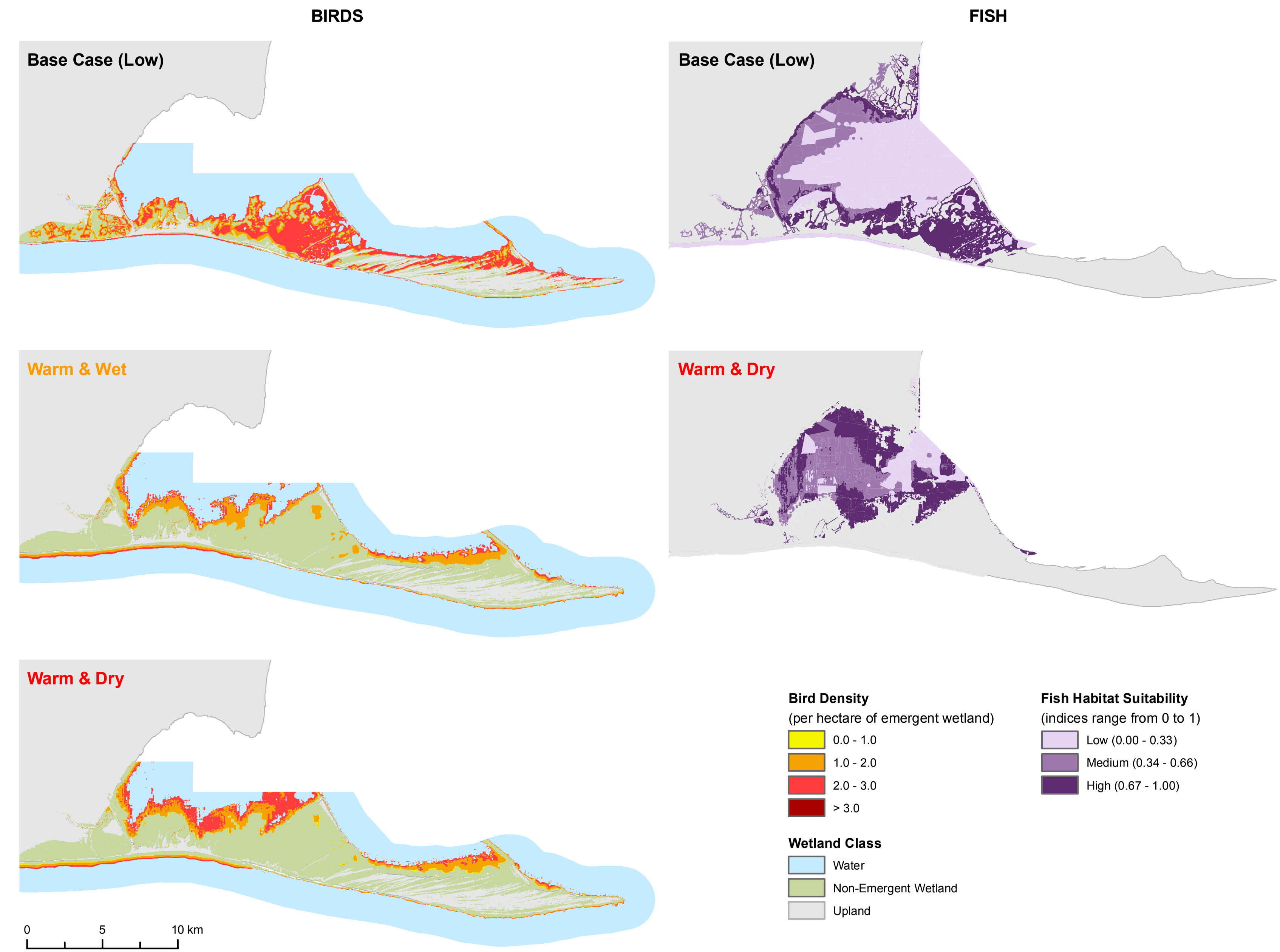

Figure 7.19 Predicted marsh nesting obligate bird densities and overall fish habitat suitability under base case and climate change scenarios at Long Point (birds) and the Inner Bay, including Long Point and Turkey Point (fish), during low water (1964) conditions (warm & wet, birds only; warm & dry, birds and fish). Note: fish suitability maps only show wetted area.

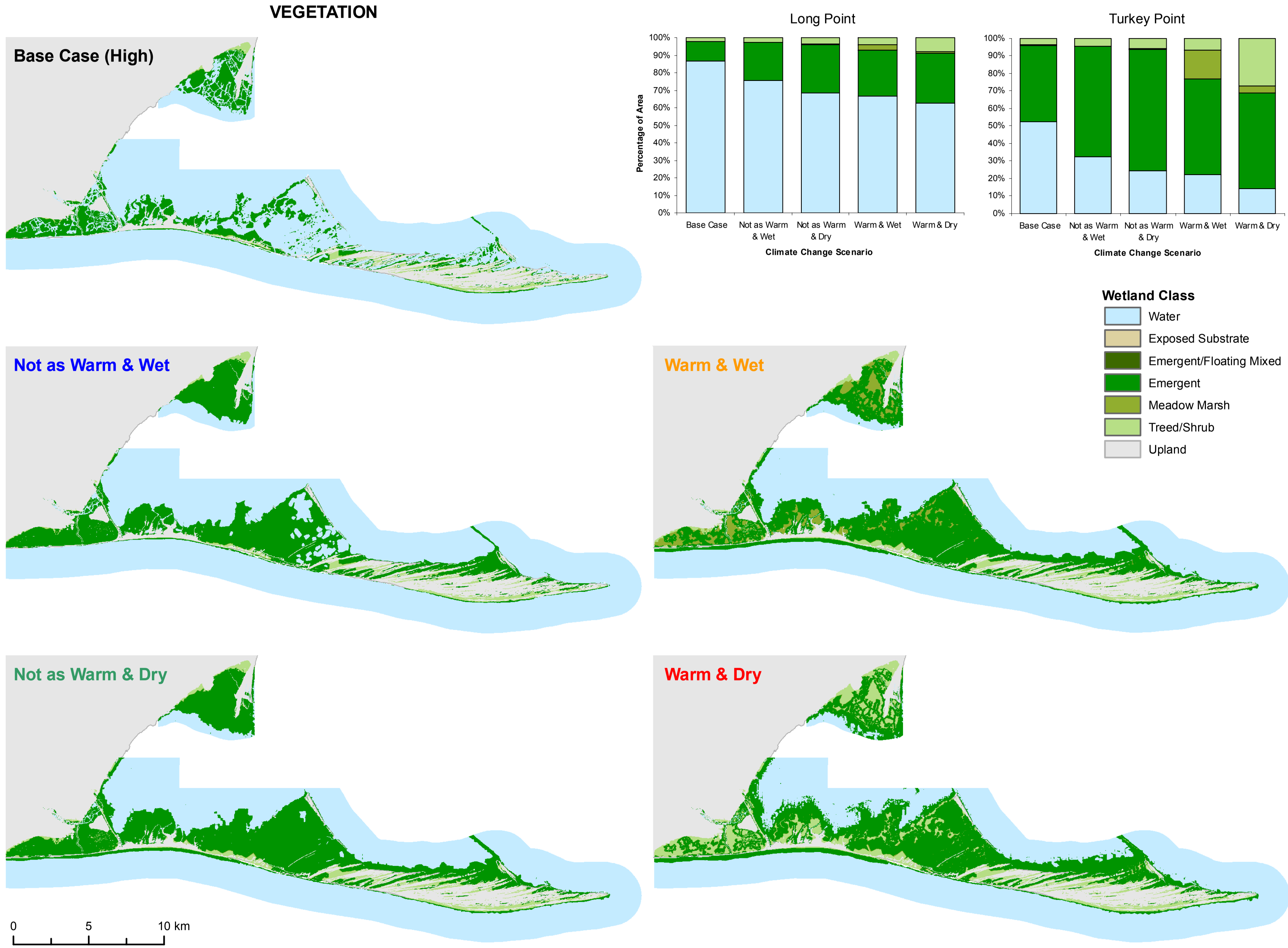

Figure 7.20 Predicted wetland class distribution under base case and climate change scenarios at Long Point and Turkey Point during high water (1978) conditions

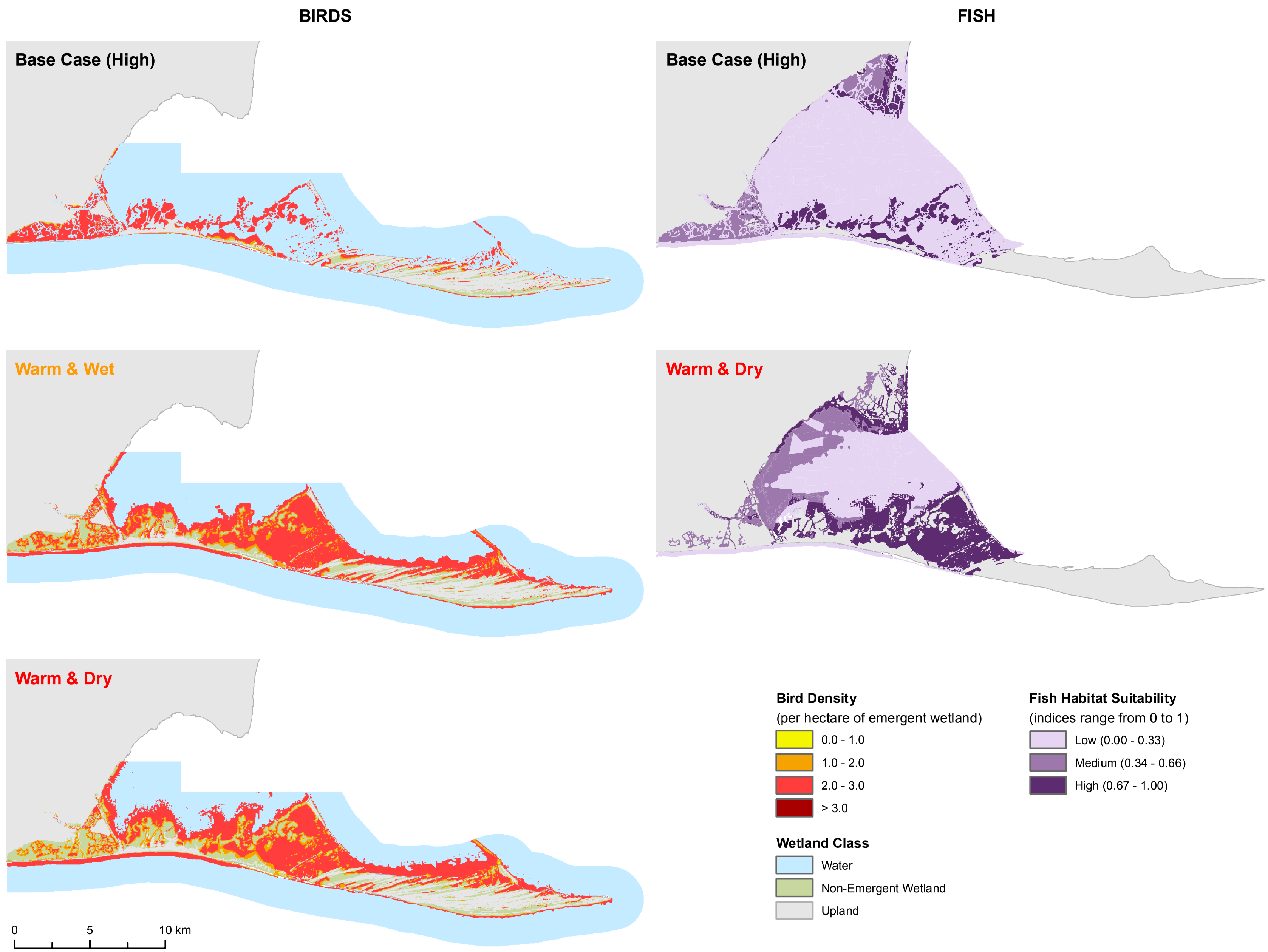

Figure 7.21 Predicted marsh nesting obligate bird densities and overall fish habitat suitability under base case and climate change scenarios at Long Point spit (birds) and the Inner Bay, including Long Point and Turkey Point (fish), during high water (1978) conditions (warm & wet, birds only; warm & dry, birds and fish). Note: fish suitability maps only show wetted area.

receded along the shore allowing emergent vegetation to migrate lakeward. Although emergent vegetation expanded lakeward, there was a net loss in area under all four climate change scenarios since the defined study area in the GIS-based vegetation model precluded a wider migration of emergent vegetation lakeward. A notable amount of meadow marsh expanded into emergent areas as water levels declined under the not as warm & wet scenario with increases of +271.1 ha (+64%) compared to base case. Meadow marsh also increased in the three more extreme scenarios, but as water levels declined with these scenarios the area of meadow marsh actually decreased compared to not as warm & dry conditions. The area of treed/shrub continually increased under all four scenarios expanding into areas previous vegetated with meadow marsh (Figure 7.18). By the most extreme warm & dry scenario, the amount of treed/shrub increased by +787.4 ha (+862%) from base case conditions and accounted for 48% of the total wetland area (Table 7.6). Emergent vegetation and meadow marsh accounted for another 22% and 27% of the total area, respectively, under this scenario while only 3% of open water remained.

High Water Year (1978)

The vegetation modelling results for the high water base case conditions showed that the Long Point wetland was dominated by open water (87%), with patches of emergent vegetation (11%) and little meadow marsh (0.2%) and treed/shrub (2%). Turkey Point was more equally composed of open water (53%) and emergent vegetation (43%) and little meadow marsh (0.2%) and treed/shrub (4%).

Open water continually declined under all four climate change scenarios in both wetlands. At Turkey Point alone, there was a -703.8 ha (-73%) reduction in open water under the warm & dry scenario and only 14% of open water remained; at Long Point, 63% of the area of open water remained. Open water losses were offset by an increase in emergent vegetation, as this community migrated into shallow, exposed areas as moisture conditions declined for the two not as warm scenarios. Both wetlands experienced a gain in emergent vegetation under the not as warm & wet scenario; emergent vegetation increased by +3,901.0 ha (+148%) at Long Point and +474.0 ha (+60%) at Turkey Point from base case conditions. As water levels declined under the warm & wet scenario, there were increases in meadow marsh as the community expanded into elevated areas previously vegetated by emergents. In this scenario the area of meadow marsh increased by +613.2 ha (+1,598%) at Long Point and +296.2 ha (+6,764%) at Turkey Point compared to base case. There were also increases in meadow marsh under the warm & dry scenarios compared to base case, but the area decreased as water levels declined compared to the warm & wet scenario. Treed/shrub vegetation continually increased in area under all four climate change scenarios, the greatest increase occurred under the most extreme warm & dry scenario. There were notable changes from meadow marsh to treed/shrub under the two warm scenarios, and by the warm & dry scenario, the area of treed/shrub increased by +1,393.0 ha (+279%) and +427.3 ha (+629%) at Long Point and Turkey Point, respectively (Table 7.6; Figure 7.20).

7.3.1.2 Bird Response

There were significant changes in the predicted wetland breeding bird community at Long Point under the warm & wet and warm & dry climate change scenarios (refer to Table 7.4; Figures 7.19, 7.21) for both low and high water years.

Low Water Year (1964)

For the low water base case conditions, Long Point supported relatively high indices of abundance for all of the wetland nesting guilds. The percentage of total index values for marsh nesting obligates, marsh nesting generalists, meadow marsh, and treed/shrub nesters were, 24%, 52%, 13%, and 11%, respectively (Table 7.4).

During the low water year at Long Point, the estimated index of abundance for both marsh nesting bird guilds decreased, and meadow marsh and treed/shrub nesting guilds increased under the climate change scenarios compared to base case. The approximately 30% to 40% reduction in marsh nesting obligate and generalist indices under the warm & dry and warm & wet climate change scenarios, respectively, were a result of less area in emergent vegetation under these scenarios (Table 7.4). Emergent vegetation, however, may be under-estimated because the “clipped” wetland study area may have limited the expansion of vegetation. Similarly, predicted increases in the meadow marsh and treed/shrub nesting guilds were the result of significant expansion in area of their habitats under the climate change scenarios (Figure 7.22). The distribution of marsh nesting birds was predicted to shift lakeward as emergent marsh vegetation was replaced by meadow and treed/shrub habitats in the higher wetland elevations (Figure 7.19). The marsh bird index density per hectare did not change for the climate change scenarios relative to base case as the estimated average breeding season water depth in emergent marsh habitats remained much the same (Table 7.4).

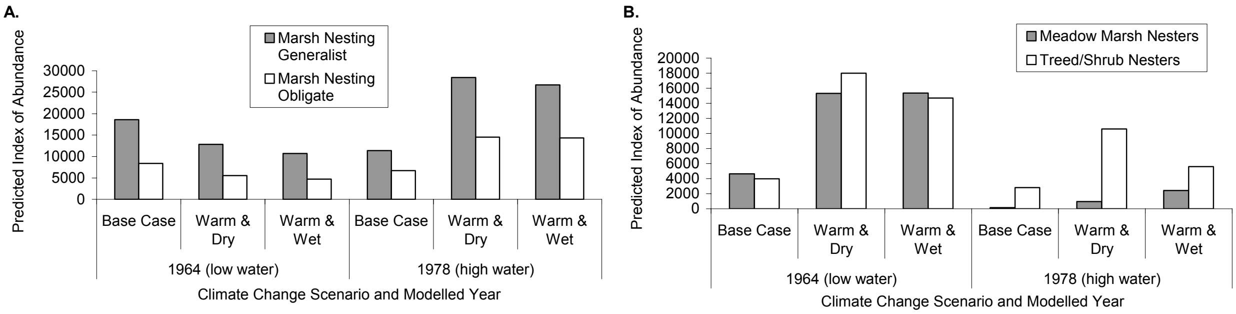

Figure 7.22 Predicted indices of abundance for marsh nesting (A), meadow marsh, and treed/shrub nesting (B) bird guilds under base case and climate change scenarios at Long Point during low water (1964) and high water (1978) conditions

High Water Year (1978)

All wetland vegetation community areas were reduced at Long Point in the high water base case in comparison to the low water base case (Figure 7.18, 7.20). As a result, the index of abundance for all nesting guilds was also lower relative to the low water base case (Figure 7.22).

In the high water year case, all wetland bird communities were predicted to increase under the warm & dry and warm & wet climate change scenarios relative to base case. Although deeper breeding season water depths in emergent marsh in the base case (0.64 m) relative to the warm & dry (0.42 m) and warm & wet (0.47 m) climate change scenarios resulted in higher densities of marsh nesting guilds per hectare, the lower water levels under the climate change scenarios resulted in a much greater area of emergent habitat being available for the marsh nesting obligate and generalist guilds (Figure 7.21). The warm & wet scenario had the highest meadow marsh nesting abundance index (2,411) compared to base case (142), and the warm & wet scenario (955). The lower water levels associated with the warm & dry scenario resulted in the highest index of treed/shrub habitat nesters (10,601) compared to the base case (2,800) and warm & wet scenario (5,570).

7.3.1.3 Fish

The fish found in surveys of the Inner Long Point Bay coastal margins and Big Creek Marsh (see Chapter 6 and wetland dyking section of Chapter 8 for fish lists) were used in an analysis of habitat supply in the Inner Bay of the Long Point and Turkey Point regions. The extent of the fish evaluation in Long Point was expanded beyond the emergent vegetation and bird evaluations to encompass as much of the wetted area in the embayment as possible but did not extend into the Outer Bay area. In this case, emergent vegetation predictions for the different scenarios were available for Turkey Point as well as Long Point and the entire embayment could be evaluated for fish habitat suitability. (Note: exposure was spatially modelled for predicting submergent vegetation only in each scenario tested.)

The overall fish community habitat response in the inner Long Point Bay area (including Turkey Point) under climate change conditions was quite different than the other wetlands assessed (Figures 7.19, 7.21). For example, the average habitat suitability of the area was predicted to increase under the climate change scenario (warm & dry) in both low and high water conditions. These differences were largely due to the low slope of the shallow areas that promoted downslope migration of emergent vegetation and the sheltering capabilities of the spit under different water levels that affected submergent vegetation growth under lower water levels.

Low Water Year (1964) & High Water Year (1978)

Long Point Bay, as part of Lake Erie, is not subject to water regulation. Therefore the total area flooded in low and high scenarios varied more than in Lake Ontario. Inundated areas under climate change conditions were much smaller during low levels, as warm & dry high and base case low scenarios had identical water levels. However, vegetation in habitats did differ slightly between the latter scenarios because of differing patterns in hydrologic cycles before the year of the scenario being tested; either 1964 or 1978.

When all fish species and their life stage requirements were evaluated under base case conditions using the composite scores (Figure 7.23), the proportion of high suitability habitat increased during low water levels, compensating for a decrease in overall wetted area. Weighted suitable area increased from 3,280 ha under high periods to 4,315 ha under low cycles (Table 7.5). Fringe habitats under high flooding events contribute proportionately small areas of medium and high suitability habitats in Long Point Bay for the predominantly coastal species evaluated.

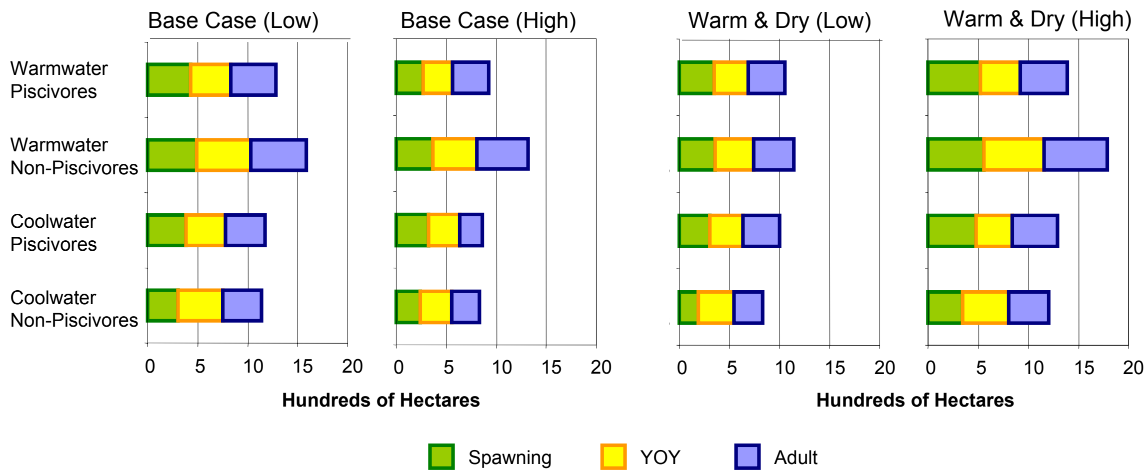

Figure 7.23 WSAs (in hundreds of hectares; areas x suitability rankings) for each of the fish guilds (feeding x thermal group) by life stage (spawning, YOY, or adult) habitats. Fish guild membership is listed in Table 6.8 (Chapter 6). WSAs are shown for base case and warm & dry climate scenarios during low water (1964) and high water (1978) conditions and vegetation changes predicted for Long Point Bay.

High suitability habitat was further reduced in area under the low climate scenario (warm & dry low); however, the proportion of good habitat increased overall between the two climate scenarios. When comparing between base case and climate conditions, the WSAs were predicted to increase by roughly 1,500 ha during high water periods but to decrease by roughly 1,000 ha under low levels (Table 7.5).

Long Point Bay appears to be an exception in the general pattern of climate change predictions for embayments, where the overall suitability compensates for a loss in flooded area at intermediate levels. All guilds showed an increase in WSA under low water levels in historic (base case) and high levels under climate change scenarios (Figure 7.24). The biotic response indicates a nonlinear relationship between weighted suitable area and water levels for this particular embayment.

Between climate change and base case scenarios, the loss of all life stage habitats ranged from 5-40% loss under low water periods but a gain of 20-100% in comparable habitats under high water levels (Table 7.5). Climate change low, life stage x guild areas were either equal to, or below, base case high WSAs, indicating that flooding events may be just as detrimental to productive capacity as reductions in water levels for most groups. Piscivore habitats increased the most under high water regime comparisons, almost doubling coolwater adult habitat and warmwater spawning habitat.

Usually YOY habitats are limiting to fish population dynamics (Minns 1997) but nursery habitat fluctuated the least of all the life stages in Long Point between scenarios (Figure 7.23). Nursery habitat for warmwater NPs was the most sensitive of the YOY stages evaluated, increasing by 40% between base case and warm & dry high scenarios but decreasing by 30% between lows. The spatial distribution of habitat suitabilities (Figure 7.24) mimicked the composite fish assemblage results (Figures 7.19, 7.21), with slight differences in the suitability of flooded fringe or transitional habitats. WSAs ranged from 4,335 to 5,355 ha in base case scenarios and from 3,890 to 6,025 ha in climate scenarios, indicating that the area of suitable nursery habitat will change more rapidly between the projected water levels, even increasing under climate highs, but decrease substantially once water levels pass a critical low threshold in the embayment.

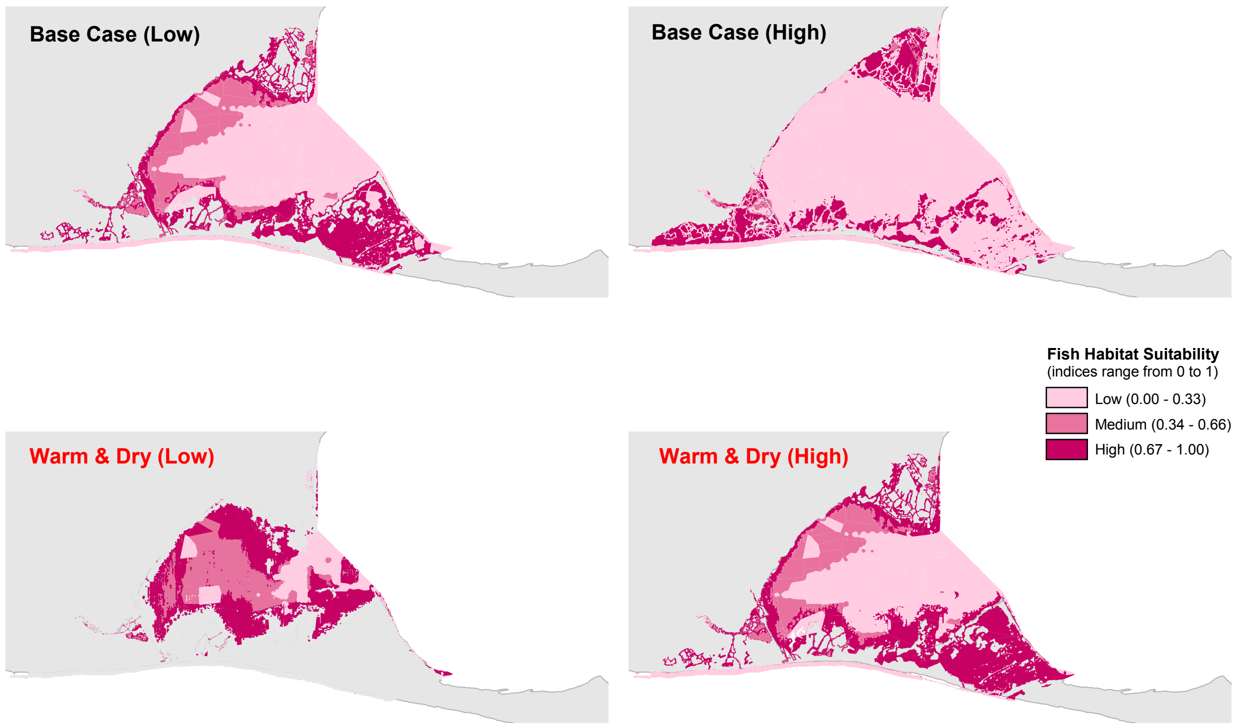

Figure 7.24 Habitat suitabilities (low, medium, and high) for the YOY life stage of warmwater non-piscivore fish species listed in Table 6.8 (Chapter 6). This life stage and guild were the most sensitive of the fish groups assessed in Long Point Bay. Habitat suitabilities were mapped for historic (base case) and warm & dry climate scenarios using high and low water level regimes and vegetation changes predicted for this area. Note: fish suitability maps show wetted area only.

7.3.2 Dunnville

7.3.2.1 Vegetation Response

Low Water Year (1965)

In the low water base case year, open water accounted for over half the total wetland area (50%). The total vegetated area within the wetland area consisted of emergent vegetation (accounting for 29% of the total wetland area), meadow marsh (16%), treed/shrub (3%), and emergent/floating mixed (2%).

Under the four climate change scenarios, the extent of open water continually decreased so by the warm & dry scenario, -59.8 ha (-25%) of open water was lost. As water levels declined under the four scenarios, there were also reductions in the amount of emergent/floating mixed and emergent vegetation of -5.7 ha (-62%) and -98.9 ha (-71%), respectively, under the warm & dry scenario compared to base case. These communities were displaced by meadow marsh and treed/shrub as moisture conditions in the wetland declined. There was a notable shift in emergent vegetation to meadow marsh during the not as warm & wet scenario, where meadow increased by +34.1 ha (46%). There were also small increases in meadow marsh under the not as warm & dry and warm & wet scenarios, though meadow marsh decreased during the warm & wet scenario as treed/shrub vegetation expanded lakeward. The area of treed/shrub continually increased under all four climate change scenarios, and by the warm & dry scenario the area had increased by +164.7 ha (+1,129%) compared to base case (Table 7.6; Figure 7.25).

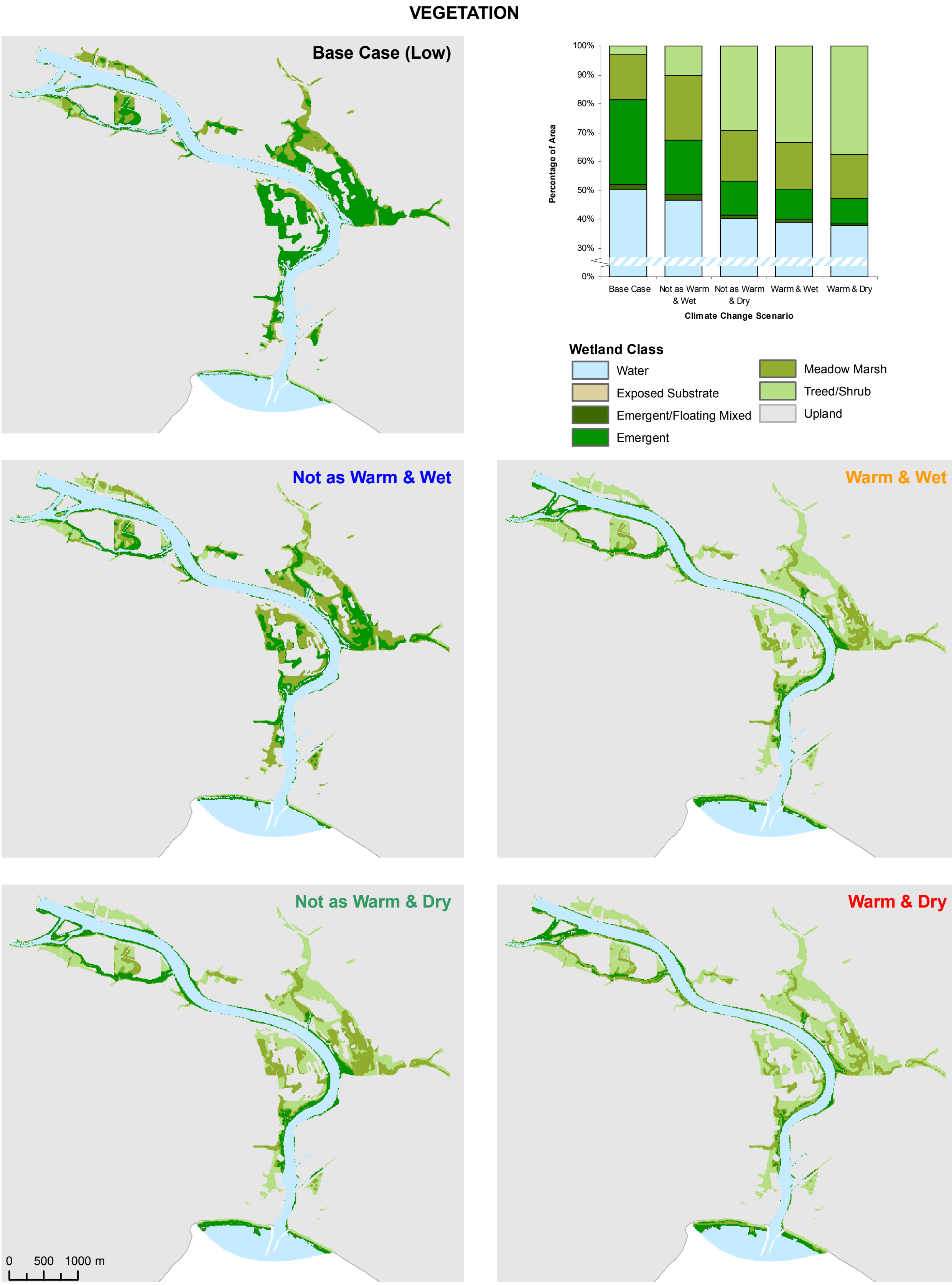

Figure 7.25 Predicted wetland class distribution under base case and climate change scenarios at Dunnville during low water (1965) conditions

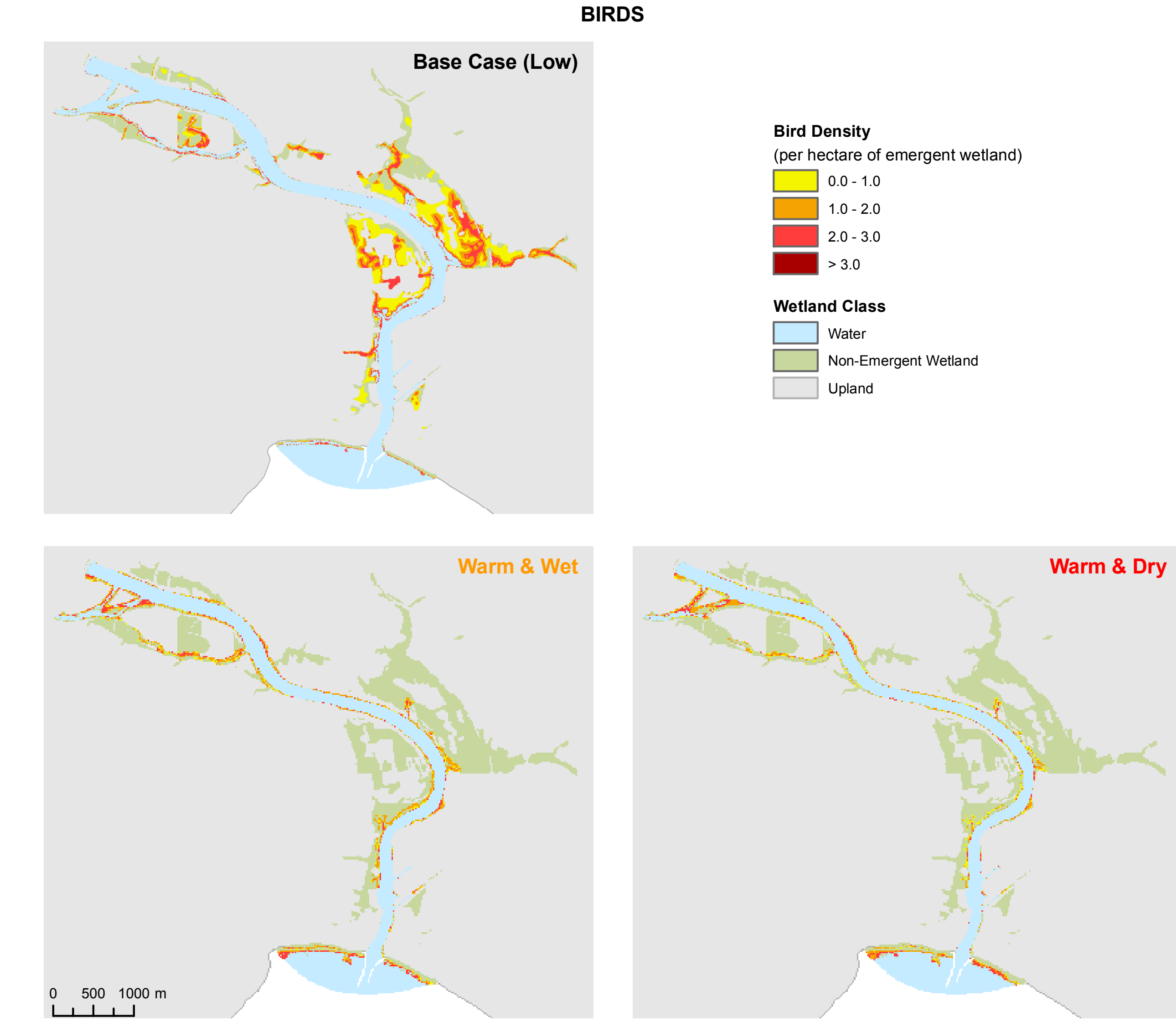

Figure 7.26 Predicted marsh nesting obligate bird distribution under base case and climate change scenarios at Dunnville during low water (1965) conditions

High Water Year (1978)

Reflecting the wet conditions within the wetland during the high water period, open water (60%) and emergents (38%) dominated, with about 2% of the area composed of treed/shrub and meadow marsh (Table 7.6; Figure 7.27).

In the most extreme water level decline under the warm & dry scenario, there was a -92.0 ha (-32%) loss of open water in the wetland. Similar to the low water condition, the greatest decrease in area occurred between the two not as warm scenarios with a drop in water levels from -15 to -55 cm (43.2 ha, 15%). In this scenario, tributary channels in the wetland dried up and filled with emergent vegetation. No floating emergent vegetation occurred during the high water conditions. Emergent vegetation increased under the not as warm scenarios, but decreased due to the larger water level drop in the two warm scenarios from base case. The community reached a threshold in which meadow marsh and treed/shrub were preferred. Under the high water levels, very little meadow marsh was predicted under base case conditions (2.4 ha). Meadow increased under all four scenarios, but the change was minor except during the warm & wet scenario, where the area increased +52.5 ha (+3,052%) from base case (Table 7.6). The area of treed/shrub increased under all four scenarios as well, most notably between the two warm scenarios (+64.5 ha, +80%) as treed/shrub

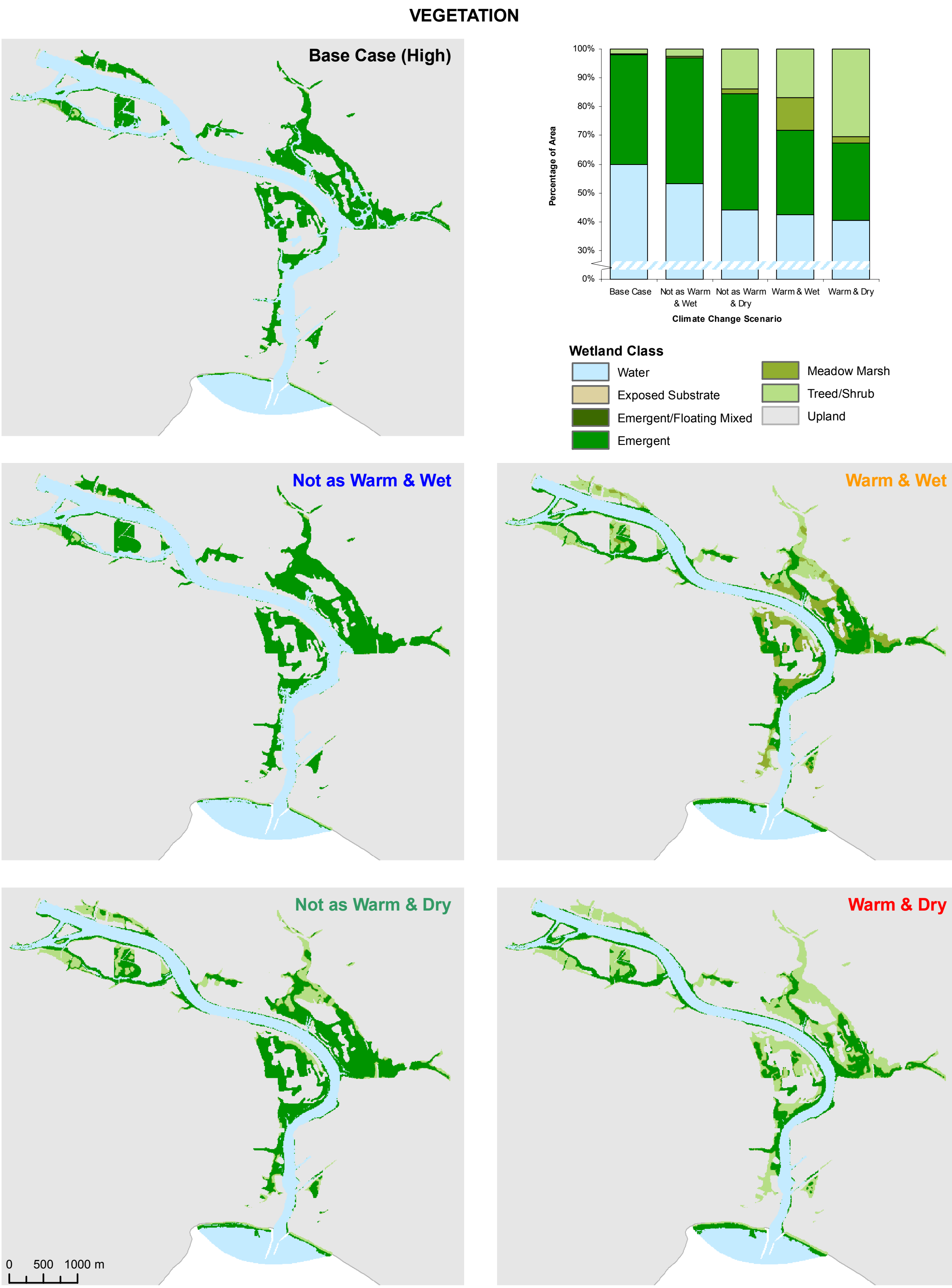

Figure 7.27 Predicted wetland class distribution under base case and climate change scenarios at Dunnville during high water (1978) conditions

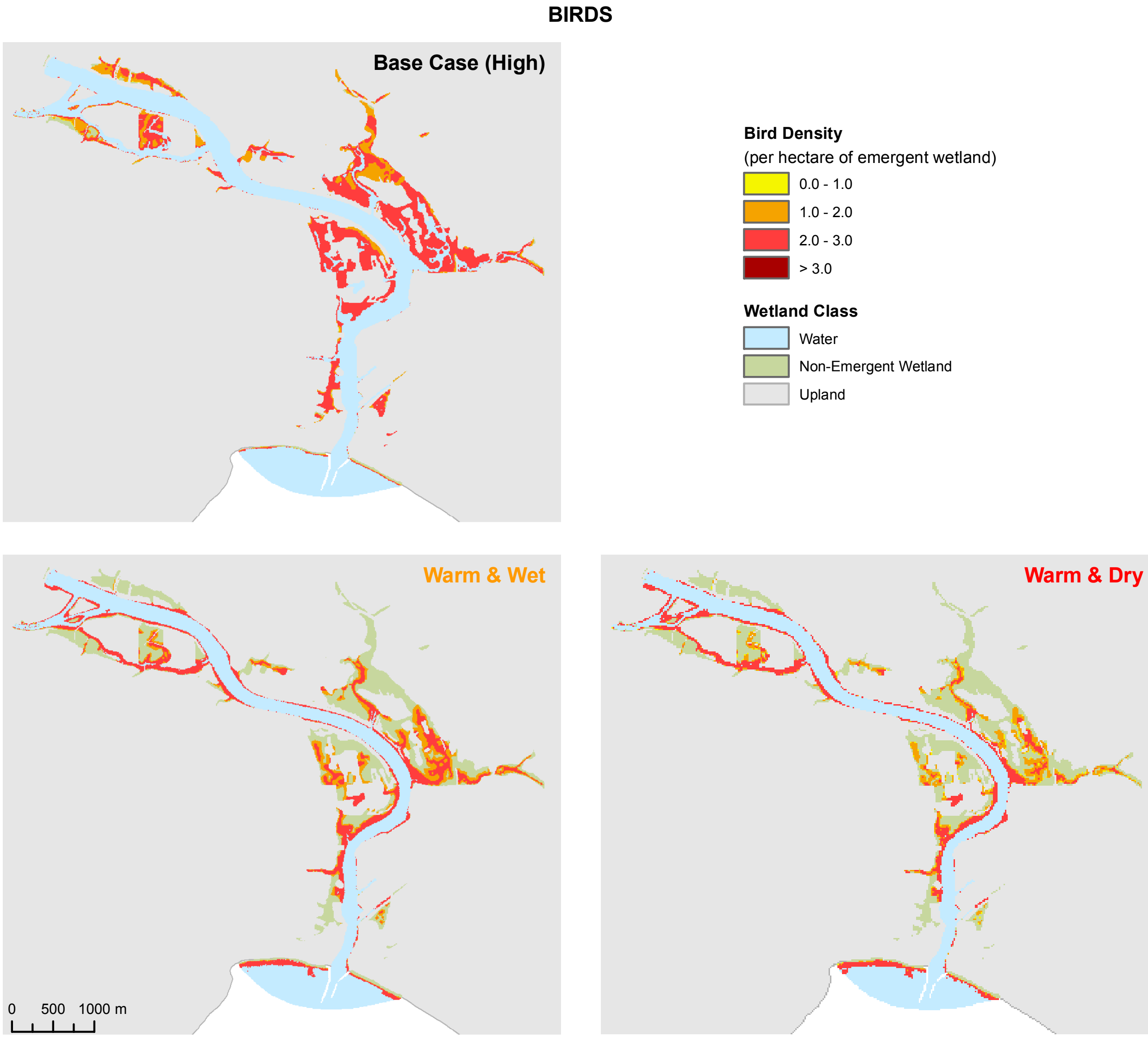

Figure 7.28 Predicted marsh nesting obligate bird distribution under base case and climate change scenarios at Dunnville during high water (1978) conditions

expanded into areas vegetated with meadow marsh and emergents (Figure 7.27). Under the warm & dry scenario, 59% of the wetland was vegetated; 30% of which was treed/shrub, 27% emergent, and 2% meadow marsh.

7.3.2.2 Bird Response

Low Water Year (1965)

The wetland bird community composition consisted of 17% marsh nesting obligates, 50% marsh nesting generalists, 25% meadow marsh nesters, and 8% treed/shrub nesters in the Dunnville wetland under the low water base case scenario (Table 7.4).

During the low water year, the estimated index of abundance for both marsh nesting bird guilds decreased, and meadow marsh and treed/shrub nesting guilds increased under climate change scenarios in comparison to base case. The -57% to -70% change in marsh nesting obligate and generalist indices under the climate change scenarios, were a result of reductions in the area of emergent vegetation (Table 7.4). The meadow marsh nesting guild index of abundance remained relatively constant (270-290) across all scenarios (Figure

7.29). The treed/shrub nesting guilds increases under the climate change scenarios were the result of expansions in area of this habitat (Figures 7.25, 7.26).

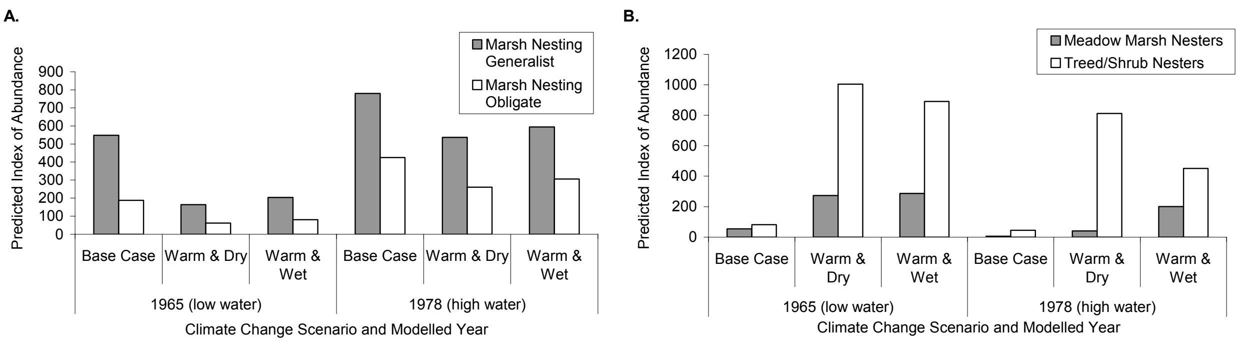

Figure 7.29 Predicted indices of abundance for marsh nesting (A), meadow marsh, and treed/shrub nesting (B) bird guilds under base case and climate change scenarios at Dunnville during low water (1965) and high water (1978) conditions

High Water Year (1978)

The wetland bird community composition consisted of 34% marsh nesting obligates, 62% marsh nesting generalists, 1% meadow marsh nesters, and 3% treed/shrub nesters in the Dunnville wetland under the high water base case scenario (Table 7.4).

Marsh nesting bird guilds decreased under the climate change scenarios in comparison to base case during the high water year. Changes ranged between -24% and -39%, and were the result of a reduction in emergent marsh area and lower water levels during the breeding period which reduced abundance index densities per hectare (Table 7.4). Large increases in the meadow marsh and treed/shrub nesting guilds were the result of expansions in area of these habitats under the climate change scenarios (Figures 7.27, 7.28). The warm & dry scenario supported the highest index of abundance for treed/shrub nesting birds at 811, while the warm & wet scenario supported the highest index of meadow marsh birds at 200 (Figure 7.29).

7.3.3 Rondeau

7.3.3.1 Vegetation Response

Low Water Year (1962)

During low water base case conditions, the vegetation model results indicated that the Rondeau wetland was dominated by open water (55%) and emergent vegetation (43%) with very little treed/shrub and meadow marsh.

The amount of open water consistently decreased under the four climate change scenarios as emergent vegetation expanded lakeward as water levels declined. By the warm & wet scenario, the amount of open water decreased by -143.6 ha (-84%) from base case. The vegetation model simulated a small amount of exposed substrate in the wetland during the low water level state, which increased in area under the four scenarios from base case. Exposed substrate actually increased the most under the warm & wet scenario (+4.5 ha, +3,207%) compared to base case. As water levels declined, substrate in deeper water areas/channels were exposed under the warm & wet scenario; for the warm & dry scenario, however, meadow marsh actually developed on these areas resulting in less exposed substrate in the wetland compared to the warm & wet conditions. Emergent vegetation increased under all four scenarios from base case, but the area actually decreased from the warm & wet to the warm & dry scenario, as large areas of emergent vegetation changed to meadow marsh; further expansion of emergent vegetation lakeward was limited by the study area boundary which did not allow for any growth beyond the clipped area. Compared to base case, meadow marsh increased +74.8 ha (+525%) from the warm & wet scenario and +87.5 ha (+5,501%) under the warm & dry scenario. The treed/shrub community continually expanded in area as water levels declined under all four scenarios (Table 7.6; Figure 7.30). As water levels declined -0.75 m under the warm & dry scenario, the wetland predominately consisted of emergents (57%) and meadow marsh (29%), with some open water (9%) and treed/shrub (5%), and little exposed substrate.

High Water Year (1978)

During the high water level state, there was very little vegetation within the wetland as open water accounted for 94% of the total wetland area. The remaining area mainly consisted of emergent vegetation with scarce treed/shrub and meadow marsh.

As water levels declined under all four climate change scenarios, the wetland was dominated by a lakeward expansion of emergent vegetation. The area of open water continually decreased from base case conditions; the greatest change occurred between the not as warm & wet and not as warm & dry scenarios, where water levels dropped -0.40 m (-126.5 ha, -54%). This reduction in open water was balanced by the expansion of emergent vegetation lakeward. Emergents increased a total of +211.1 ha (+1,207%) from base case and +124.1 ha (+176%) between the two not as warm scenarios. The model predicted very little occurrence of meadow marsh, which increased marginally as water levels declined, and no exposed substrate. There was very little treed/shrub in the wetland in the base case although it expanded along the upland boundary of the wetland in the more severe scenarios of water level decline (warm & wet, warm & dry). By the warm & dry scenario, 75% of the wetland consisted of emergent vegetation, while another 22% of the wetland remained open water; there were minimal amounts of meadow marsh and treed/shrub in the wetland (Table 7.6; Figure 7.32).

7.3.3.2 Bird Response

Low Water Year (1962)

In the low water year base case, the Rondeau wetland supported primarily marsh nesting bird guilds (obligate and generalists combined), 98% of the total index of abundance (Figure 7.34). This was primarily due to the abundance of emergent marsh habitat relative to meadow and treed/shrub habitats in this scenario (Figure 7.30).

For the low water year case at Rondeau, the estimated index of abundance for marsh nesting obligate bird guild decreased despite increases in emergent marsh area under both climate change scenarios compared to base case. The -41% and -24% changes from base case for the warm & dry, warm & wet scenarios, respectively, were the result of less suitable habitat being available due to lower/shallower breeding season water depths (Table 7.4). The drier emergent marsh habitat under the climate change scenarios supported lower bird abundance index densities relative to base case (Figure 7.31). The marsh nesting generalists model was less sensitive to water depth and predictions increased slightly under the climate change scenarios due to predicted increases in the area of emergent marsh habitat. The base case scenario had little meadow marsh and treed/shrub habitat, and associated bird species. The lower water levels associated with the climate change scenarios were predicted to support higher numbers of meadow marsh and treed/shrub bird nesting guilds, especially the warm & dry scenario with a high predicted index of abundance for meadow marsh birds (5,500) (Figure 7.34).

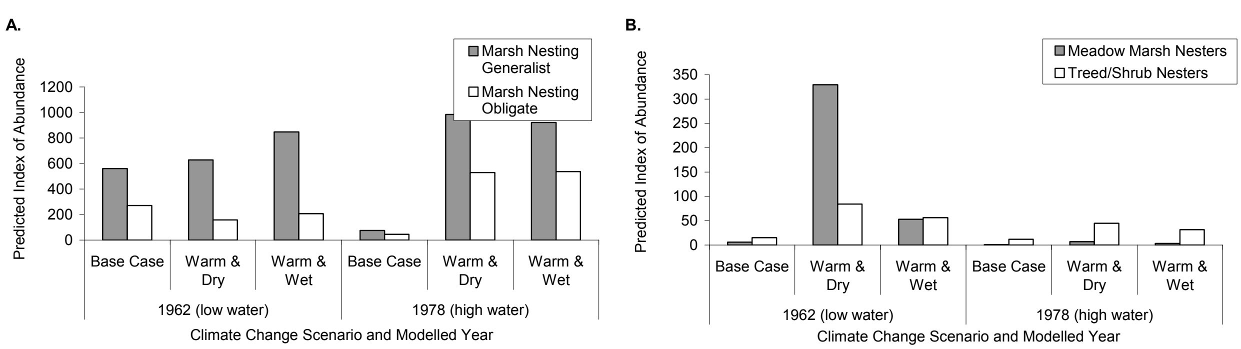

Figure 7.34 Predicted indices of abundance for marsh nesting (A), meadow marsh, and treed/shrub nesting (B) bird guilds under base case and climate change scenarios at Rondeau during low water (1962) and high water (1978) conditions

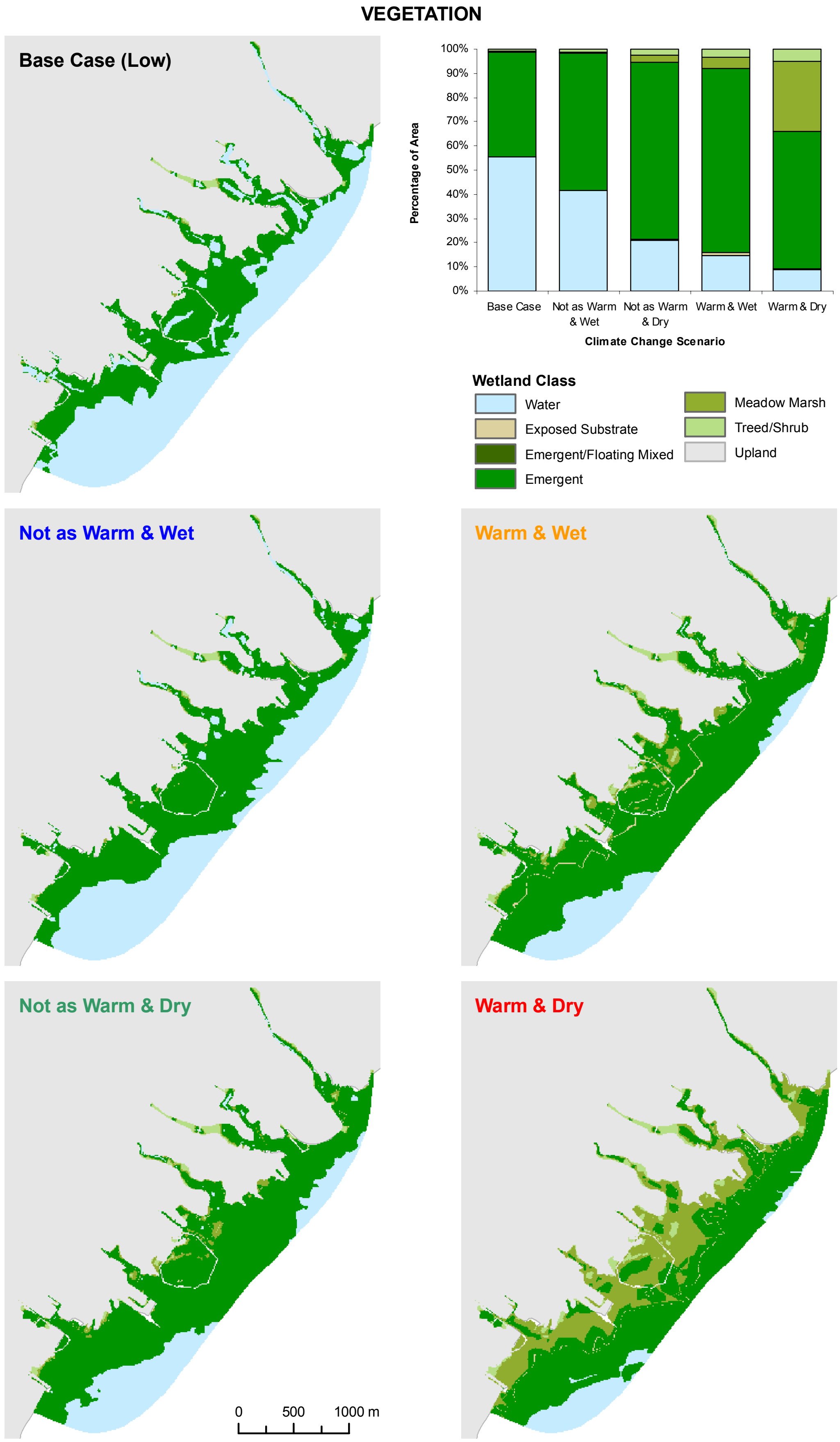

Figure 7.30 Predicted wetland class distribution under base case and climate change scenarios at Rondeau during low water (1962) conditions

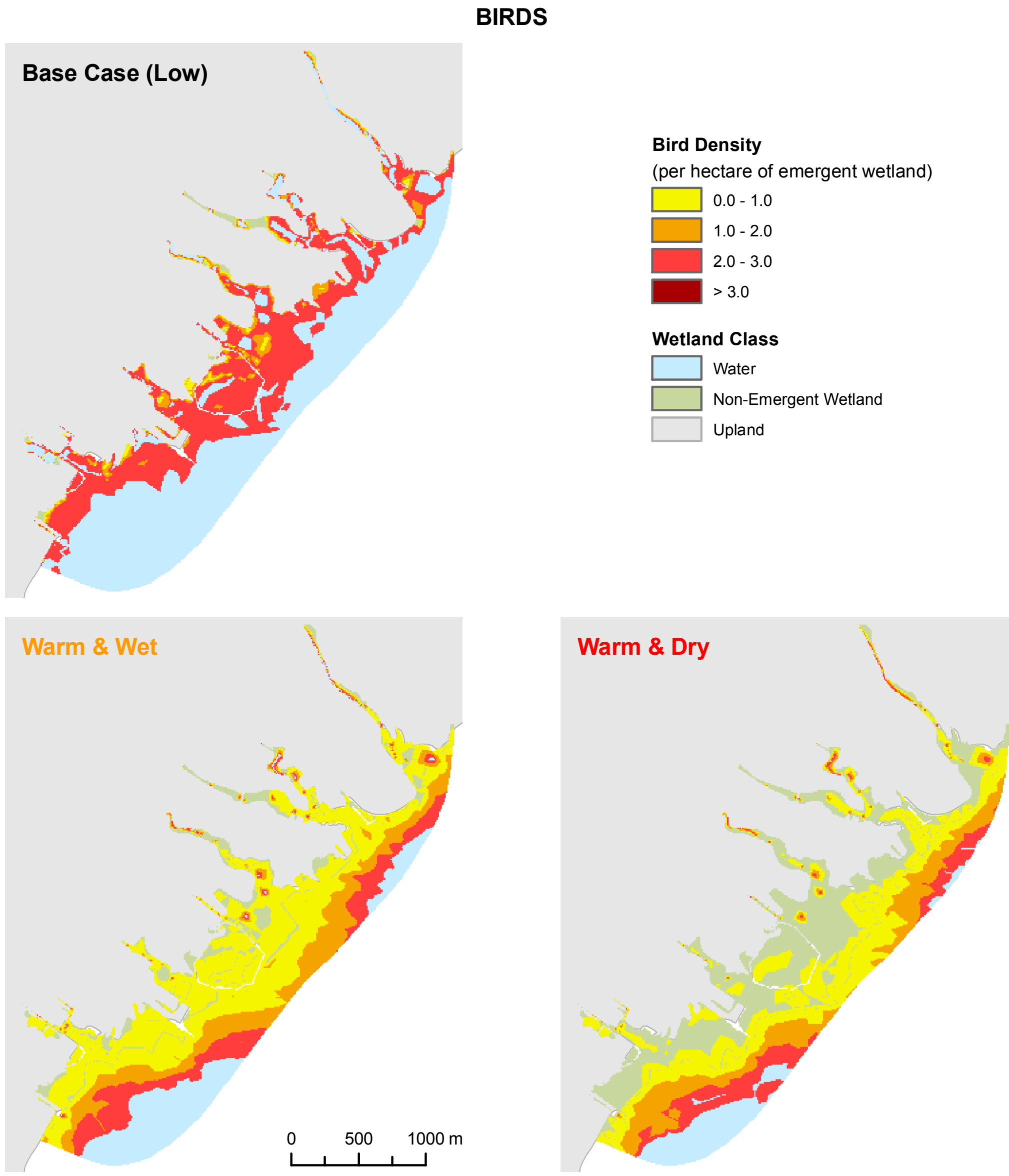

Figure 7.31 Predicted marsh nesting obligate bird distribution under base case and climate change scenarios at Rondeau during low water (1962) conditions

High Water Year (1978)

Little wetland habitat of any type was available for breeding birds under the high water year, base case scenario (Figure 7.33). As a result, all bird nesting guild indices of abundance increased under the climate change scenarios (Figure 7.34). Increases were largest for the marsh nesting guilds (>1,000%) which were related to large areas of deeply flooded emergent marsh habitat becoming available under the climate change scenarios relative to base case (Table 7.4). Although the percent change in abundance for the meadow marsh and treed/shrub guilds was also large, the actual abundance indices remain relatively low under the climate change scenarios during the high water year.

7.3.4 Lake Erie Summary

Overall, wetland communities had notable responses across all wetlands and for high and low water level conditions to the progressive water level decreases in the four climate change scenarios. Open water in all wetlands irrespective of geomorphic conditions decreased in the four scenarios while meadow marsh and treed/shrub expanded markedly. Emergent response was less clear; it declined in the low water condition for some wetlands and increased in the lacustrine wetland sites during the high water case.

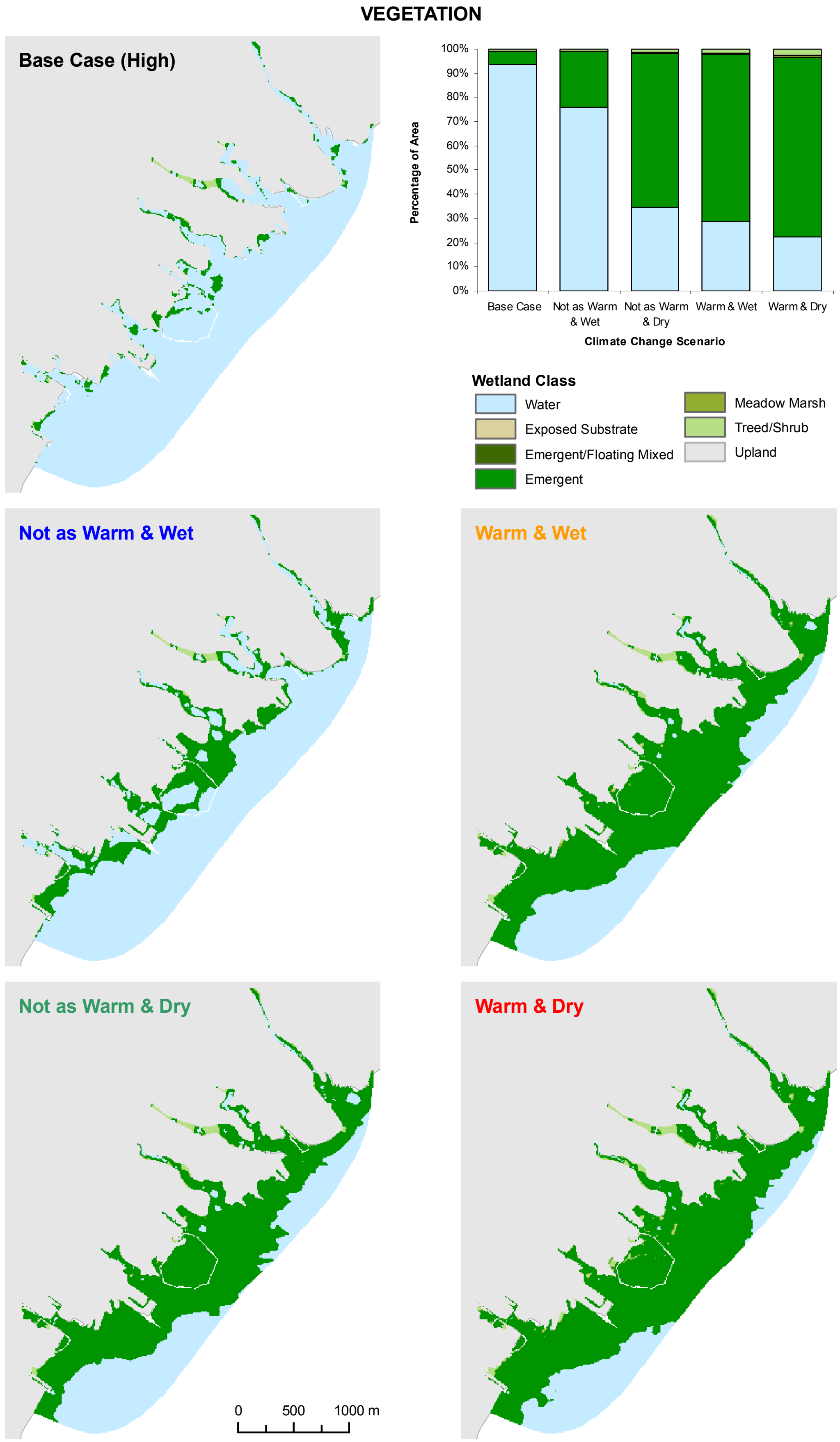

Figure 7.32 Predicted wetland class distribution under base case and climate change scenarios at Rondeau during high water (1978) conditions

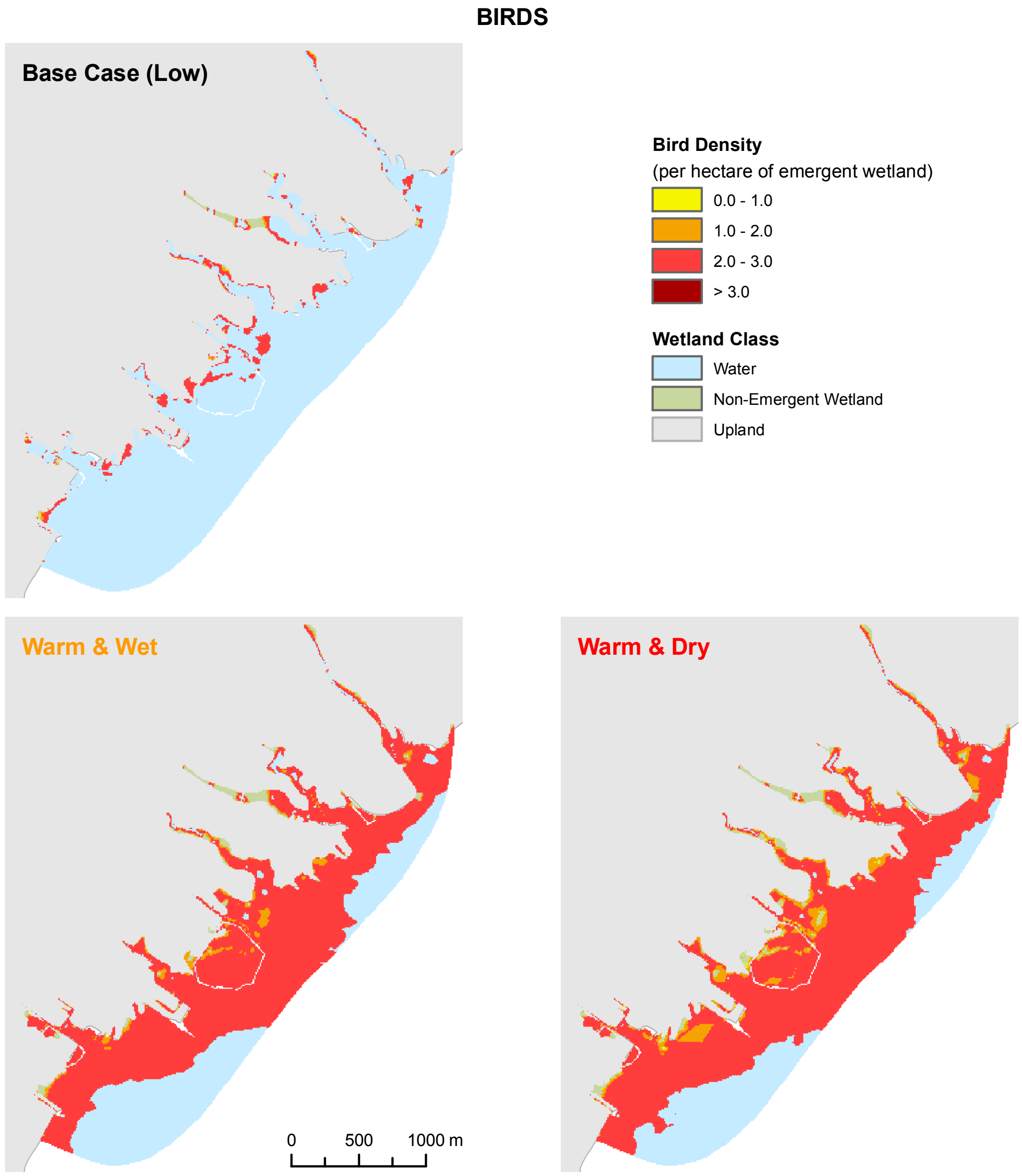

Figure 7.33 Predicted marsh nesting obligate bird distribution under base case and climate change scenarios at Rondeau during high water (1978) conditions

Wetland breeding bird communities showed some common responses to the warm & wet and warm & dry climate change scenarios. The abundance of marsh nesting obligate guilds declined in the low water level case across all wetlands for both scenarios. In the high water level case, marsh nesting obligates declined in the riverine conditions and increased in lacustrine wetlands. In Dunnville, loss of emergent habitat along the steep river banks was not compensated for in other parts of the wetland while in the lacustrine situations emergents expanded lakeward along shallow, sloping areas.

Fish responses in riverine environments were not tested but the response in habitat supply in Rondeau Harbour would be similar to fish results predicted for Long Point Bay based on water levels and vegetation changes predicted. Under high water conditions lacustrine embayments would lose high quality vegetated habitat but gain wetted area. Moderate water level scenarios would have an overall increase in habitat supply with intermediate increases in habitat quality but under extreme conditions, like the warm & dry scenario, the wetted (inundated) area loss would drive the habitat supply down even though quality remained high in the embayments for coastal fish using those areas. Warmwater non-piscivores, or coastal forage fish, were predicted to be the most sensitive guild in Lake Erie coastal wetlands.

7.4 Lake St. Clair Climate Change Assessment

7.4.1 Mitchell's Bay Fish Response

Low Water Year (1964) & High Water Year (1978)

Lake St. Clair, like Lake Erie, is not subject to water regulation. Therefore the total area flooded in Mitchell's Bay during low and high scenarios varied proportionately greater than in Lake Ontario. Because of steeper elevations near a somewhat developed shoreline in Mitchell's Bay, compared to previous embayments, the changes in inundated areas between all scenarios were more gradual. The flooded area within the model extent, ranked from highest to lowest scenario, were: base case high, base case low, warm & dry high, and warm & dry low. Much of the habitat was homogeneous between scenarios. There was some extension of emergent wetland on the northside of the bay with submergents throughout the remainder of the shallow area. The factor that dictated habitat suitability differences between patches was substrate type. However, most of the bay was uniform in substrate type.

Therefore, because much of the shallow and vegetated habitat was moderate suitability for the fish assemblage found here (Figure 7.35), the differences in composite habitat supply were completely based on the wetted area of the scenario. WSAs ranged from 30 and 50 ha under historic water levels and remained relatively constant at 30 ha under predicted climate change water level ranges (Table 7.5). Higher suitability habitats at the fringe of the wetland were lost under the 50-year projected water levels.

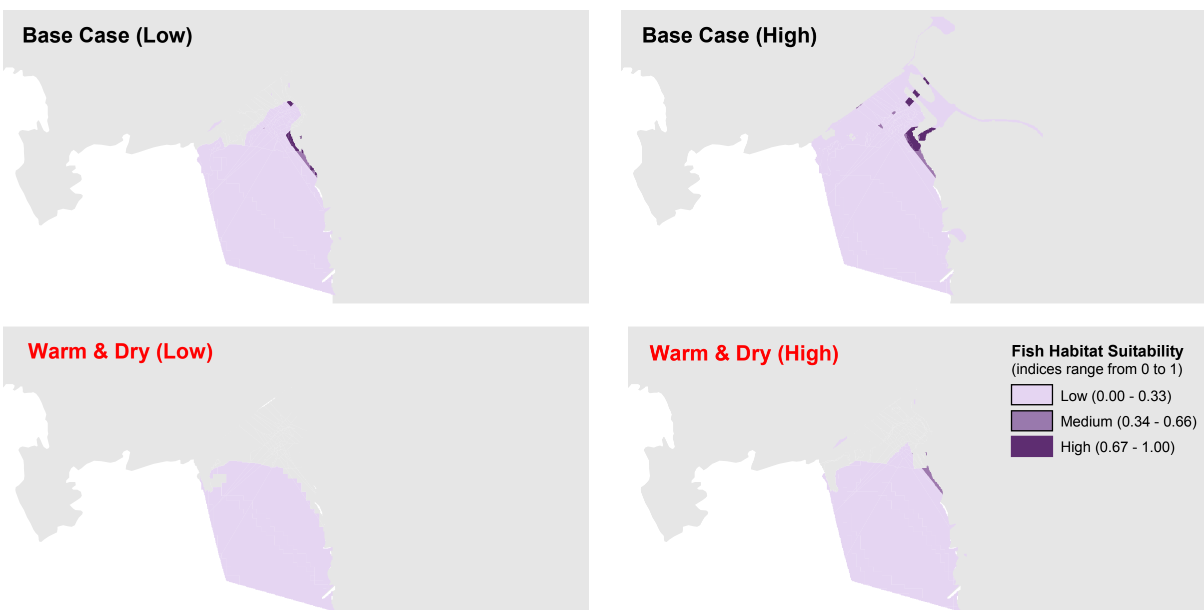

Figure 7.35 Composite habitat suitabilities (low, medium, and high) for all life stages of the fish species listed in Table 6.9 (Chapter 6). Habitat suitabilities were mapped for base case and warm & dry climate scenarios using low water (1964) and high water (1978) conditions and vegetation changes predicted for Mitchell's Bay.

YOY habitat was usually the lowest WSA of the three life stages for most of the guilds evaluated (Figure 7.36). Exceptions included adult habitat for warmwater non-piscivores and spawning habitat for coolwater non-piscivores. Piscivores comparatively had very little nursery habitat available in the scenarios, with the greatest decreases occurring at a water level within the range of historic variability, indicating that it would take a small drop in water levels to lose prime nursery habitat in this embayment. Overall, life stage WSAs between base case and warm & dry scenarios decreased between 25 and 70% between highs. Low water level comparisons between climate scenarios varied widely by life stage and guild; some increased by 30% while others decreased up to 55% (Table 7.5).

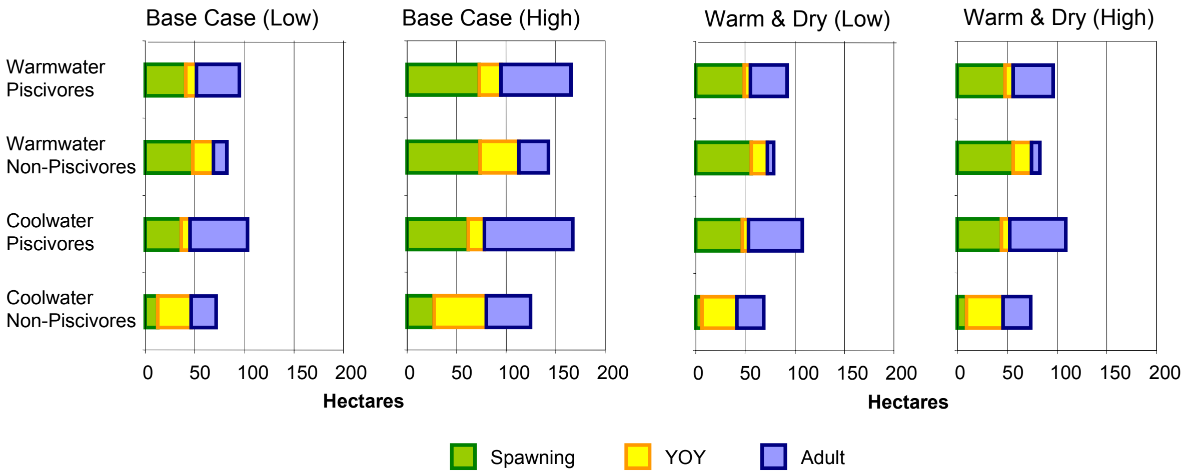

Figure 7.36 WSAs (in hectares; areas x suitability rankings) for each of the fish guilds (feeding x thermal group) by life stage (spawning, YOY, or adult habitats). Fish guild membership is listed in Table 6.9 (Chapter 6). WSAs are shown for base case and warm & dry climate scenarios using low water (1964) and high water (1978) conditions and vegetation changes predicted for Mitchell's Bay.

Warmwater piscivores (WM-P) were the most sensitive of the YOY stages (Figure 7.37). WSA decreased by 60% between high regime comparisons and 45% of predicted WSA was lost under low water cycles from base case to climate change scenarios. The spatial pattern of habitat suitabilities between scenarios for YOY WM-P was similar to the composite findings based on all life stage requirements of the entire fish assemblage. The pattern indicates that high suitability habitats are rare and found along the shoreline, which are subsequently lost during slightly lower water levels.

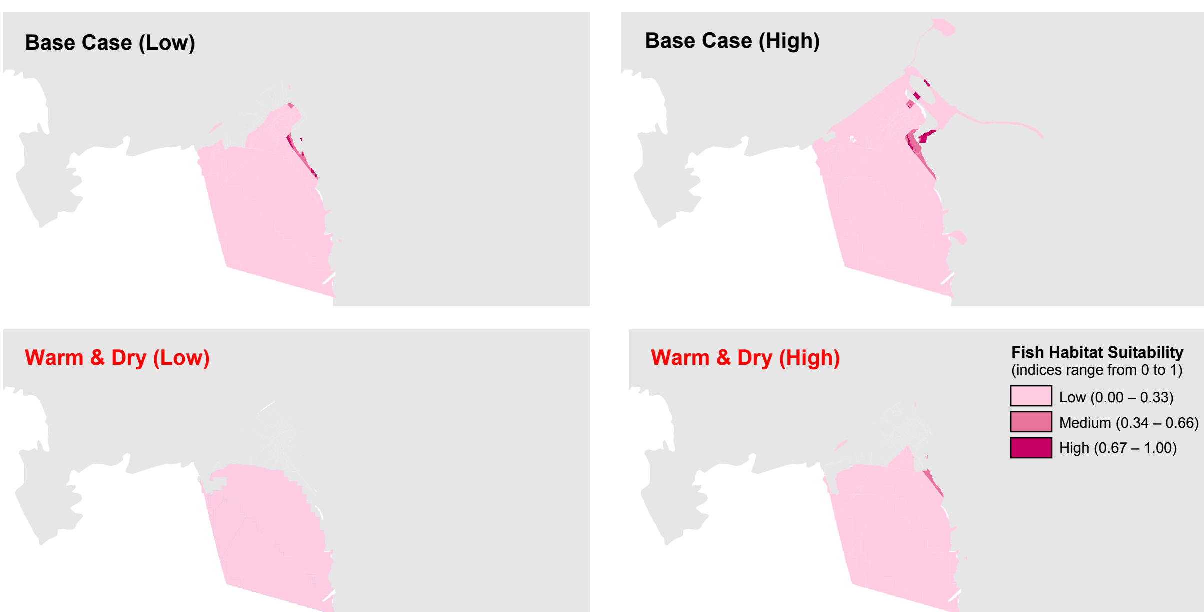

Figure 7.37 Habitat suitabilities (low, medium, and high) for the YOY life stage of warmwater piscivore fish species listed in Table 6.9 (Chapter 6). This life stage and guild were the most sensitive of the fish groups assessed in Mitchell's Bay. Habitat suitabilities were mapped for base case and warm & dry climate scenarios using low water (1964) and high water (1978) conditions and vegetation changes predicted for this area.

7.5 INTEGRATED ASSESSMENT SUMMARY

The assessment demonstrated that Great Lakes coastal wetlands have a natural capability to adapt to changing water levels but there were notable changes in vegetation communities that influenced wetland "quality" in terms of bird and fish habitat. The following questions were used in assessing the effects of water level changes due to climate change:

- Are there different responses in wetland vegetation, birds, and fishes to water level change between regulated Lake Ontario and unregulated Lake Erie?
- Does wetland hydrogeomorphology (i.e. lacustrine or riverine conditions) affect vegetation community response to water level changes?
- Are ecosystem responses different when initial water level conditions are low or high?
- Are there thresholds of water level change that are identifiable through notable responses in vegetation, bird, and fish communities?
- Do certain communities or groups exhibit greater responses to altered water level conditions relative to others?
- Is the overall "quality" of wetland communities affected by water level decline?

Although rule-based, empirical and matrix modelling, rather than process-based modelling, was used to project community responses for wetland vegetation, birds, and fishes, this initial first-step made a significant contribution to understanding the effects of water level changes on wetland ecosystems. For example, modelling of base case conditions in wetland communities indicated that the responses of vegetation communities were consistent with historical, spatiotemporal analyses, and theoretical responses from the literature. As such, it was reasonable to use rule-based modelling as a tool for understanding ecosystem effects of future water level changes. Incorporation of breeding season water depth based regression models for emergent marsh bird nesting guilds highlighted the importance of standing water and depth when considering the breeding bird habitat supply and quality within emergent marsh. A matrix modelling approach was applied to gauge fish responses and used fish habitat associations and preferences at different life stages to assess fish habitat supply and quality based on vegetation, depth, and substrate characteristics. Associations and preferences are based on literature values but have been validated by habitat surveys conducted during the course of the dyked wetland evaluations (Chapter 8).

Most responses of the vegetation communities to the climate change water level scenarios were similar in regulated Lake Ontario and unregulated Lake Erie. Furthermore, no definitive thresholds emerged that mark distinctive wetland vegetation community responses as water levels progressively declined under the climate change scenarios for Lakes Ontario (three scenarios) and Erie (all four scenarios). However, for Lake Ontario, the modest 6 cm increase in water levels for the not as warm & wet scenario clearly exhibited different modelled wetland vegetation responses than the three scenarios where water levels decreased. In the case of the small water level increase under both low and high initial conditions, there were only minor changes in wetland communities; the area of open water shows a small increase, meadow marsh and treed/shrub decrease, and emergent has both very small increases and decreases. The opposite response occurs with water level declines. As wetlands dried with decreasing water levels, conditions were preferable for treed/shrub and meadow marsh communities and their areas expanded, particularly along the upper, drier margins of the wetland. Open water, interspersed in pockets within a wetland and along the shoreline, decreased in all scenarios with water level decline.

Hydrogeomorphic form and initial water level condition (low and high) seemed to be more important in influencing vegetation community outcomes. Depending on the magnitude of the water level decrease and the wetland hydrogeomorphic form, a wetland such as Lynde Creek, can completely dry out. Area of meadow marsh in all wetland types of Lake Erie increased in both low and high initial water level conditions. In Lake Ontario, hydrogeomorphic form influenced the response in low conditions where area of meadow marsh decreased in riverine wetlands while it increased in all other situations. Hydrogeomorphic conditions also influenced emergent community responses on both lakes under high initial water level conditions. Emergent community area decreased in riverine wetlands and increased in lacustrine environments. Lacustrine, protected embayments such as Long Point Bay and Presqu'ile Bay have gentle, gradual sloping bathymetry that allowed for an even colonization of emergent vegetation. In riverine wetlands (e.g. drowned river mouths with steep river banks and rapid bathymetric changes), such as Hay Bay, the uneven bathymetry resulted in large, abrupt shifts in vegetation, particularly in the emergent community, as the steep river channels constrained colonization and movement downslope. Under low initial water conditions with progressive water level decline, the area of emergents decreased in most wetlands. However, the wetland study areas, defined in the GIS by a fixed boundary area, limited modelling the downslope expansion in emergent vegetation to within the defined study area, particularly for low initial conditions and resulted in the under-estimation of the potential expansion of emergent vegetation. Antecedent water level conditions had a significant influence on vegetation communities which in turn influenced fish and bird habitat even though final water level conditions may be the same. The low base case year with the climate change water level reductions exhibited greater increases in meadow marsh and emergent vegetation than the high water year comparisons.

Responses of bird communities, particularly marsh nesting generalists, meadow marsh nesters and treed/shrub nesters, to water level changes were primarily influenced by whether the wetland vegetation community providing breeding habitat expanded or contracted with water level decline. As water levels progressively dropped with more severe climate change scenarios, area of treed/shrub and meadow marsh vegetation communities expanded with an associated increase in abundance of treed/shrub and meadow marsh nesting guilds. The index of abundance for the marsh nesting generalist guild also responded positively to increases in area of emergent marsh despite the shallower average water depth during breeding season. However, marsh nesting obligate guilds are sensitive to breeding season water depth in emergent marsh habitat as well as areal extent of emergents. Their abundance and density per hectare decreased for both high and low conditions between historic and climate scenarios in all wetlands in Lakes Erie and Ontario.

Site-specific differences in fish habitat were apparent when selected wetlands were modelled using a spatially explicit approach. Presqu'ile Bay exhibited a large shift in fish habitats under the warm & dry climate change scenario. Long Point Bay was moderated by local geography, as suitabilities were predicted to increase but area decreased until it outweighed compensatory mechanisms under extreme conditions (i.e. low water levels under warm & dry conditions). Mitchell's Bay seemed to already be experiencing loss of shoreline fish habitats under current (base case) fluctuations in water levels and the loss of area due to further drops in water levels was not compensated by changes in habitat suitability because Lake St. Clair was relatively uniform in physical characteristics. However, subtle local depth changes in Mitchell's Bay may actually benefit some species or guilds based on the results.

Each of the wetlands had different rankings for the sensitivity of fish guilds, responses which were mainly gauged by percent YOY habitat changes. Coolwater non-piscivores in Presqu'ile lost the greatest area but all guilds generally followed the same patterns in habitat supply. Similarly in Long Point, most responses had the same pattern regardless of guild or life stage, with some subtle differences. Warmwater non-piscivores were the most variable, and climate change appeared to favour adult coolwater piscivores over non-piscivores in the same thermal guild; however, spawning habitat of warmwater piscivores was favoured over coolwater piscivores. The implications of this are unclear to production in these groups. Mitchell's Bay warmwater piscivores decreased steadily with lowering water levels in Lake St. Clair (i.e. through high historic to low climate change scenarios). This guild had a different response than others in the area where most weighted suitable areas flat-lined beyond a certain water level rather than continuing to decrease. This pattern may indicate a threshold for water levels within the range tested.

It should be noted that temperature changes in nearshore areas were not considered in the assessment of fish habitat supply, only coastal habitat changes. Therefore, the responses of guilds based on their thermal requirements, like egg development and survival needs, as well as YOY tolerances, could produce different responses under climate change but likely not change trajectories projected for guild responses. These thermal requirements were taken into consideration in the vulnerability assessment of Great Lakes fish species earlier and the proportion of high risk species in the habitat assessment was also high. Twelve species of the 28 fishes evaluated in Presqu'ile Bay, 17 of 36 species in Long Point Bay, and 15 of 37 species in Mitchell's Bay, were of high risk. The guilds with the most sensitive habitats, those which experienced the greatest decrease under climate change, were also comprised of 50% high-risk coastal species. Assignment of high risk included spawning and adult thermal requirements for those guilds as well as habitat requirements.

As the scenarios of water level decline became progressively more severe, particularly under low water level conditions, wetlands transition to drier vegetation communities (treed/shrub and meadow marsh) with less open water and interspersion. This wetland habitat is not as productive because it is not as complex and favourable to many wetland-dependent birds (for breeding and migration), fishes (for nursery areas and feeding), and other wetland-dependent wildlife except for some songbirds, migratory birds, and fish species such as northern pike who use treed/shrub and meadow marsh habitat. Yet, this habitat is historically most impacted by, and lost to agricultural, residential, and recreational development. With lower water levels and migration of these vegetation communities downslope, there is potential for expansion and restoration of habitat providing benefit to associated birds and other wildlife. However, some land use policy is required that prevents development from encroaching downslope and utilizing the newly exposed areas. It should be noted that this habitat would still be intermittently flooded during high water periods in the springtime and used by species such as northern pike for spawning and nursery habitat. This was not captured in the scenarios because average annual breeding and growing season values were used for birds and fish responses respectively.

Overall, the low water level, warm & dry scenario was the worst scenario for vegetation, birds, and fishes, where significant shifts in the distribution and abundance of wetland communities were projected relative to historic conditions in particular embayments. It is apparent that in many of the areas modelled that the wetland vegetation and habitat for birds and fish will be altered hydrologically and that all vegetation types will not migrate downslope with decreasing water levels. The interactions between local bathymetry and other physical features will also determine the ultimate suitability of these changing embayments and the overall suitable habitat availability for the species using them.

7.6 REFERENCES

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8.0 Preparing for Climate Change: Assessing Adaptation Strategies for Coastal Wetlands

Susan Doka, Joel Ingram, Linda Mortsch, and Andrea Hebb

Three adaptation strategies to climate change were investigated and evaluated for Great Lakes coastal wetlands. The evaluated adaptations are a subset of measures that are already used in the lower Great Lakes and impact water levels as well as coastal areas. The subset includes: lake-wide water level regulation on Lake Ontario; dyking of wetlands in Lakes Ontario, Erie, and St. Clair; and land use planning and policy (Figure 8.1). This part of the study was a compilation of model-based assessment, field study, literature review, and stakeholder consultation of the different strategies. The three strategies are introduced here but are detailed in subsequent sections.

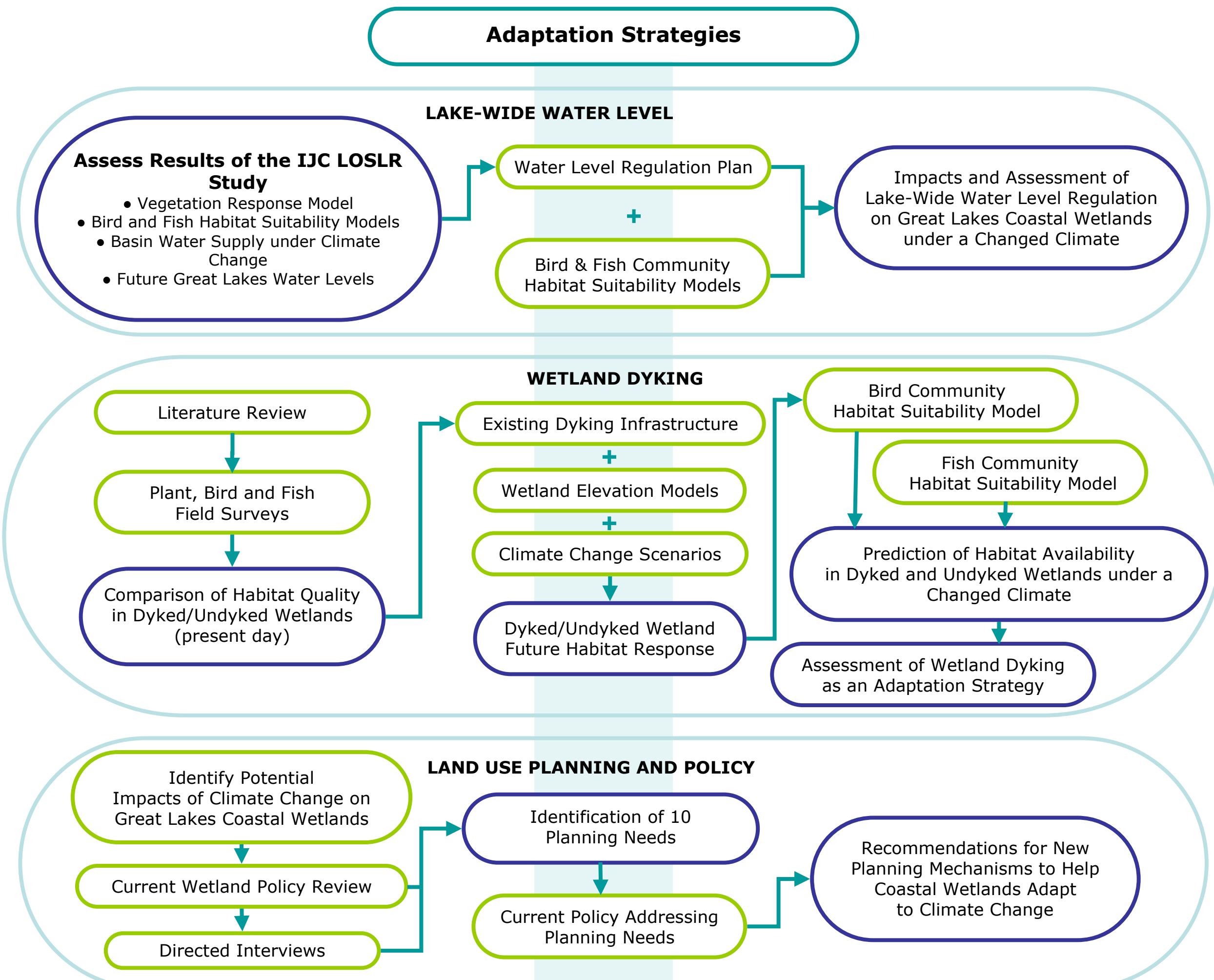


Figure 8.1 Flow diagram of approaches used to evaluate adaptation strategies as potential responses to climate-induced hydrological change

The evaluation of lake-level regulation stemmed from research conducted during the IJC LOSLR Study on the effects of lake-wide regulation of Lake Ontario. As part of the LOSLR Study, hydrologic modellers estimated potential changes in water supply to Lake Ontario and St. Lawrence River systems under different climate change scenarios. The anticipated changes in water supply were used to predict changes to Lake Ontario water levels using different regulation schemes, an unregulated scenario and several regulated scenarios. The predicted water levels were used to evaluate the effectiveness of large-scale water regulation as an adaptation strategy for ameliorating the effects of climate-induced changes to vegetation, birds, and fishes, as well as their habitats.

Dyking, and the associated small-scale regulation of local water levels within coastal wetlands, is a common management practice to maintain wetland plant diversity and extent. Wetland dyking, as an adaptation strategy to modify future lower water levels, was evaluated in a comparative study of dyked and undyked coastal wetlands with varying connectivity to the lower Great Lakes. The preliminary investigation of dyking effects included a thorough literature review of physical, chemical, and biological changes after dyking. This was followed by a detailed field study of wetland plant, bird, and fish communities in the comparative study. Also, predictive models of wetland vegetation generated for the climate change assessment (Chapter 4) were applied to selected dyked wetlands to evaluate conditions under future water level scenarios in closed systems. The ability of existing dyke infrastructure to operate under low water levels was also evaluated.

As a result of climate change, Great Lakes water level changes will result in changes to the distribution of wetlands along the Great Lakes shoreline and expose relatively large areas in some regions. Therefore, existing land use planning and policies were reviewed for their potential as adaptation strategies. Potential strategies included the protection of existing wetlands from increased development pressure as lands dry and the protection of transitional habitats that will be important as water level regimes change.

8.1 LAKE-WIDE WATER LEVEL REGULATION

Joel Ingram, Susan Doka, and Kathy Leisti

8.1.1 History of Lake Ontario Water Level Regulation

Water levels on Lake Ontario have been regulated since the completion of the Moses-Saunders hydropower dam in Cornwall, Ontario in 1960. At the time, hydropower, commercial navigation, domestic water use, and shoreline protection were considered in the development of a water regulation plan (referred to as Plan 1958D) but environmental effects were not. Because of unprecedented high water levels post-1960, deviation criteria and rules were developed to diverge from the 1958D regulation plan so that extreme events, usually flooding events, could be ameliorated. This plan is referred to as Plan 1958DD, which is Plan 1958D with deviations. The primary criteria used in the 1958DD water regulation plan have moderated long-term, high water level fluctuations on Lake Ontario. A consequence of this regulation has been that high lake levels normally experienced during high water-supply periods have been lowered and low lake levels during low water periods have been raised (Fay and Fan pers. comm.). The moderating effect of regulation is illustrated when historical mean August water levels since 1860 are compared with estimated unregulated August water levels (Figure 8.2).

As discussed in Chapters 4 to 7, long-term water level fluctuations are critically important to the maintenance of diverse and productive coastal wetland communities. Environmental impacts related to water level regulation were not addressed in the construction of the dam; therefore, the current water regulation plan does not contain environmental criteria that are used for rules governing regulation or planned deviations. The need for studying the effects of water level regulation on the natural environment was recognised during a mandatory review of the current regulation scheme and included in the LOSLR Study.

8.1.2 Lake Ontario-St. Lawrence River (LOSLR) Study

The LOSLR Study had specific objectives for developing, evaluating, and recommending updates to the original water regulation plan based upon an evaluation of impacts on various interests within the Lake Ontario-St. Lawrence River system. The LOSLR Study provided an opportunity to improve our understanding of water regulation impacts on coastal wetlands and nearshore biota using quantified relationships between water levels and wetland plant, bird, and fish communities (DesGranges *et al.* 2005; Doka *et al.* 2005; Wilcox *et al.* 2005).

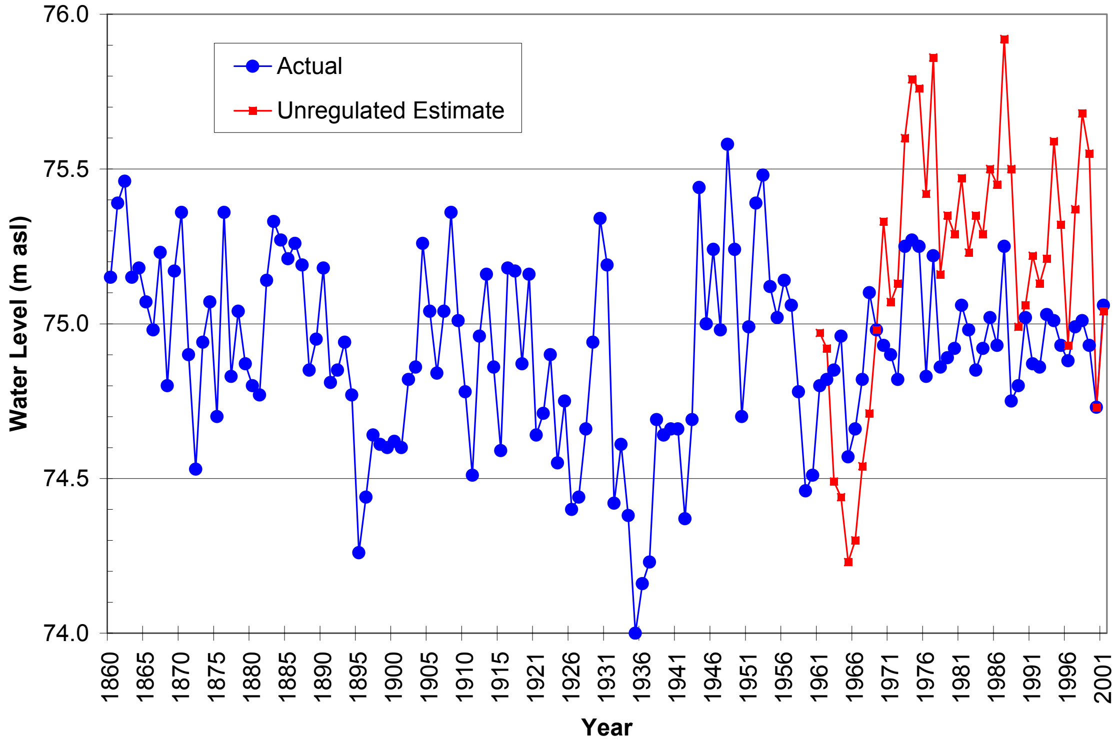


Figure 8.2 Mean August Lake Ontario water level record and estimated unregulated August water levels since 1960 (Environment Canada 2006)

A secondary objective of the LOSLR Study was to evaluate the performance of current (1958DD) and alternate (e.g. Plan A) water regulation plans under potential climate-induced changes in water supply. This component of the LOSLR Study provided an opportunity to evaluate the Lake Ontario regulation plans as a climate change adaptation strategy geared specifically to managing lake levels during lower water supplies. Specifically, current water level regulation could be employed to moderate the potential impacts of future climate-induced water level changes within the system on the nearshore zone.

The LOSLR Study produced theoretical 100-year Lake Ontario water level scenarios by applying water regulation plans to different water supply scenarios. Water supply scenarios included historic 1900-2000 supplies, and theoretical supplies based upon a 1000-year stochastic supply series using historic variability boundaries. The lowest 100-year supply period within this series was used in the regulation plan evaluations presented here. Relative comparisons among alternate regulation plans used the ecological models developed by the Environment Technical Work Group to predict the response of wetland plant, bird, and fish communities under each water level scenario. The results from this evaluation form the bases for our evaluation of lake-level regulation as an adaptation strategy to climate change for coastal wetlands (Figure 8.3).

In the evaluation, the current regulation plan (1958DD) held water levels higher than would occur without regulation during the first 50 years of the low water supply scenario (climate scenario). Even with regulation, water levels frequently dropped below 74.15 m, a lower water level target of the current regulation plan, indicating that water supplies during the first half of this time series were not sufficient to meet current plan conditions (Yee *et al.* 1993). Newly developed alternate water regulation plans (such as Alternate Plan A) were able to maintain water levels above current plan target levels (Figure 8.4) and, from this perspective, performed much better than the current plan under the 100-year low water supply scenario. The regulation plans were compared by using ratios of unregulated conditions, and either Plan A, or current regulation plan water levels. Plan A water levels were higher and unregulated levels much lower than the current plan levels; however, Plan A did not maintain natural long-term variability or cyclic patterns (Figure 8.5).

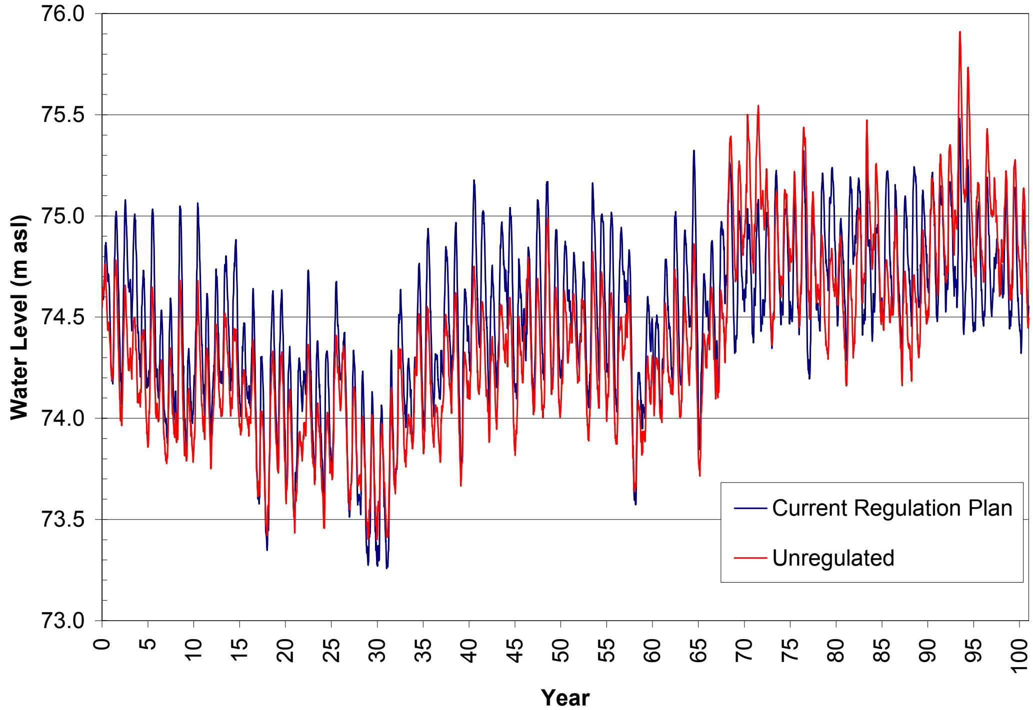


Figure 8.3 Estimated 100-year Lake Ontario water level scenarios under the current water level regulation plan (1958DD) and unregulated during a low water supply period within a 1000-year stochastic supply series (LOSLR Study) that represents a potential climate change condition

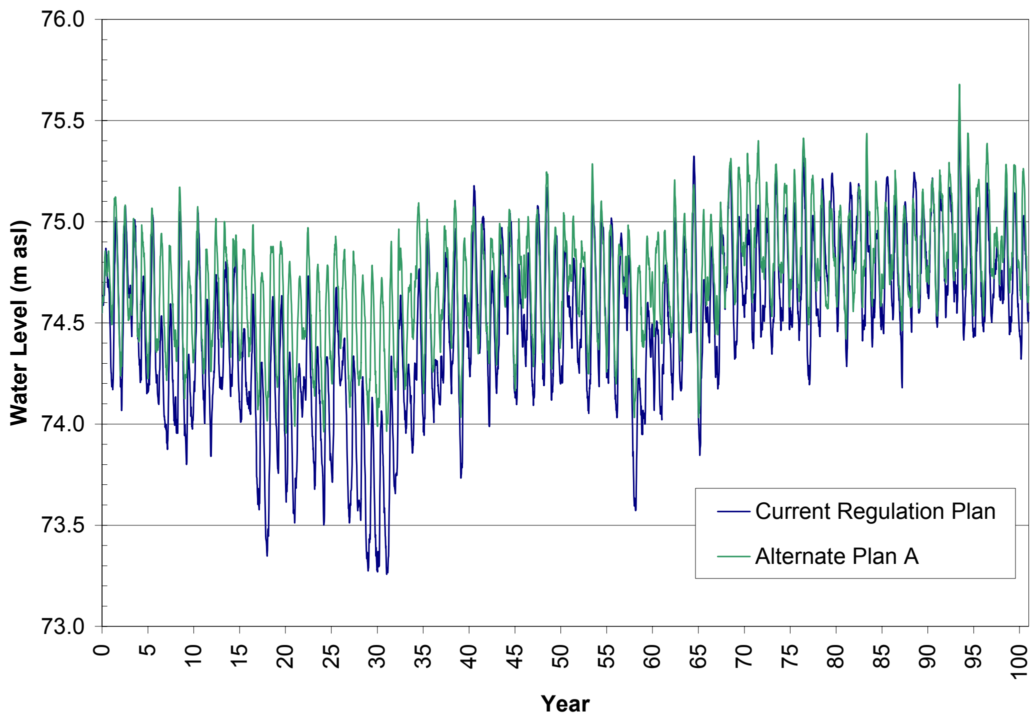
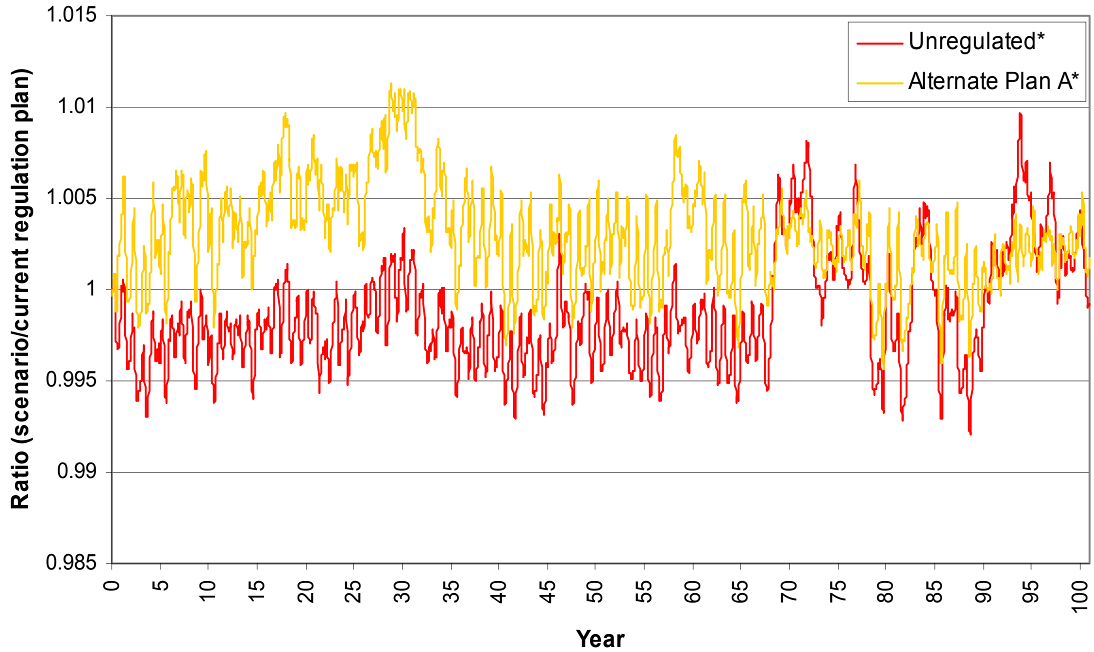


Figure 8.4 Estimated 100-year Lake Ontario water level scenarios under the current water level regulation plan and Alternate Plan A regulation during a low water supply period from a 1000-year stochastic supply series (LOSLR Study) used as a potential climate scenario



* >1 water levels lower, <1 water levels higher than current regulation plan

Figure 8.5 Ratio scores of unregulated and Plan A water levels relative to the current 1958DD regulation plan over 100-year Lake Ontario scenarios. All regulation plans were applied to a low water supply period from a 100-year stochastic supply series (LOSLR Study).

Temperature, although an input to fish habitat and population models for the Study, did not reflect anticipated warming trends under climate change in the low-supply, water level scenarios. Temperature is recognised as a major factor in nearshore fish production and fish distributions, but specific 100-year climate scenarios for Lake Ontario temperatures were not generated and historic temperature inputs to the environmental models could not be modified. The results presented here, therefore, reflect the changes in fish habitat due to water level fluctuations only. Of note, the fish responses, simulated using the historical temperatures of the past century in all regulation scenarios, showed annual and decadal year-class variability in different areas of Lake Ontario in response to annual temperatures and longer term warming and cooling trends. The effect was especially noticeable in the last 20 years of the century where warmwater fishes responded to increasing temperatures (Chu *et al.* 2005; Doka *et al.* 2005). The reader should bear this in mind when reviewing the results presented here for fish responses.

8.1.3 Environmental Assessment of Regulation Plans

Current and alternate water regulation plans were evaluated using environmental performance indicators (DesGranges *et al.* 2005; Wilcox *et al.* 2005; Doka *et al.* 2006). Performance indicators were developed using: (1) statistical associations between a hydrologic variable (e.g. peak annual water levels) and a biological attribute (e.g. area of meadow marsh), or (2) key output variables from habitat supply models or process-based population models linked to habitat availability. Suitable habitat availability, defined separately for different species or similar groups can be partly affected by hydrology. Both empirical and process-based models were applied to the 100-year water level scenarios, and aggregated or extrapolated to basin-level biological responses. The estimated biological response, or performance indicator (PI), from each plan was compared to the response from a base plan (usually 1958DD) using a selected ratio as a metric.

Alternate regulation plans were ranked using the metrics by comparing the magnitude of positive change measured by the ratios across a suite of performance indicators. Plans were compared by using ratios of aggregated 100-year PI outputs (e.g. the ratio of 100-year averages for each plan or annual ratios that were averaged). Because ratios were relative measures of a plan's performance compared to a base plan, a PI ratio score of >1 indicated an alternate plan that performed better than the current plan (or base plan). Conversely, ratio scores of < 1 indicated that the new plan performed worse than the base plan.

Table 8.1 summarizes ratio scores for key environmental PIs from the LOSLR Study. Key performance indicators were selected for their sensitivity to water level fluctuations and to represent key elements of the ecosystem response. Indicators included meadow marsh area, spawning habitat supply for warmwater fish guilds, nursery habitat for two key fish species, and reproductive success indices for several bird species. Ratios were considered significantly different from 1 (i.e. no difference between plans) at 5% difference for fishes and 10% difference for birds and plants.

Table 8.1 Environmental performance indicators and 100-year aggregate ratios of evaluated plan (unregulated or plan A) relative to the current plan 1958DD (Low Veg 18°C: low vegetation and 18°C thermal spawning preference fish guild; High Veg 24°C: high vegetation and thermal spawning preference fish guild; Low Veg 24°C: low vegetation, high thermal spawning guild). Red indicates a greater than 10% difference and blue indicates a greater than 5% but less then 10% difference (LOSLR Study).

Environmental Performance Indicator	Unregulated*	Alternate Plan A*
Meadow Marsh - total surface area (ha) during selected supply-based periods	1.76	1.17
Low Veg 18°C fish guild - spawning habitat supply (ha)	0.99	0.97
High Veg 24°C fish guild - spawning habitat supply (ha)	0.95	1.17
Low Veg 24°C fish guild - spawning habitat supply (ha)	0.99	0.97
Northern Pike - YOY recruitment (#/ha index)	0.93	0.95
Largemouth Bass - YOY recruitment (#/ha index)	1.01	1.00
Least Bittern - reproductive index	1.08	0.60
Virginia Rail - reproductive index	1.09	0.64
Black Tern - reproductive index	1.09	0.63

***** >1 plan is performing better than 1958DD (current) regulation, < 1 plan is performing worse than current regulation plan

8.1.3.1 Plant and Bird Responses

Most PIs responded positively to the unregulated water level scenario, despite the lower water levels associated with this plan compared to the current plan (Table 8.1). The wider long-term water level range for the unregulated plan increased the surface area of meadow (Figure 8.6) and emergent marsh habitat. The wetland bird performance indicators (based on Least Bittern, Virginia Rail, and Black Tern that nest primarily in emergent marsh habitat) also responded positively to an increase in area of emergent marsh (Figure 8.6) (DesGranges *et al.* 2005). In contrast, Alternate Plan A performed very poorly for the wetland bird PIs due to the reduction in the long-term water level elevation relative to the current plan.

8.1.3.2 Fish Response

The fish PIs (i.e. spawning habitat supply for different guilds, and northern pike and largemouth bass recruitment) were generally less sensitive to hydrologic differences among the water level scenarios than birds and emergent vegetation (Table 8.1). However, species that prefer higher temperatures and vegetation for spawning showed a marked difference in their spawning habitat supply response to unregulated and alternate plans (Figure 8.6). The northern pike PI was the only indicator to show a significant negative difference (>5% change in ratio score) relative to the current plan for the unregulated low-supply scenario over 100 years (Figure 8.6).

In general, the unregulated plan and Alternate Plan A both performed worse for most fish indicators than the current plan under low supply conditions. However, within the bounds of acceptable error established by the LOSLR Study, these decreases were not significantly different than the current plan. The negative trend was likely due to a decreased availability of habitat in general under the unregulated condition and to a loss in vegetated habitat under the less variable Plan A scenario.

When averaged over the 100-year period or presented as a time series, the ratio score or performance indicator was still an aggregate measure of the response across the whole lake. Responses in PIs differed regionally, and this is particularly important when considering individual wetland responses to climate change. The aggregate measure was a weighted combination across regions for the fish indicators that reflected the differing levels of production across the lakes, so the whole-lake responses are not directly related to regional responses. Largemouth bass and northern pike (Table 8.2) population indicators, like total population density (abundance), total biomass, and YOY recruitment showed subtle differences across metrics and more pronounced differences across regions when comparing the effects of regulation during low supplies.

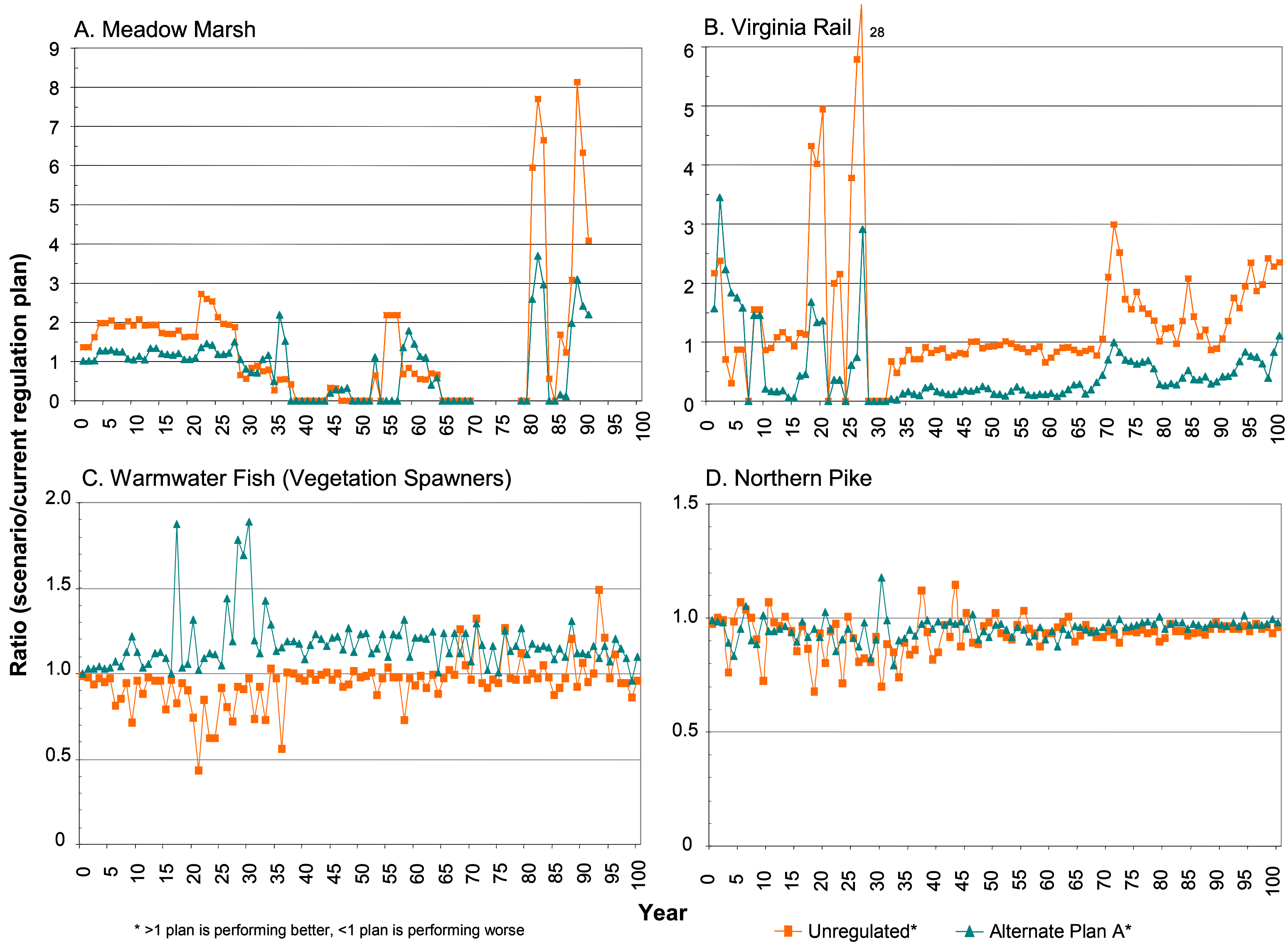


Figure 8.6 Annual ratio scores of environmental performance indicators for Lake Ontario (A) meadow marsh coverage, (B) Virginia Rail reproduction, (C), habitat supply for fish in the 24°C average - high macrophyte spawning preference guild, and (D) YOY recruitment for northern pike. Ratios were calculated for the unregulated plan and Plan A, relative to the current regulation plan (1958DD), for a low water supply scenario representing climate change.

Table 8.2 Population level environmental performance indicators for largemouth bass and northern pike, determined for six Lake Ontario locations. The values are 100-year aggregated ratio scores relative to the current plan based on numbers per hectare (abundance and YOY recruitment) and kg/ha (biomass). Blue scores indicate a >5% difference between plans and red indicates >10% difference.

	Unregulated*			Alternate Plan A*		
Location	**Total Abundance**	**Total Biomass**	**YOY Recruitment**	**Total Abundance**	**Total Biomass**	**YOY Recruitment**
Largemouth Bass						
Bay of Quinte	1.04	1.04	1.04	*0.96*	*0.96*	*0.96*
Presqu'ile	*0.87*	*0.87*	*0.90*	1.28	1.22	1.75
Lake Ontario - north central	*0.91*	*0.91*	*0.93*	1.11	1.10	1.19
Lake Ontario - west shore	*0.96*	*0.96*	*0.95*	1.07	1.07	1.11
Lake Ontario - south shore	*0.77*	*0.77*	*0.78*	1.48	1.38	2.24
Lake Ontario - outlet basin	*0.99*	*0.99*	1.00	1.05	1.05	1.10
Lake Ontario - all sections			1.01			1.00
Northern Pike						
Bay of Quinte	1.01	1.01	1.01	*0.95*	*0.95*	*0.95*
Presqu'ile	*0.97*	*0.97*	1.00	*0.87*	*0.88*	*0.93*
Lake Ontario - north central	*0.84*	*0.84*	*0.84*	1.16	1.18	1.18
Lake Ontario - west shore	*0.88*	*0.88*	*0.88*	1.10	1.10	1.11
Lake Ontario - south shore	*0.69*	*0.70*	*0.70*	1.08	1.08	1.10
Lake Ontario - outlet basin	*0.79*	*0.79*	*0.80*	*0.88*	*0.89*	*0.88*
Lake Ontario - all sections			*0.927*			

* >1 plan is performing better, *< 1 plan is performing worse*

8.1.4 Conclusion

The LOSLR environmental study results highlighted the importance of considering whole-lake responses to changes in Great Lakes water levels. Performance indicator responses indicated that an unregulated scenario that allows water levels to fluctuate through high and low water levels cycles was better for several biological communities than a regulation plan that maintained higher water levels but reduced the inter-annual variability. Management of whole-lake water levels benefited some biological communities and negatively impacts others. As explained in Section 8.2 of this chapter, this is the same issue that was identified in evaluation of wetland dyking as an adaptation strategy. However, water level manipulation at a lake level represented a much more complex issue, with large scale, spatially explicit consequences. A lake-wide assessment provided a landscape perspective of potential changes in wetland functions and values and amalgamated overall changes. It also provided a level of reference necessary for setting basin level objectives, climate change adaptation, and coastal wetland conservation planning.

8.1.5 References

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8.2 Evaluation Of Current Wetland Dyking Effects on Coastal Wetlands and Biota

Maggie Galloway, Lynn Bouvier, Shawn Meyer, Joel Ingram, Susan Doka, Greg Grabas, Krista Holmes, and Nicholas Mandrak

Wetlands depend on the variability in their hydrological regime to maintain vegetative diversity and species richness (Jaworski *et al.* 1979; Keddy and Reznicek 1986; Quinlan and Mulamoottil 1987; Casanova and Brock 2000). Long-term fluctuations in Great Lakes water levels can significantly impact coastal wetland extent and distribution. For example, during high water levels, wetland habitat, particularly in wetlands with restricted upland borders, has been lost along Lake Erie (Sherman *et al.* 1996; Gottgens *et al.* 1998). Further, while some wetlands may advance lakeward during low water levels, if suitable off-shore slope and substrate conditions exist, other wetlands will dry up due to the absence of these critical factors. This unpredictable natural variability in water levels, combined with extensive and ongoing drainage of wetlands for agriculture and urban development, has prompted the enclosure of many Great Lakes coastal wetlands by artificial dykes to preserve specific habitat characteristics.

8.2.1 Dyking in the Great Lakes

Dyking a wetland involves modifying the existing hydrological connection between the wetland and its water source by a human-made barrier, designed to alter the inflow or outflow of water and to protect the wetland from direct lake influences (Environment Canada 2002). Dyked wetland hydrology becomes primarily regulated through the use of pumps, underground flumes, gravity-flow gates, or stop-logs (Sherman *et al.* 1996). This infrastructure allows manipulation of water levels in the wetland to achieve specific vegetation diversity and interspersion that is desirable to wildlife, particularly waterfowl (Robb and Mitsch 1990; Payne 1992). Historically, wetland dyking was promoted by waterfowl hunters and private hunting clubs and organizations, and this motivation continues today. In the early 20th century, wetlands of western Lake Erie were gaining fame for the waterfowl hunting opportunities they provided while at the same time, wetland losses in the area were extensive due to draining and filling for land conversion to agriculture and urban development. As a result, many wetlands were dyked to preserve waterfowl hunting opportunities (Herdendorf 1992). Throughout the 20th century, periods of natural high water levels on the Great Lakes resulted in the construction of dykes to protect and manage waterfowl habitat. While only approximately three percent of coastal wetlands in Lake Ontario are dyked, at least 31% of the remaining wetlands in Lake Erie are dyked, including between 77 and 85% of Ohio's coastal wetlands (Robb and Mitsch 1990; Wilcox and Whillans 1999). Coastal wetlands along Lake St. Clair have been equally affected with almost half of the remaining wetlands in eastern Lake St. Clair being dyked by the early 1980s (McCullough 1985).

Payne (1992) provides an extensive review of water level management techniques to manage wildlife habitat in wetlands. Water level managers need to experiment to understand soil and water characteristics, the natural hydrograph, seed bank composition, and the response of vegetation to water level manipulation. If required, water level regimes within dyked wetlands can be managed differently than those in wetlands connected to the lake depending on yearly water supply and the habitat characteristics desired.

In addition to regular water level management within a dyked wetland, a controlled water level disturbance (approximately every 5 years) has been recommended to maximize the diversity of wetland habitat (Payne 1992). In general, full wetland drawdowns are recommended when emergent vegetation cover has declined and open water dominates. Conversely, wetland flooding is used to kill dense stands of emergent vegetation, and, thereby, increase habitat interspersion by creating irregular pockets of open water habitat within emergent vegetation.

8.2.2 Benefits of Wetland Dyking

The most commonly cited benefit of dyking a wetland is the ability to enhance, restore, or create habitat. However, what comprises "desirable habitat" is different depending on the perspectives of humans and the species or biotic communities of interest (Harris *et al.* 1983). Habitat quality can be defined for a single species, a community of species (i.e. bird or fish community), or an entire ecosystem (i.e. wetland) (Ball 1985). Historically, wetlands were often managed for single species or a specific group of species. This way, the habitat requirements of one species could be determined and then attempts could be made to create and manage those requirements. It was realized, however, that many groups or guilds of species have similar habitat requirements (i.e. waterfowl and shorebirds) (Rundle and Fredrickson 1981). Thus, by managing wetland habitat for a particular species, many other species were also benefiting. Ball (1985) provided a review of studies documenting positive correlations between wetland management for waterfowl and benefits to other groups of species.

Currently, management of dyked wetlands in Ontario is changing to a more inclusive ecological community approach. Some publicly owned dyked wetlands, such as National Wildlife Areas at Long Point and Lake St. Clair, are being managed to maintain high plant and structural diversity by creating well interspersed habitats. Consequently, many species of birds, fishes, amphibians, reptiles, and mammals benefit from these dyked wetlands. Ducks Unlimited Canada has been a leader in wetland conservation, dyke project engineering and management for decades and in recent years, has become a strong supporter of overall wetland conservation and wetland management from an ecosystem perspective.

Great Lakes coastal wetlands are usually located at the outlets of large watersheds. In urban and agricultural regions, this location often results in heavy sediment deposition leading to high turbidity. As a result, light penetration through the water column declines and reduces the establishment of submerged aquatic vegetation (Carter and Rybicki 1985; Chambers and Kalff 1987; Havens 2003). This results in a loss of critical food and cover habitat for fishes and other wetland-dependent wildlife. Dykes, however, can provide a barrier to sediment deposition by channelling watershed flow around the wetland. This technique was recently used at Oshawa Second Marsh on Lake Ontario. In the first year after dyke construction, new beds of submerged aquatic vegetation emerged as a result of low turbidity (Environment Canada 2004).

In many Great Lakes coastal wetlands, feeding and spawning activities of common carp increase turbidity and uproot vegetation reducing submerged aquatic habitat (Chow-Fraser 1998; Sager *et al.* 1998). Dyked wetlands with selective fish passage structures can improve turbidity by eliminating the entry of common carp into the wetland from the lake and possibly create refuge from dreissenids for native mussels (Zanatta *et al.* 2002). For example, these structures were successfully installed at Oshawa Second Marsh on Lake Ontario and at Metzger Marsh on Lake Erie (Wilcox and Whillans 1999).

8.2.3 Problems Associated with Wetland Dyking

While managing a dyked wetland can have ecological benefits, there are also inherent problems associated with wetland dyking. By modifying a wetland's hydrological connection between the watershed and the lake, physical and chemical functions of a wetland can be lost. For example, the ability of a dyked wetland to convey and store flood waters may be altered compared to an undyked wetland. In addition, diverting natural watersheds around wetlands may reduce their functions of sediment control, and nutrient and contaminant cycling (Wilcox 1995).

Many coastal wetlands act as a sink for some chemicals, nutrients, contaminants, and suspended solids because of their location at the outlet of large agricultural and urban watersheds (Heath 1992). Within these wetlands, many biogeochemical processes naturally occur. Some of these processes are positive, such as denitrification and phosphorus retention, while others can result in undesirable by-products such as methylmercury or hydrogen sulphide (Mitsch and Gosselink 2000). When a wetland is dyked, chemical and nutrient cycling and sediment transport are altered. Usually dyked wetland design and management does not consider the important role of wetlands in removing contaminants from runoff. Thus, when runoff from a watershed bypasses a dyked wetland through drainage ditches, nutrients and contaminants are directly flushed into the lake (Gottgens and Liptak 1998). As a result, local concentrations of nitrates and phosphorus in a lake can increase causing excessive algal growth and euthrophication.

When a dyked wetland is left unmanaged and remains isolated from the natural flushing process between the lake and watershed, stagnant conditions create anaerobic environments. These low oxygen conditions promote the production of methylmercury in wetlands (Zillioux *et al.* 1993). Subsequently, if an isolated dyked wetland or other body of water is drawn down and then reflooded, methylmercury is likely to become available for uptake in the food chain, and thus, potentially bioacculmulate in higher trophic levels (Brigham *et al.* 2002).

The most commonly cited problem associated with wetland dyking is a lack of fish access from the lake into coastal wetlands for feeding, cover, spawning, and nursery habitat (Johnson *et al.* 1997). Many fish species rely on Great Lakes coastal wetlands during various stages of their life history (Jude and Pappas 1992). For example, northern pike commonly spawn in flooded meadow marsh habitat. In addition, fish larvae may be trapped in dyked wetlands during filling operations, and are lost to the lake population. Similarly, common carp may remain trapped through to adulthood in dyked wetlands and cause management problems by uprooting vegetation and increasing turbidity (Wilcox 1995). To alleviate some of these problems, fish ladders and other selective fish passage structures have been installed in some dyked wetlands to allow fish movement between the wetland and the lake (Wilcox and Whillans 1999). To be effective, however, these structures must be designed with consideration of site-specific fish community objectives and require ongoing maintenance and monitoring.

Well-managed dyked wetlands can provide excellent interspersed habitat between mixed emergent vegetation and open water. However, the transitional wetland habitat provided by a continually changing boundary between land and water is typically reduced (Maynard and Wilcox 1997). For example, many fully enclosed dyked wetlands do not include transitional habitats such as meadow marsh, shrub, treed swamps, or mudflats. These habitats provide essential nesting, breeding, and foraging sites for many species of mammals, birds, reptiles, and amphibians. When assessing dyking as an adaptation strategy to climate change, the ongoing requirement for these habitats by many species must be considered.

Dyking a wetland can be a major physical alteration with significant ecological consequences. The design and engineering process must be carefully planned to ensure successful dyke operation and wetland functioning. The surficial geology, soils, existing vegetation, watershed hydrology, climate, and wind and wave forces must be considered during the design process or dyke washout, breaching, erosion, or stability may become a problem.

To manage a dyked wetland effectively and to enhance habitat, well maintained pumps and intakes must be established with trained technicians on site. This requires ongoing financial and personnel resources. In cases where active management does not occur or funding does not allow for ongoing maintenance, water levels in dyked wetlands may become stagnant allowing vegetation to either expand and form monotypic stands, or be flooded out depending on the water depth. Even with active management, extremely low Great Lakes water levels can prevent the source lake water from reaching the wetland through the pumping system. As a result, additional resources may be required to dredge intake channels or extend water intake pipes further into the lake.

Finally, dyked wetlands may facilitate the expansion of some invasive species. Although active management of water levels, including consistent spring and early summer flooding can limit the establishment of purple loosestrife (Weiher *et al.* 1996). The rapid expansion of invasive common reed is a problem in many Great Lakes coastal wetlands (Wilcox *et al.* 2003), and it often germinates and expands rapidly in disturbed areas to form highly dense monotypic stands. These stands are often impenetrable to some wetland-dependent wildlife, thus providing limited habitat value for these species (Marks *et al.* 1994; Benoit and Askins 1999).

Climate change is projected to lead to lower mean lake levels in the Great Lakes, an increase in the frequency of extreme precipitation events, and changes in the timing of the annual hydrograph in each of the Great Lakes. These factors will have a significant effect on the hydrology of both dyked and undyked wetlands on the Great Lakes. In some cases, the ability to manage water levels within a dyked wetland may provide an opportunity to preserve specific wetland functions where they might otherwise be lost because of climate change.

8.2.4 Comparison of Emergent Marsh Vegetation and Bird Communities between Dyked and Undyked Coastal Wetlands

The viability of wetland dyking as a strategy to help coastal wetland emergent marsh bird and vegetation communities adapt to hydrological variability caused by climate change was assessed through:

- A comparison of bird and vegetation communities in paired dyked and undyked wetlands on the lower Great Lakes to assess whether differences in habitat quality exist between wetland types;
- An assessment of the potential response of the wetland communities in an undyked state to hydrological variability induced by climate change using wetland basin elevation models, and vegetation and bird models; and
- An evaluation of the ability of dyked wetland infrastructure to operate under altered lake levels caused by a changed climate.

8.2.4.1 Study Sites

Two sets of paired coastal wetlands (dyked and undyked) on each of Lakes Ontario, Erie, and St. Clair were selected for a total of 12 wetlands (Figure 8.7; Table 8.3). Each pair consisted of a dyked and undyked wetland. All dyked wetlands were isolated from the lake and have management structures to permit control of water levels. Four of the six dyked wetlands were completely enclosed wetland cells. One was a partial dyke isolating the marsh from the lake but maintaining its connection to upstream watershed inputs, and one had a water control structure at the outlet to the lake to allow management of water levels.

Figure 8.7 Location of dyked and undyked wetlands on the lower Great Lakes evaluated in the CCIAP study

Table 8.3 Dyked and undyked coastal wetland pairs used to compare marsh bird and plant communities

Wetland Pair	Wetland Type	
	Dyked	**Undyked**
Lake Ontario		
Lynde Shores	Cranberry Marsh	Lynde Creek Marsh
Amherst Island Marsh	Dyked	Undyked
Lake Erie		
Big Creek NWA	Dyked	Undyked
Hillman Marsh	Dyked	Undyked
Lake St. Clair		
St. Clair NWA 1	East Dyke	Mitchell's Bay
St. Clair NWA 2	West Dyke	St. Clair West Shoreline

To eliminate potential bias between dyked and undyked wetlands because of geographic location, climate, and adjacent land use, dyked and undyked pairs were selected from wetlands where a dyke was constructed within a large contiguous wetland, thereby leaving dyked and undyked wetlands adjacent to one another. In areas where this was not possible, dyked and undyked wetlands were located as close as geographically possible to one another and had similar characteristics (i.e. size, elevation, and geomorphic features). Identical field surveys were completed in each pair to facilitate comparative analysis of data and results.

8.2.4.2 Methodology

Elevation surveys

The development of habitat response models within a wetland system required topological and bathymetric surveys of high vertical resolution and accuracy. Establishing centimetre-level vertical precision was necessary along these low gradient systems where a difference in 10 cm in elevation could create a quantitatively different environment. Thus, a rigorous elevation surveying protocol was implemented within the dyked and undyked wetland sites. The output product was a high accuracy DTM of each wetland site created from newly collected orthometric heights.

The wetland sites in this study had limited accessibility due to varying patches of dense vegetation and hemi-marsh conditions, requiring both boat and foot access. A survey design was developed to derive optimal accuracy, subject to the constraints of efficiency and cost effectiveness. Surveys were completed between April and June 2004. The early spring timing ensured vegetation growth was at a minimum to increase sampling efficiency. Survey density was wetland area dependant, with most sites having around 200 points per wetland.

Survey points were randomly selected using GIS. A 20-m vector grid was created in *ArcGIS 8.3* (ESRI 2003) and superimposed on 1:10,000 digital colour infrared aerial images (resolution of 0.5 m ground pixel) of the 12 wetland sites. Each node, or grid intersection, was a potential survey point. The nodes were randomly queried to select 200 points (or less, for smaller sites) distributed across the wetland. Sample point coordinates were uploaded into a Trimble GeoXT handheld Global Positioning System (GPS) unit, to locate position in the field.

Field elevations were collected using a Thales Navigation Promark 2 surveying system. Utilizing kinematic GPS technology, a base station established a control point while rover units collected relational horizontal coordinates (x,y) and vertical heights (z). This survey system provided centimetre-level survey accuracy and metre-level positional accuracy. A Trimble GeoXT GPS unit was used to correct the horizontal accuracy of the Promark 2 base station to centimetre-level. At each point, a vegetation community attribute (emergent marsh, open water, or meadow marsh) was also stored in the GPS. The rover units held a two minute occupation at each point and sampling was planned during peak times of satellite coverage.

Post-processing used Thales Navigation's (2002) *Ashtech Solutions* software. This software is specific to the Thales Survey Systems and provides automated corrections for satellite data. To correct ellipsoid heights to a reference height on the ground, the base station z height was corrected to lake level at time of survey. In the dyked wetlands, the baseline water level was corrected using laser level surveying between the actual lake level and the water level behind the barrier. All orthometric heights were calculated to IGLD85, in metres, a commonly used vertical datum for the Great Lakes. All x,y coordinates were established in zone specific UTM, NAD83. Further post-processing included performing a least-squares adjustment and calculating vector networks, which are performed to improve accuracy of the collected data and determine confidence levels for each elevation point.

To determine the best interpolator algorithm for the data, the *Geostatistical Analyst* extension for *ArcGIS 9.0* (ESRI 2004) was utilized for its exploratory data analysis (ESD) tools. Analyzing the distribution of the data and trends as well as characterizing the error and variability of the predicted surface was of primary importance in selecting a data interpolation method. A deterministic model, Radial Basis Function (RBF), Spline with Tension proved to be the best overall predictor for the elevation data. This method captures the local and global trends in the surface, more representative of an elevation surface, and can tie exactly to the high accuracy data points collected.

This model was applied to the elevation data for each site to generate a terrain surface in raster format. The created surface was clipped to either the sample area or the wetland boundary to minimize prediction error. This wetland basin model provided the baseline elevation to which all water level scenarios were applied.

Vegetation surveys

Vegetation sampling occurred in late July, 2004 to coincide with the period of peak growth and ensure that percent cover at each wetland was exclusive of seasonality and directly related to wetland characteristics. The focus of the vegetation sampling was on emergent marsh and open water vegetation communities. Most dyked wetlands have little or no meadow marsh, shrub, or treed swamp communities, and therefore, these communities were not surveyed. This stratified sampling made results more directly comparable to support a paired study design.

Twenty-five vegetation survey points for each of the emergent marsh and open water communities were randomly sub-sampled from the pool of potential elevation points identified during bathymetric surveys. At each point, total vegetation cover and percent cover of each plant species within a one metre by one metre quadrat were estimated and recorded. In addition, water depth, sediment type, and sediment depth were also recorded in each quadrat.

Marsh bird surveys

In 2004, each dyked and undyked wetland was visited three times with survey cycles beginning May 17, June 7, and June 28, ensuring at least 10 days between surveys at the same location. This survey period coincided with the peak egg-laying period for most marsh breeding birds in Great Lakes coastal wetlands. Point counts were conducted in a circular survey area with a 50-m radius. The number of survey stations per wetland was limited by wetland size; however, an equal number of survey stations were placed in each dyked and undyked wetland pair. Sample stations were randomly placed within emergent marsh patches (comprising at least 20% but not more than 80% cover) in each wetland. Stations were separated by at least 250 m (to ensure independence among sample stations).

Surveys occurred from one half-hour before sunrise for five hours. Surveys were conducted under good weather conditions (i.e. low winds (< 3 on Beaufort Wind Scale), no or trace precipitation (light drizzle), and high visibility (no fog)). Each point count lasted for 10 minutes and consisted of five minutes of passive listening followed by five minutes of song broadcasting using Bird Studies Canada's Marsh Monitoring Program tape. All bird species heard and seen within the survey radius were recorded. Marsh nesting obligate bird species outside of the survey radius were also recorded with an estimate of their distance from the surveyor. All birds were categorized as either a marsh forager or nester based on their use of emergent marsh (Figure 8.8).

At the end of each bird survey, water depth (cm), total vegetation cover, percent cover of standing water, and dominate plant species were recorded. Where available, nest information was also recorded for selected marsh nesting obligate species including Least Bittern, Black Tern, and American Coot. Nest information included nest stage, number of eggs or chicks, vegetation community, water depth below the nest, and height of the nest above the water.

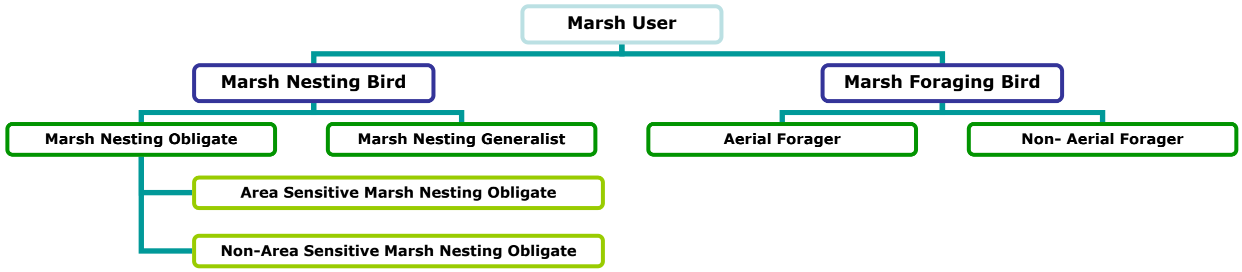


Figure 8.8 Marsh user categories for marsh bird species observed based on emergent marsh nesting and foraging ecology

8.2.4.3 Statistical Analyses

Comparisons between dyked and undyked wetland bird and vegetation communities (i.e. emergent marsh and open water communities) were analyzed separately using factorial multivariate analysis of variance (MANOVA). Factors for each MANOVA were the same, site and treatment (dyked or undyked). Site-treatment interactions were also examined.

Dependent variables for each community are listed in Table 8.4. For bird communities, abundance indices were calculated from the maximum number of all species pooled for each bird community from each survey station among the three visits. Species richness values represented the cumulative number of species pooled for each bird community from each survey station for the three visits.

Table 8.4 Response variables examined in relation to biotic communities sampled

Marsh Breeding Birds	Emergent Wetland Vegetation	Open Water Vegetation
Marsh User - Index of Abundance	Invasive Species Richness	Invasive Species Richness
Marsh Nesting Bird (MNB) - Index of Abundance	Invasive Plant Relative Coverage[†]	Invasive Plant Relative Coverage
Marsh Nesting Obligate (MNO) - Index of Abundance	Native Vascular Plant Species Richness	Native Vascular Plant Species Richness
Marsh Nesting Generalist (MNG) - Index of Abundance	Native Vascular Plant Coverage	Native Vascular Plant Coverage
Area Sensitive MNO - Index of Abundance	Species Richness of Species at Risk (SAR)*	Species Richness of Species at Risk (SAR)[*]
Non-Area Sensitive MNO - Index of Abundance	Coverage of SAR*	Coverage of SAR[*]
Marsh Foraging Bird - Index of Abundance	Vascular Plant Species Richness	Vascular Plant Species Richness
Aerial Forager (AF) - Index of Abundance	Vascular Plant Coverage	Vascular Plant Coverage
Non-Aerial Forager (NAF) - Index of Abundance	Colonial Autotroph Species Richness	Colonial Autotroph Species Richness
Marsh User Cumulative Species Richness	Colonial Autotroph Coverage	Colonial Autotroph Coverage
MNB Cumulative Species Richness	Filamentous Algae Coverage[*]	Filamentous Algae Coverage[*]
MNO Cumulative Species Richness	Persistent Plant Species Richness	Persistent Plant Species Richness
MNG Cumulative Species Richness	Dead Persistent Plant Coverage[†]	Dead Persistent Plant Coverage
Area Sensitive MNO Cumulative Species Richness	Total Areal Coverage[†]	Total Areal Coverage
Non-Area Sensitive MNO Cumulative Species Richness		
Marsh Foraging Bird Cumulative Species Richness		
AF Cumulative Species Richness		
NAF Cumulative Species Richness		

[*] Variables that deviated severely from the normal distribution (by inspection) or had no variance were omitted from the analysis; [†] variables that were arcsine transformations to reduce non-normality

In many cases, dependent variables were not normally distributed. Most of these variables were retained for analysis as Zar (1999) notes that MANOVA is particularly robust to non-normality. Variables that deviated severely from the normal distribution (by inspection) or had no variance were omitted from the analysis. Coverage variables for vegetation communities (measured as percent coverage within the one metre by one metre quadrat) that exhibited non-normality were converted to ratios and arcsine transformed.

MANOVAs and *post hoc* testing were done with *STATISTICA* (StatSoft, Inc. 2003). Significant differences in MANOVA were detected through calculation of Wilks' likelihood ratio. When detected, Tukey HSD (Honestly Significantly Different) was used to identify which variables exhibited statistically significant differences at $p<0.05$.

8.2.4.4 Bathymetric Survey Results

Table 8.5 summarizes the total number of points surveyed for bathymetric elevation at each wetland. These points were applied to generate wetland basin elevation models (see Section 8.2.6) designed to assess whether each dyked and undyked wetland can adapt to climate-induced water level change.

Table 8.5 Number of points surveyed per wetland for bathymetric data

Wetland	# Points surveyed
Lake Ontario	
Cranberry Marsh (Dyked)	131
Lynde Creek Marsh (Undyked)	213
Amherst Island Marsh Dyked*	76
Amherst Island Marsh Undyked*	113
Lake Erie	
Big Creek NWA Dyked	181
Big Creek NWA Undyked	199
Hillman Marsh Dyked	167
Hillman Marsh Undyked	237
Lake St. Clair	
St. Clair NWA East Dyke	176
St. Clair NWA West Dyke	152
St. Clair West Shoreline (Undyked)	238
Mitchell's Bay (Undyked)	278

*Accessibility and safety issues prevented the collection of 200 points at these sites.

8.2.4.5 Marsh Bird Communities Results

A total of 3,751 birds of 62 species were observed within the 50-m survey area radius during all surveys (see Appendix 8.1 for bird species and marsh user categories). There were 2,269 birds of 42 species recorded as mapped observations (i.e. birds that landed within the 50-m survey area). Marsh nesting generalists such as Red-winged Blackbird and Common Yellowthroat were more common (48% of total mapped observations) than marsh nesting obligates such as Black Tern and Virginia Rail (37% of total mapped observations) and mapped aerial foragers such as Tree Swallow and Belted Kingfisher (15% of total mapped observations).

The most commonly recorded bird species were Red-winged Blackbird and Marsh Wren. However, Swamp Sparrow, Common Moorhen, Canada Goose, Mallard, and Wood Duck were also recorded in relatively high numbers at specific wetlands. The nationally threatened Least Bittern was recorded within the 50-m survey radius on three occasions in dyked wetlands (once at Long Point and twice within the St. Clair NWA west dyke).

Comparison of marsh bird communities between dyked and undyked wetlands

Overall differences in dependent variables between dyked and undyked wetland were compared as well as differences in specific wetland pairs. When data for all dyked and undyked wetlands were pooled and compared, dyked wetlands showed statistically higher indices of abundance for marsh users, marsh nesting birds, marsh nesting obligates, marsh nesting generalists, area sensitive marsh nesting obligates, and non-area sensitive marsh nesting obligates than undyked wetlands. There were no significant differences in the indices of abundance for total marsh foragers, aerial foragers, and non-aerial foragers between dyked and undyked wetlands (Figure 8.9).

Cumulative species richness of marsh nesting birds, marsh nesting obligates, non-area sensitive marsh nesting obligates, and non-aerial foragers was significantly higher in dyked than in undyked wetlands (Figure 8.10). Cumulative species richness of aerial foragers, however, was higher in undyked than dyked wetlands.

Few significant differences in the indices of abundance and cumulative species richness of birds were observed between dyked and undyked wetlands at the paired-site level (Table 8.6). The dyked wetland on Amherst Island had a higher index of abundance of area-sensitive marsh nesting obligates than its paired undyked wetland, while the undyked wetland at Long Point had higher cumulative species richness of marsh nesting generalists than the dyked wetland. More individuals and species of aerial foragers were recorded in the undyked wetland at Mitchell's Bay on Lake St. Clair than in its dyked wetland pair. Cranberry Marsh (dyked) and Lynde Creek (undyked), despite being located only a few hundred metres apart, showed more differences in response variables (8 of 18 different variables analysed) than did all other sites.

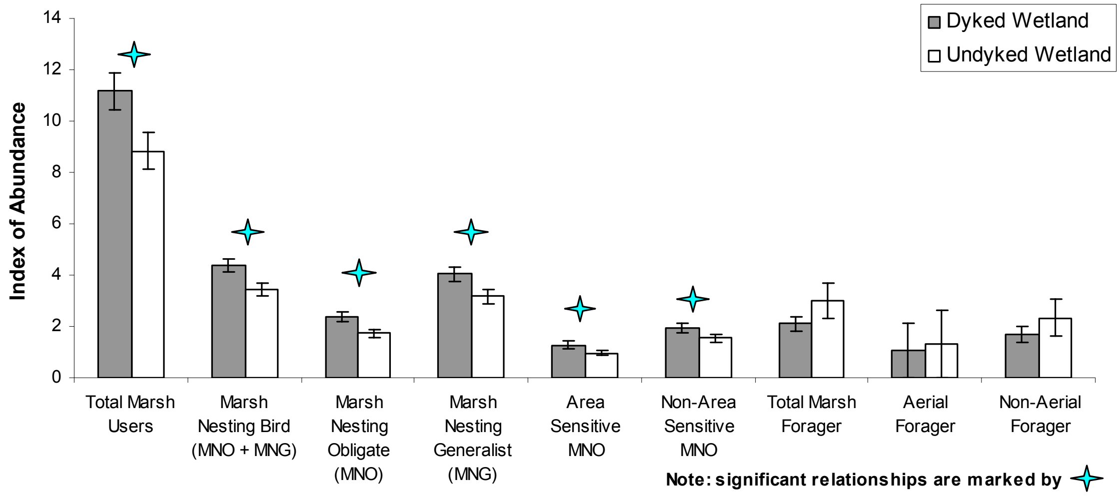


Figure 8.9 Indices of bird abundance (± standard error) by marsh user category observed in dyked and undyked coastal wetlands (significance, $p<0.05$, df =100)

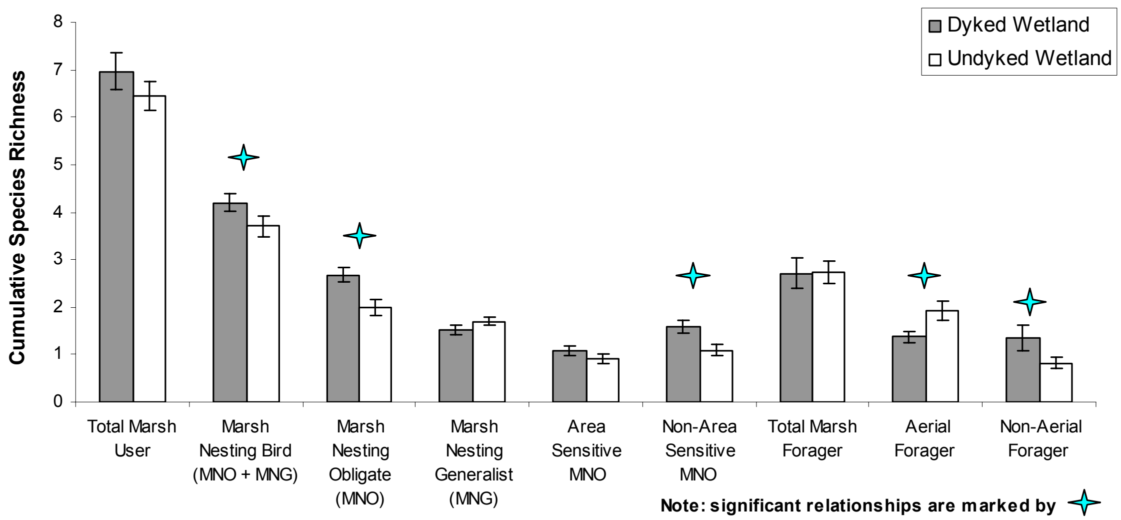


Figure 8.10 Cumulative species richness (± standard error) of birds by marsh user category observed in dyked and undyked coastal wetlands (significance, $p<0.05$, df=100)

8.2.4.6 Plant Communities Results

General summary

Six hundred and forty plant quadrats (318 in emergent marsh and 322 in open water) were surveyed across the study sites between July 12 and 27, 2004. In total, 115 plants were identified to genus, with 93 identified to species. Plants included both vascular and nonvascular species (e.g. *Chara* spp., *Nitella* spp., *Riccia* spp.). Swamp Rose Mallow (*Hibiscus moscheutos*), a species of Special Concern in Canada, was recorded in the east NWA dyke at St. Clair. The three most common plant species recorded in one metre by one metre quadrats in emergent and open water communities in the dyked and undyked wetlands are listed in Table 8.7.

The highest species richness of vascular and native plants in emergent marsh and open water communities was found in the dyked wetlands at Long Point and St. Clair NWA west dyke, respectively (Table 8.8). Conversely, the dyked wetland at Hillman Marsh had the lowest species richness of vascular and native plants in emergent and open water communities. Highest coverage of vascular plant and native species in emergent and open water communities was found at Cranberry Marsh and St. Clair NWA west dyke, respectively, while the lowest values for these species and communities were found in the St. Clair NWA east dyke and in the dyke at Hillman Marsh.

Table 8.6 Indices of abundance (Abd) and cumulative species richness (SR) of birds by marsh user category observed in dyked and undyked wetland pairs (significance, p<0.05, df =100). See Table 8.4 for bird species codes. Significant differences (p < 0.05) are indicated by "**y**". (Abd = index of abundance; SR = cumulative species richness).

Wetland	Total Birds		MNG		MNB		Area Sensitive MNO		Non-Area Sensitive MNO		MNO		Total Foragers		NAF		AF	
	Abd	SR	Abd	SR	Abd	SR	Abd	SR	Abd	SR	Abd	SR	Abd	SR	Abd	SR	Abd	SR
Amherst Island Dyke	6.56	6.11	1.44	1.00	2.67	3.56	2.67 y	1.67	1.00	0.89	2.67	2.56	2.00	2.56	1.11	1.22	1.67	1.33
Amherst Island Undyked	6.22	4.67	2.44	1.33	3.11	3.11	1.22 y	1.11	1.22	0.67	1.89	1.78	1.00	1.56	0.78	0.78	0.56	0.78
Hillman Marsh Dyke	9.88	7.38	4.88	1.88	5.00	3.25	0.88	0.88	0.75	0.50	1.13	1.38	2.75	3.75	2.38	2.25	1.50	1.50
Hillman Marsh Undyked	9.13	4.75	4.38	1.50	4.38	2.38	0.38	0.25	0.50	0.63	0.50	0.88	3.00	2.38	3.00	1.50	0.88	0.88
Long Point NWA Dyke	9.80	5.20	4.00	1.10 y	4.10	3.40	1.10	0.80	2.20	1.50	2.30	2.30	1.50	1.80	1.30	0.50	0.40	1.30
Long Point Undyked	11.20	6.70	2.60	2.10 y	3.10	4.00	1.30	0.90	2.00	1.00	2.10	1.90	6.10	2.70	5.00	0.60	1.60	2.10
Lynde Shores Dyke	21.40 y	13.20 y	4.00	2.00	5.00	6.00 y	1.00	1.00	4.00 y	3.00 y	4.00 y	4.00 y	4.40	7.20	4.00	5.20 y	2.60	2.00
Lynde Shores Undyked	7.60 y	6.20 y	2.80	1.20	2.80	1.40 y	0.20	0.20	0.00 y	0.00 y	0.20 y	0.20 y	3.20	4.80	2.60	1.40 y	18.00	3.40
St. Clair Marsh NE Dyke	10.25	6.42	4.67	1.42	4.67	4.67	1.00	1.00	2.00	2.25	2.33	3.25	1.50	1.75	1.25	0.42	0.50 y	1.33 y
St. Clair Marsh NE Undyked	10.25	8.83	4.08	2.08	4.17	4.83	0.92	0.92	1.67	1.83	1.92	2.75	3.42	4.00	2.25	0.83	2.25 y	3.17 y
St. Clair Marsh SW Dyke	13.25	6.75	4.83	1.92	4.92	4.75	1.00	1.17	2.33	1.67	2.42	2.83	1.92	2.00	1.42	0.83	0.83	1.17
St. Clair Marsh SW Undyked	7.75	6.42	2.58	1.58	2.92	4.58	1.17	1.50	2.58	1.50	2.58	3.00	1.42	1.83	0.83	0.33	0.75	1.50

Table 8.8 Species richness (SR) and percent coverage of emergent (A) and open water (B) invasive, native, and vascular plants and colonial autotrophs and total coverage (± standard error) of dead persistent plants and all emergent plants recorded in dyked and undyked coastal wetlands (significant differences are indicated by "**y**", p<0.05, df =306)

A.

Wetland	Invasive Plants			Native Plants		Vascular Plants		Colonial Autotrophs		Dead Persistent	Total Coverage
	SR	cover	rel cover	SR	cover	SR	cover	SR	cover	cover	
Amherst Island Dyke	0.44 y	13.64	0.065 y	6.72	144.60	6.92	110.92	7.20	113.68	18.40	94.60
Amherst Island Undyked	0.92 y	28.48	0.151 y	5.20	135.24	6.12	94.72	6.28	97.60	25.44	93.00
Hillman Marsh Dyke	0.12	1.08	0.011	2.68	113.64	2.24	89.60	2.24	89.60	23.60	72.40 y
Hillman Marsh Undyked	0	0	0	2.84	136.36	2.72	114.48	2.72	114.48	8.12	96.00 y
Long Point NWA Dyke	0.60	6.36 y	0.040 y	6.96 y	111.96	7.68	91.24	7.92	94.56	9.80	93.60
Long Point Undyked	0.72	30.92 y	0.153 y	4.92 y	131.80	5.48	113.92	5.72	121.92	7.80	96.40
Lynde Shores Dyke	0.32	2.00	0.009 y	5.20 y	212.60 y	5.48 y	138.96 y	6.48 y	168.48 y	14.68 y	100.00 y
Lynde Shores Undyked	0.60	18.20	0.104 y	2.96 y	110.84 y	3.48 y	74.04 y	3.56 y	74.32 y	33.72 y	80.80 y
St. Clair Marsh NE Dyke	0.57	6.26	0.047	3.71	93.57	4.14	59.20	4.23	59.80	29.66	86.57
St. Clair Marsh NE Undyked	0.44	5.12	0.036	4.48	102.20	4.80	80.32	4.84	80.36	12.24	86.40
St. Clair Marsh SW Dyke	0.18 y	0.76	0.006	3.56	139.68 y	3.74	87.03	3.82	87.29	36.77 y	94.41 y
St. Clair Marsh SW Undyked	0.63 y	5.08	0.043	4.17	97.42 y	4.42	76.21	4.50	77.29	7.17 y	79.79 y

B.

Wetland	Invasive Plants			Native Plants		Vascular Plants		Colonial Autotrophs		Filamentous Algae	Persistent Plants	Dead Persistent	Total Coverage
	SR	cover	rel cover	SR	cover	SR	cover	SR	cover	cover	SR	cover	
Amherst Island Dyke	0.48	19.32	0.118	3.40	79.68	3.88 y	53.08	4.12	54.80	12.00	0.08	0	49.08 y
Amherst Island Undyked	0.64	7.64	0.046	4.96	115.12	5.60 y	80.76	5.72	81.24	16.32	0	0	79.80 y
Hillman Marsh Dyke	0.08	0.06	0.015	1.00	17.40	1.08	15.92	1.08	15.92	0.80	0.04	0.6	16.12
Hillman Marsh Undyked	0	0	0.000	1.20	54.92	1.20	41.84	1.20	41.84	11.40	0.04	0.04	31.00
Long Point NWA Dyke	0.23	3.35	0.019	4.65 y	120.58	4.92 y	83.58	5.92 y	108.58	7.23	0.12	0	87.31
Long Point Undyked	0.04	3.80	0.019	2.56 y	113.36	2.60 y	76.72	3.28 y	100.72	0.80	0.00	0	83.20
Lynde Shores Dyke	0.12	0.64	0.012	2.68	110.56 y	2.96	79.44 y	3.36 y	88.76 y	12.00	0.28	0	84.40 y
Lynde Shores Undyked	0.37	2.41	0.009	1.56	22.85 y	1.59	20.67 y	1.63 y	20.70 y	1.52	0	0	16.41 y
St. Clair Marsh NE Dyke	0.11	0.40	0.003	3.66	121.89 y	3.77	75.20	4.72 y	117.20 y	1.46	0.06	0	94.00 y
St. Clair Marsh NE Undyked	0.12	0.72	0.005	2.56	60.24 y	2.68	42.32	3.04 y	54.52 y	4.40	0	0	53.64 y
St. Clair Marsh SW Dyke	0.12	1.09	0.013	5.32 y	132.59 y	5.44 y	97.56 y	6.03 y	121.09 y	5.62	0.09	0	93.09 y
St. Clair Marsh SW Undyked	0.08	0.60	0.010	1.48 y	36.88 y	1.56 y	32.76 y	1.64 y	33.08 y	2.68	0	0	34.08 y

Table 8.7 Most common wetland plant species recorded in one metre by one metre sampling quadrats in CCIAP studied wetlands

Dyked Wetlands		Undyked Wetlands	
Open water quadrants	**Emergent marsh quadrats**	**Open water quadrats**	**Emergent marsh quadrants**
White water lily (*Nymphaea odorata*)	Cattail (*Typha glauca, Typha angustifolia*)	Duckweed (*Lemna minor*)	Duckweed (*Lemna minor*)
Greater duckweed (*Spirodela polyrhiza*)	Duckweed (*Lemna minor, Spirodela polyrhiza*)	Slender naiad (*Najas flexilis*)	Cattail (*Typha angustifolia*)
Coontail (*Ceratopyllum demersum*)	Jewelweed (*Impatiens capensis*)	Coontail (*Ceratopyllum demersum*)	Frog's bit (*Hydrocharis morsus-ranae*)

Comparison of emergent marsh plant community between dyked and undyked wetlands

Dyked wetlands had lower species richness and relative coverage of invasive plants and higher species richness and total coverage of native plants than undyked wetlands (Figures 8.11, 8.12). Dyked wetlands also had higher coverage of dead persistent emergent plants, an important structural element in wetlands, than undyked wetlands. There were no significant differences in species richness or coverage of vascular plants, colonial autotrophs, and filamentous algae between dyked and undyked wetlands. While total vegetative coverage was slightly higher in the dyked than undyked wetlands, the difference was not significant.

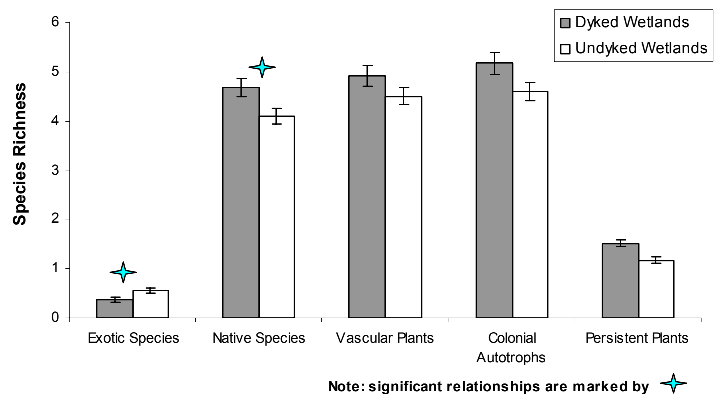


Figure 8.11 Species richness of emergent marsh vegetation (± standard error) recorded in dyked and undyked coastal wetlands (significance, $p<0.05$, df =306)

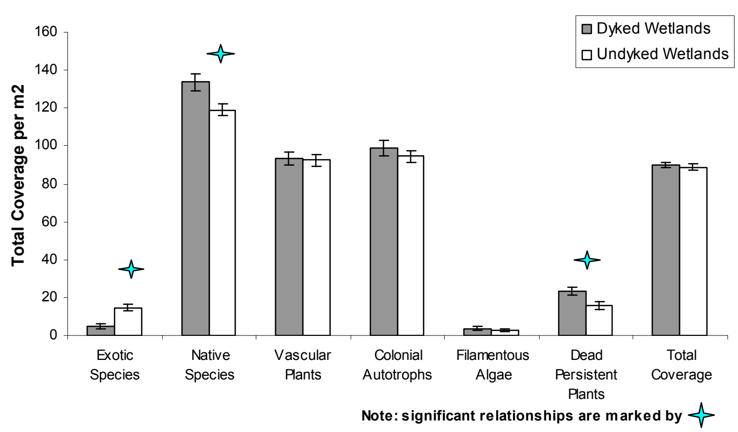


Figure 8.12 Total coverage per m^2 of emergent wetland vegetation (± standard error) recorded in dyked and undyked coastal wetlands (significance, $p<0.05$, df =306)

Individual pairs of wetlands also showed significant differences in species richness and total coverage of wetland plants. Three of six pairs (Long Point, Amherst Island, and Lynde Creek Marsh/Cranberry Marsh) had higher relative coverage of invasive emergent plants in the undyked than dyked wetlands (Table 8.8A). Long Point and Cranberry Marsh also had higher native species richness in the dyked than undyked wetlands. Most differences in vegetation communities were observed at Lynde Creek/Cranberry Marsh. The dyked wetland (Cranberry Marsh) had significantly less relative coverage of invasive species and dead persistent plants, higher coverage of native plants and colonial autotrophs, and higher species richness of native plants, vascular plants, and colonial autotrophs than the undyked wetland (Lynde Creek Marsh).

Comparison of open water plant community between dyked and undyked wetlands

There were more species and higher coverage of native plants, vascular plants, and colonial autotrophs in the open water community in dyked than in undyked wetlands (Figures 8.13, 8.14). Dyked wetlands also had significantly higher total areal coverage of vegetation in open water quadrats compared to undyked wetlands.

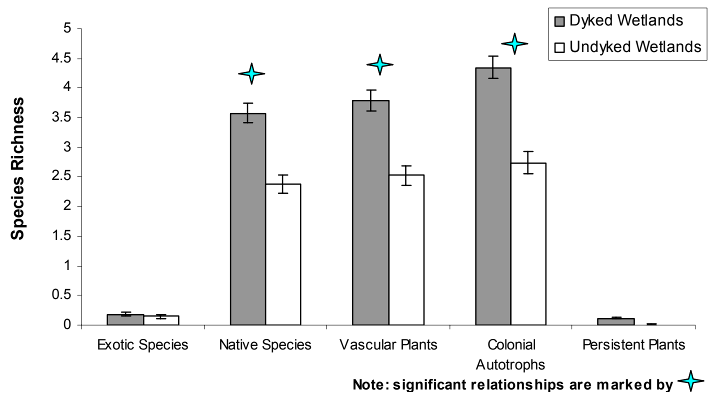


Figure 8.13 Species richness (± standard error) of open water vegetation recorded in dyked and undyked coastal wetlands (significance, p<0.05, df =306)

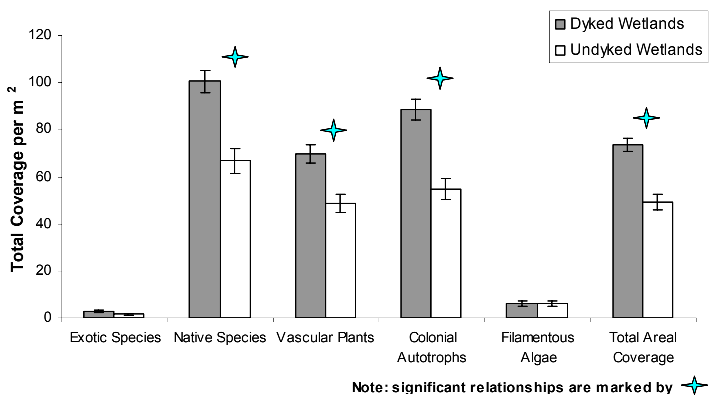


Figure 8.14 Total coverage per m^2 (± standard error) of open water vegetation recorded in dyked and undyked coastal wetlands (significance, p<0.05, df =306)

There were more differences in species richness and coverage of open water vegetation between the NWA west dyke and the undyked shoreline on Lake St. Clair than at all other paired sites (Table 8.8B). The undyked shoreline had significantly lower native, vascular, and colonial autotroph species richness and lower coverage of native plants, vascular plants, colonial autotrophs, and total areal coverage than did the dyked wetland (Figure 8.15).

8.2.4.7 Discussion

With the exception of Cranberry Marsh and Lynde Creek Marsh, few statistically significant differences existed between the paired dyked and undyked wetland bird and plant communities. This result was noteworthy. Despite drastically different historical water level regimes, the communities of emergent marsh and open water plant species in dyked and undyked wetlands have evolved in a similar fashion, thereby, supporting similar bird communities. At Lynde Shores (Lake Ontario), however, significant differences in wetland bird and plant communities were found between the dyked and undyked wetlands. In every instance where a response variable was significant, the more desirable attribute was found in the dyked wetland.

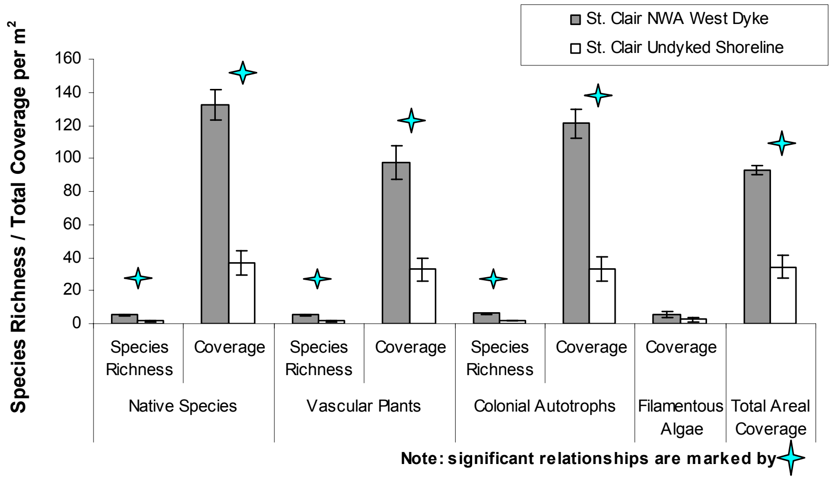


Figure 8.15 Species richness and total coverage per m^2 (± standard error) of open water vegetation recorded in the west NWA dyke and undyked shoreline on Lake St. Clair (significance, $p<0.05$, df =306)

Although additional years of data collection will be required to fully understand the driving factors behind these differences, these preliminary results suggest that, with appropriate infrastructure and management, wetland dyking may be a useful management strategy in some Great Lakes coastal wetlands. Thus, in some wetlands where rapid climate-induced water level declines might otherwise eliminate wetlands, dyking may provide an option to maintain, or increase, wetland plant and bird community diversity.

For example, dyked wetlands had significantly more dead persistent vegetation (predominantly *Typha* spp.) and higher total coverage of open water vegetation (predominantly native submerged aquatic vegetation) than did undyked wetlands (Figures 8.12, 8.14). Dead cattails provide breeding structure (i.e. territorial perches and nesting material) for birds throughout the breeding season. In addition, submerged aquatic vegetation provides an important food source both directly and indirectly (i.e. habitat for invertebrates) for birds. Thus, more breeding structure and potentially more food allow dyked wetlands to support more breeding birds compared to undyked wetlands (Figures 8.9, 8.10). Similarly, Prince (1985) reported that undyked wetlands characterized by low amounts of submerged aquatic vegetation had fewer breeding birds than did dyked wetlands. In this study, dyked wetlands also generally had greater interspersion of emergent marsh and open water habitat than did their undyked counterparts. Wetlands supporting diverse plant communities interspersed with open water provide suitable nesting habitat for more birds and bird species (Kantrud and Stewart 1984; Craig and Beal 1992). This may also explain why bird abundance and species richness in dyked wetlands tended to be greater than in undyked wetlands.

Dyked wetlands also had fewer invasive emergent plants (primarily purple loosestrife) than the undyked wetland sites. Although reasons for this result require further study, one possible explanation could be the presence of unsuitable growing/survival conditions. Although many invasive plants can germinate with water below, at, or above the soil surface, continued growth and survival may not be possible in inundated conditions (Kelsall and Leopold 2002; also see Chapter 3). Therefore, permanently inundated soils, which are commonly found within dyked wetlands, may result in fewer invasive emergent plants. However, plant seeds may still accumulate in dyked wetland soil by dispersal from nearby wetlands. For example, a study of dyked and undyked coastal wetlands in Lakes Michigan and Huron revealed that dyked wetlands had seed banks with more invasive species than did their undyked wetland pairs (Herrick 2003). Similarly, anecdotal evidence of seed banks in the dyked wetlands on the National Wildlife Area at Lake St. Clair also revealed a high abundance of invasive plant seeds (Haggeman pers. comm.). In contrast to this study, Herrick (2003) also showed a higher coverage of invasive plants in dyked than in undyked wetlands. This difference may be due to different water level management among the dyked wetlands in that study compared to this study. Generally, water level drawdowns are recommended to maximize the diversity of wetland habitat (Payne 1992). However, drawdowns may also inadvertently facilitate the germination and survival of invasive plants thus allowing their proliferation.

Overall species richness and coverage of native wetland plants was higher in dyked wetlands compared to undyked wetlands. This difference may be the result of a high native seed bank in dyked wetlands. Herrick (2003) showed that dyked wetlands had a greater richness and abundance of seeds compared to undyked sites. These differences may be due to greater topographic heterogeneity in dyked than in undyked sites and to differences in water circulation patterns. High topographic heterogeneity may provide a greater diversity of substrate conditions, thereby, providing more potential germinating and growing conditions for plants. In addition, dyked wetlands generally have a closed water circulation pattern with water input limited to runoff and/or dyke pumping. As a result, wetland plant seeds may be transported into the dyked wetland by water from runoff or pumps where they may be subsequently trapped by the physical presence of the dyke. Seeds may accumulate eventually leading to a greater diversity of wetland plants as suitable germinating/growing conditions develop. Thiet (2002) also found higher wetland species richness in dyked (59 plant species) compared to undyked wetlands (32 species) on Lake Erie These results were attributed to the dyked wetland management strategy, more heterogeneous topography in the dyked wetland, more disturbance in the dyked wetland, and selective spraying of invasive species in the dyked wetland to eliminate monotypic stands. These explanations may also account for the site-specific differences in wetland plant communities between dyked and undyked wetlands found in this study.

Site-specific analysis

Lynde Creek and Cranberry Marsh on Lake Ontario had the most differences in wetland plant and marsh bird communities between paired dyked and undyked wetlands (Tables 8.6, 8.8A, 8.8B). Cranberry Marsh (dyked) supported a significantly higher abundance and diversity of both marsh bird and plant species than Lynde Creek Marsh (undyked). These differences are likely attributable to dyke management as well as anthropogenic and natural disturbance. The Central Lake Ontario Conservation Authority manages both Lynde Creek Marsh and Cranberry Marsh. In 1999, the *Cranberry Marsh Management Zone Strategy* (Central Lake Ontario Conservation Authority 1999) was released that outlined strategies to manage unique vegetation communities (i.e. meadow marsh and mixed shallow aquatic) within that wetland. These communities were managed using water level manipulation during critical germination and growth periods. This strategy of active management appears to have been successful in establishing a diversity of wetland plants thus providing many habitats for marsh birds.

The differences in wetland plant and marsh bird communities between these wetlands cannot be solely attributed to wetland dyking and its associated water level management. Cranberry Marsh also has a very small watershed consisting primarily of undisturbed land. In contrast, the undyked Lynde Creek Marsh has a highly degraded watershed due to runoff from a much larger, urbanized area than Cranberry Marsh. In addition, common carp and brown bullhead (*Ameiurus nebulosus*) easily access Lynde Creek from Lake Ontario. High runoff, in conjunction with fish spawning and foraging activities, cause elevated turbidity and reduce the growth of submerged aquatic vegetation (Environment Canada 2004). These stressors (and likely others) in Lynde Creek Marsh have reduced the amount and quality of available habitat for birds, fishes, reptiles, and amphibians during various life history stages.

There were also several notable differences in wetland vegetation and marsh bird communities between the undyked wetland along the Lake St. Clair shoreline and the St. Clair NWA west dyke (Tables 8.6, 8.8A, 8.8B; Figure 8.15). These differences may be attributed to water level management within the dyked wetland and differences in wetland topography and disturbance. All of these factors were also cited as potential explanations for observed differences in wetland communities between dyked and undyked wetlands by Thiet (2002). For example, along the undyked Lake St. Clair shoreline, wetland vegetation has advanced lakeward with low water levels due to a very consistent, gradual sandy slope. However, periods of extensive wind, wave, and ice disturbance limit the establishment of some submerged aquatic vegetation (Doyle 2001) and in some areas, the growth of emergent vegetation. In contrast, the NWA west dyke is protected from wind and wave action and has relatively deep clear water with highly organic substrates. As a result, submerged aquatic plant communities are well developed particularly submerged plant species that tolerate deep water.

The dyked and undyked wetlands at Hillman Marsh had very few plant species and low plant coverage (Tables 8.8A, 8.8B). Moreover, the diversity of the marsh bird community was underdeveloped (Table 8.6). Again, factors suggested by Thiet (2002) and Payne (1992) may explain these trends. The dyked wetland at Hillman Marsh has not experienced a full drawdown since 1994 (Essex Region Conservation Authority 2003), a time frame much longer than the 5-year disturbance schedule recommended to maximize habitat diversity (Payne 1992). Although the 1994 drawdown resulted in an increase in vegetation coverage by almost 50%, in the following 10 years, the area of open water increased and dense stands of common reed proliferated. Consequently, species richness and coverage of vegetation communities declined resulting in reduced availability and suitability of wetland habitat for marsh birds. In the undyked wetland, common carp and agricultural runoff degraded wetland habitat quality, reducing the numbers and species of marsh birds using this wetland (Essex Region Conservation Authority 2003; Galloway pers. obs.).

The dyked and undyked wetlands on Amherst Island also had low abundance and species richness of marsh birds. This underdeveloped bird community was likely due to the characteristic vegetation stands that are particularly prevalent with the dyked wetland. Historically, the wetland consisted of a dense, monotypic stand of cattail that resulted from water level regulation of Lake Ontario in conjunction with the protection of the wetland by an extensive barrier beach. In 1997, a dyke was constructed with the intent of reducing dense stands of emergent vegetation by flooding to create wetland habitat with greater interspersion and species diversity. However, instead of flooding the cattails, the very thick organic cattail mats separated from the underlying substrate to become floating islands that appeared to be largely unaffected by water levels within the wetland. Dyke management (i.e. water pumping) appears to have had limited effect on the plant communities, and the vegetation communities between the dyked and undyked wetlands remained very similar. Likewise, the bird communities were also very similar with low species richness and abundance as a result of the persistence of these monotypic stands of vegetation.

8.2.4.8 Conclusion

There are many additional trends and associations found in the data that suggest similarities among wetland dyking, wetland plant, and marsh bird communities at individual sites and between pooled dyked and undyked wetlands. Although attempts were made to minimize the possible effects of other external factors (i.e. adjacent land uses) through site selection, the influences of other stressors and landscape factors on marsh bird and wetland plant communities at each site were not quantified as part of this study. However, these factors must also be considered before management decisions pertaining to wetland dyking are evaluated.

It must also be noted that the findings presented here are the results of a one-year study designed to assess the effects of wetland dyking on two wetland functions – vegetation diversity and bird habitat. As such the results do not capture the long-term variability in the vegetation and bird communities over high and low water level cycles and management phases. It is important to remember that despite the potential for wetland dyking to maintain specific vegetation communities and bird habitat, dyking a coastal wetland will almost certainly have significant impacts on many other wetland functions. Additional years of data collection, and expanded quantitative assessment of the functions affected by dyking must be completed prior to making explicit recommendations regarding the use of dyking as an adaptation strategy to climate change.

8.2.5 Comparison of the Fish Assemblage and Wetland Habitat between Dyked and Undyked Coastal Wetlands on the Lower Great Lakes

The objective of this sub-component was to assess whether wetland dykes could be used to maintain fish assemblages in lower Great Lake coastal wetlands by maintaining water levels in the face of climate change. This objective was achieved through a comparison of fish assemblages in paired sets of open (undyked) and closed (dyked or natural barrier) wetlands in the southern Great Lakes basin.

8.2.5.1 Study Sites

Fish assemblages were examined in six paired coastal wetlands in Lakes Ontario, Erie, and St. Clair (Figure 8.16). Each pair consisted of a closed (dyked or natural barrier) coastal wetland and an adjacent open coastal wetland (Table 8.9). Paired wetlands were relatively the same size, if possible. If not, then similarly sized wetlands were chosen as study sites. To minimize geographical and location bias, wetlands pairs that were close to each other and physically similar in relative habitat diversity were selected. The goal was to select wetlands that could be accessible to fishes from the same species pool, if all barriers were removed. Of the closed wetlands, two were completely enclosed wetland cells, two were completely enclosed but water levels were controlled with the aid of a pump, and two were connected to the lake during high water level periods. Five of the open wetlands were open to the lake year round, while one, Hillman Marsh, was only connected to the lake during high water level periods. Fluctuations in water levels played a role in the level of connectivity between some of the wetlands and the open water; therefore, a connectivity gradient was assigned (Table 8.10).

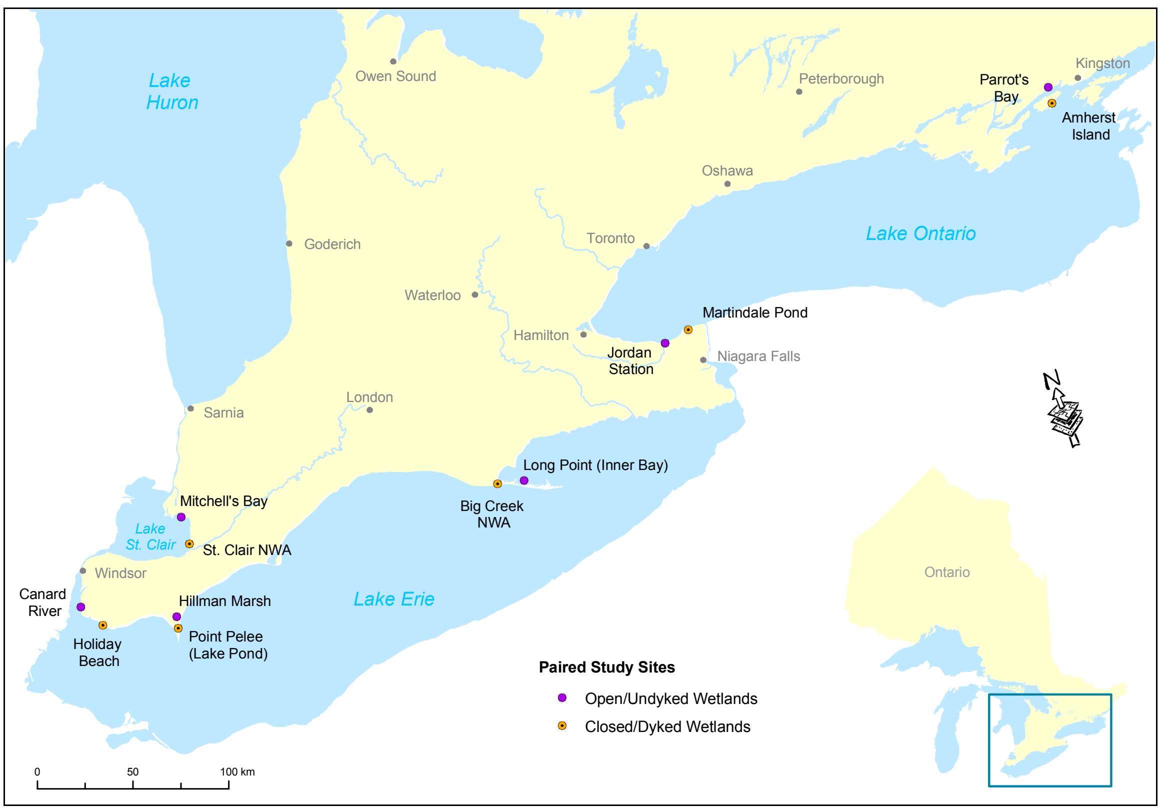


Figure 8.16 Location of the coastal wetland field sites included in the fish assessment portion of this study

This gradient was a rough estimate of the amount of time the wetland was connected to the open water annually, as well as the amount of water pumped into the wetland annually, if any. It was important to consider whether or not water was manually pumped into the wetland because it has not been established with certainty whether fish fry are able to enter the wetland unharmed through pumping (Haggeman pers. comm.; Faucher pers. comm.). The wetlands were visited in the spring and fall of 2003 and 2004. Some of the wetlands were not sampled during the four occasions because of low water levels, lack of accessibility, or mechanical difficulties (Table 8.11).

Table 8.9 Open and closed coastal wetland field sites used to compare fish assemblages

Open Wetland	Wetland Pair Abbr.	Closed Wetland	Abbr.	Associated Water Body	Time of Sampling Spring 2003	Fall 2003	Spring 2004	Fall 2004
Hillman Marsh	HM	Point Pelee (Lake Pond)	PP	Lake Erie	Jun 9-13	Aug 18-22	Jun 7-10	Sept 27-30
Long Point (Inner Bay)	LP	Big Creek Marshes	BC	Lake Erie	Jun 23-27	Sept 15	May 31-4	Sept 20-23
Mitchell's Bay	MB	St. Clair NWA	NWA	Lake St. Clair	Jul 7-11	Aug 25-29	May 10-13	Sept 13-16
Canard River	CR	Holiday Beach	HB	Lake St. Clair/ Lake Erie	Jul 14-18	Sept 8-12	May 3-6	Sept 7-10
Parrott's Bay	PB	Amherst Island	AI	Lake Ontario	Jul 21-25	Oct 6-10	Jun 14-15 Aug 11-13	Oct 12-15
Jordan Harbour	JH	Martindale Pond	MP	Lake Ontario	Jul 28-1	Oct 13-17	Jun 17-22	Oct 4-8

Table 8.10 Connectivity gradient (%) created to account for the level of connectivity between each wetland and their associated open water body. Level of connectivity was assigned in terms of annual lake accessibility by fishes.

Site	Classification	Connectivity Description	% Open	% Closed
Mitchell's Bay	open	Part of Lake St. Clair.	100	0
St. Clair NWA	closed	Always closed to Lake St. Clair. Water is pumped manually from the lake. Very unlikely that fishes could survive the pump.	0	100
Canard River	open	River flows into Lake Erie.	100	0
Holiday Beach	closed	During very high water level periods, wetland is open to Lake Erie. Occurs once a year for a very short period of time. Water is pumped manually from the lake. Fish accessibility to wetland through pump non-existent because of screen on pump.	40	60
Hillman Marsh	open	Wetland open to Lake Erie during high water level periods and closed during low water level periods.	50	50
Point Pelee	closed	Wetland closed to Lake Erie. Breaching event occurs every 10 years or so, but did not occur during the two years of sampling.	0	100
Long Point	open	Part of Lake Erie.	100	0
Big Creek Marsh	closed	Connected to Lake Erie through a series of small channels. Considerable distance from the Lake Erie to sampled wetland.	100	0
Jordan Harbour	open	Connected to Lake Ontario through passage under the QEW.	100	0
Martindale Pond	closed	Closed to Lake Ontario. Sixteen Mile Creek flows into Martindale Pond.	0	100
Parrott's Bay	open	Connected to Lake Ontario through passage under highway.	100	0
Amherst Island	closed	Closed during low water level periods. Open during high water level periods.	40	60

Table 8.11 Summary table depicting inconsistencies in the sampling protocol

	2003				2004			
	Spring		Fall		Spring		Fall	
Wetland	E-fish	Hoop	E-fish	Hoop	E-fish	Hoop	E-fish	Hoop
Mitchell's Bay	✓	✓	I	✓	✓	✓	✓	✓
St. Clair NWA	✓	✓	✓	✓	✓	✓	✓	✓
Canard River	✓	✓	✓	✓	✓	✓	✓	✓
Holiday Beach	✓	✓	✓	✓	✓	✓	✓	II
Hillman Marsh	III	✓	✓	✓	✓	✓	✓	✓
Point Pelee	III	✓	✓	✓	✓	✓	✓	✓
Long Point	✓	✓	✓	✓	✓	✓	✓	✓
Big Creek Marsh	✓	✓	IV	IV	✓	✓	✓	✓
Jordan Harbour	✓	✓	✓	✓	✓	✓	✓	✓
Martindale Pond	✓	✓	✓	✓	✓	✓	✓	✓
Parrott's Bay	✓	✓	✓	✓	V	V	✓	✓
Amherst Island	✓	✓	✓	✓	✓	✓	✓	✓

I. Two of the four quadrats could not be sampled by boat electrofishing because of mechanical problem with boat electrofishing equipment.
II. No fyke nets were set because sampling time interfered with waterfowl hunting season
III. Electrofishing was not complete because of technical problems.
IV. Sampling could not complete because water levels were too low and wetland was not accessible by boat.
V. Spring sampling could not be completed because water levels were too high and wetland was not accessible by boat. Wetland was sampled in the summer when water levels had decreased.

8.2.5.2 Methods

Fish assemblage assessment

Two methods of fish sampling, boat electrofishing and fyke netting, were used to obtain data on fish assemblage composition in the study wetlands.

Electrofishing sampling design

A transect-based sampling approach was used in the boat electrofishing protocol (Figure 8.17). Ideally, a 160,000 m² area (200 m x 200 m x 4 quadrats) was covered in each of the 12 study wetlands, consisting of four equal quadrats. In some wetlands the area of the wetland sampled was maintained although transect lengths were altered (e.g. 100 m x 400 m). In three wetlands, the sampling area was reduced to three quadrats due to the lack of navigable water. Typically, there were five transects per quadrat. The first transect was placed within the emergents (if possible) and subsequent transects were staggered at 50-m intervals moving outward parallel to the initial transect. The transects were placed to capture the vegetation gradient from emergents to floating to submergent macrophytes.

A.

B.

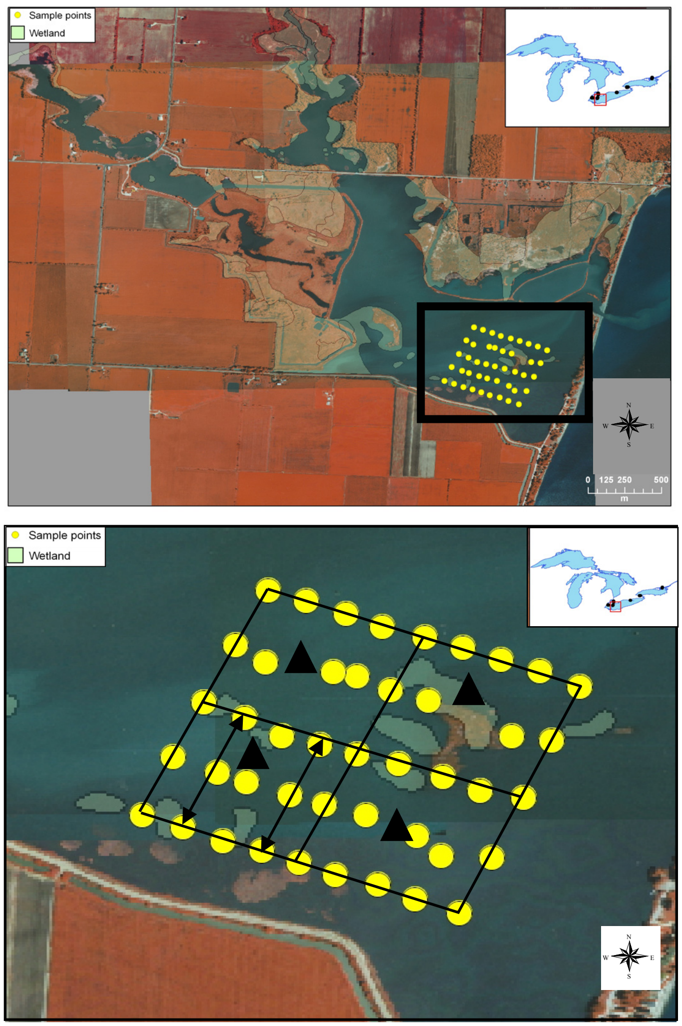


Figure 8.17 (A) An example of the wetland sampling protocol for a typical study site (Hilman Marsh, Lake Erie). (B) Each study site was composed of four quadrats; electrofishing was completed along 200 m transects. Arrows indicate the boat electrofishing path, triangles indicate fyke net positions, and yellow dots represent the points where local environmental characteristics were recorded.

Each transect was divided into three approximately equal parts that corresponded to the points where the physical habitat data were recorded. Boat electrofishing was completed along the length of the transect but fishes were kept separate based on the section in which they were caught along the transect. Voltage was adjusted to ensure a current of six amps was entering the water at all times. Attempts were made to maintain the same shocking time for each transect. To account for the level of effort of the boat electrofisher, total shock seconds for each transect were recorded. At the end of each transect, total length of each fish was recorded up to a total of 10 individuals of a single species per segment. The remaining fishes were tallied. Voucher specimens and any fish of questionable identification were preserved in 10% formalin and identified to species in the laboratory.

Fyke netting sampling design

One fyke net (mesh size = 0.64 cm, inner hoop diameter = 15 cm, wing length = 7.62 m, lead length = 15.24 m) was placed in each quadrat for approximately 24 hours. Fyke nets were retrieved and the total length of each fish was recorded up to a total of 10 individuals of a single species per fyke net. The remaining fish were tallied. Voucher specimens and any fish of questionable identification were preserved in 10% formalin and identified in the laboratory.

Habitat assessment

Physical habitat data were recorded at the beginning, mid-point, and end of each transect (Figure 8.17). In 2003, information collected included geographic coordinates using a handheld GPS unit, as well as dissolved oxygen levels (mg/L), temperature (°C), depth of water column (m), secchi depth (m), substrate composition (%), and macrophyte composition (%). In 2004, specific conductivity (ųs/cm), pH, total dissolved solids (g/L), nitrogen levels (mg/L), and turbidity (NTU) were recorded in addition to the characteristics recorded in 2003. In 2003, only the four most dominant macrophyte species were determined and given a percentage value. This was accomplished by visual inspection of a one metre by one metre quadrat. In 2004, all macrophyte species present were recorded and given either trace, subdominant, or dominant classification. These categorical classifications were converted to a representative percent coverage. In addition to visual inspection, submergent vegetation was sampled using a double-sided rake dragged across the bottom of the same one metre by one metre quadrat. The rake was thrown twice per sampling point. To verify sediment classifications, sediment samples were taken at each sample site during the fall 2004 sampling period. These samples were quantified in terms of silt, sand, and clay by sieving, and organic using a loss-on-ignition technique (Heiri *et al.* 2001).

The average of all habitat characteristics was calculated for a single sampling trip. These individual averages were then averaged to obtain an overall average (± standard error) for each sampling site over the two-year sampling period. It should be noted that standard error was not calculated for the additional habitat characteristics recorded exclusively in 2004.

Temperature loggers

One temperature logger was set in each site during the spring or early summer. Temperature loggers were placed away from boat traffic but near the lake access of the wetland (does not necessarily apply to closed wetlands). Water depth was measured and recorded, and the temperature logger was placed mid-depth in the water column. All temperature loggers were marked with site codes and geographic coordinates were recorded. The temperature loggers were programmed to record temperature at 30-minute intervals. Each temperature logger was placed in the center of a piece of plastic tubing and was secured in place with the aid of wire which passed through pre-drilled holes. The plastic tubing was then secured to an I-beam with the aid of two cable ties. Temperature loggers were retrieved in the fall of the same year. This procedure was repeated in 2003 and 2004. Unfortunately, some of the temperature loggers were missing; therefore, temperature data at these sites could not be retrieved (Table 8.12).

Table 8.12 Summary of temperature loggers that could (✓) and could not (×) be retrieved

Site	2003	2004
Mitchell's Bay	✓	×
St. Clair NWA	✓	✓
Canard River	✓	✓
Holiday Beach	✓	✓
Hillman Marsh	×	✓
Point Pelee	✓	×
Long Point	✓	✓
Big Creek Marsh	×	✓
Jordan Harbour	✓	✓
Martindale Pond	✓	×
Parrott's Bay	✓	✓
Amherst Island	✓	✓

8.2.5.3 Statistical Analysis

Fish assemblages in open and closed wetlands were compared in terms of species abundance, species richness, and species diversity. Overall species abundance was compared between open and closed wetlands. Area corrections were applied to account for the number of quadrats sampled in each wetland. Both uncorrected and corrected values were analyzed. ANOVAs and *post hoc* paired, two-sample, t-tests for means were used to test for significant differences at $p<0.05$.

Species richness between open and closed wetlands was compared across seasons (by pooling data from both years) and across years (by pooling data from both seasons). Also, overall species richness between open and closed wetlands was compared by pooling all data from all four sampling visits.

The Simpson's diversity index was used to compare species diversity. The Shannon-Weaver diversity index was considered but the data was in violation of the random sampling assumption. Because the transects were intentionally placed along the vegetation gradient, a grid-based sampling approach was used, the Simpson's diversity index was more appropriate. Seasonal, yearly, and overall species diversity averages were compared between open and closed wetlands.

8.2.5.4 Fish Assemblages Results

A total of 17,607 fishes representing 18 families and 63 species were captured during the two year study by means of boat electrofishing and fyke netting. A complete species list for each individual wetland can be found in Appendix 8.2. All fishes that could not be identified to species (0.10% of total catch), individuals identified as hybrids (0.08% of total catch) and larval fishes that could not be identified (0.18% of total catch) were removed from the dataset before analysis. The individuals that could not be identified to species included one *Notropis* spp., two *Ameiurus* spp., five *Lepomis* spp., and nine *Ictiobus* spp.

The largest number of fishes was observed during the fall 2003 sampling season with a total of 6687 individuals recorded (Table 8.13). The least number of fishes recorded was during the 2004 spring sampling season (n=2,521). The most common species was brown bullhead *(Ameiurus nebulosus*,), comprising of 18% of the overall total catch (n=3,135). The five species with the greatest pooled abundance across all sites and seasons were brown bullhead (18%), gizzard shad *(Dorosoma cepedianum)* (13%), pumpkinseed (*Lepomis gibbosus*) (12%), emerald shiner *(Notropis atherinoides*) (12%), and yellow perch (*Perca flavescens*) (10%) (Table 8.14). These five species accounted for 65% of the total catch. All other species observed individually accounted for less than 5% of the total catch.

Table 8.13 Number of fishes caught in sampling year and by sampling method

	2003		2004	
	Spring	**Fall**	**Spring**	**Fall**
Electrofishing	3631	3745	1889	2753
Fyke netting	1457	2942	632	558
Total	5088	6687	2521	3311
Grand Total (for both years)				**17607**

Six species at risk were observed during this study (Table 8.15). These species were bigmouth buffalo (*Ictiobus cyprinellus*), pugnose minnow, orange-spotted sunfish (*Lepomis humilis*), warmouth, spotted gar, and pugnose shiner. The six species at risk accounted for 0.32% of the total catch abundance.

Table 8.15 Fish species at risk recorded during this study, the percentage of total catch, and the COSEWICs status for each species

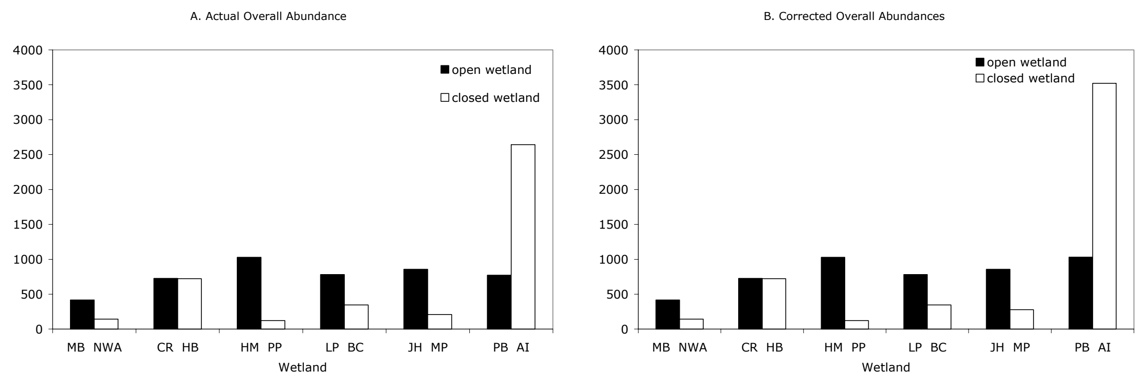
Common Name	Scientific Name	# Recorded	% of Total Catch	COSEWIC Status
bigmouth buffalo	*Ictiobus cyprinellus*	8	0.05	Special Concern
pugnose minnow	*Opsopoeodus emiliae*	1	0.01	Special Concern
orange-spotted sunfish	*Lepomis humilis*	24	0.14	Special Concern
warmouth	*Lepomis gulosus*	6	0.03	Special Concern
spotted gar	*Lepisosteus oculatus*	4	0.02	Threatened
pugnose shiner	*Notropis anogenus*	14	0.08	Endangered

Table 8.14 Abundance of fishes recorded during this study represented as actual numbers and percentage of total catch (common and scientific names according to Nelson *et al*. 2004)

Common Name	Scientific Name	Method of Capture Boat Electrofishing	Method of Capture Fyke netting	Total	% of Total Catch
alewife	*Alosa pseudoharengus*	21	33	54	0.31
rock bass	*Ambloplites rupestris*	22	89	111	0.63
black bullhead	*Ameiurus melas*	64	37	101	0.58
yellow bullhead	*Ameiurus natalis*	1	3	4	0.02
brown bullhead	*Ameiurus nebulosus*	2099	1036	3135	17.87
bowfin	*Amia calva*	20	40	60	0.34
freshwater drum	*Aplodinotus grunniens*	125	0	125	0.71
goldfish	*Carassius auratus*	713	85	798	4.55
quillback	*Carpiodes cyprinus*	11	1	12	0.07
white sucker	*Catostomus commersoni*	81	8	89	0.51
brook stickleback	*Culaea inconstans*	1	0	1	0.01
common carp	*Cyprinella spiloptera*	0	1	1	0.01
spotfin shiner	*Cyprinus carpio*	87	43	130	0.74
gizzard shad	*Dorosoma cepedianum*	2027	205	2232	12.72
lake chubsucker	*Erimyzon sucetta*	5	1	6	0.03
grass pickerel	*Esox americanus vermiculatus*	1	0	1	0.01
northern pike	*Esox lucius*	15	17	32	0.18
muskellunge	*Etheostoma exile*	1	0	1	0.01
Iowa darter	*Etheostoma nigrum*	3	0	3	0.02
johnny darter	*Esox masquinongy*	0	2	2	0.01
banded killifish	*Fundulus diaphanus*	467	153	620	3.53
channel catfish	*Ictalurus punctatus*	2	44	46	0.26
bigmouth buffalo	*Ictiobus cyprinellus*	7	1	8	0.05
brook silverside	*Labidesthes sicculus*	198	4	202	1.15
spotted gar	*Lepisosteus oculatus*	2	2	4	0.02
longnose gar	*Lepisosteus osseus*	14	40	54	0.31
green sunfish	*Lepomis cyanellus*	2	37	39	0.22
pumpkinseed	*Lepomis gibbosus*	1114	1039	2153	12.27
warmouth	*Lepomis gulosus*	2	4	6	0.03
orangespotted sunfish	*Lepomis humilis*	3	21	24	0.14
bluegill	*Lepomis macrochirus*	346	524	870	4.96
striped shiner	*Luxilus chrysocephalus*	2	1	3	0.02
smallmouth bass	*Micropterus dolomieu*	5	1	6	0.03
largemouth bass	*Micropterus salmoides*	266	144	410	2.34
spotted sucker	*Minytrema melanops*	1	1	2	0.01
white perch	*Morone americana*	32	94	126	0.72
white bass	*Morone chrysops*	81	174	255	1.45
shorthead redhorse	*Moxostoma macrolepidotum*	4	0	4	0.02
greater redhorse	*Moxostoma valenciennesi*	1	0	1	0.01
round goby	*Neogobius melanostomus*	5	22	27	0.15
river chub	*Nocomis micropogon*	0	1	1	0.01
golden shiner	*Notemigonus crysoleucas*	439	367	806	4.59
pugnose shiner	*Notropis anogenus*	12	2	14	0.08
emerald shiner	*Notropis atherinoides*	1467	569	2036	11.61
ghost shiner	*Notropis buchanani*	6	0	6	0.03
blackchin shiner	*Notropis heterodon*	60	5	65	0.37
blacknose shiner	*Notropis heterolepis*	5	0	5	0.03
spottail shiner	*Notropis hudsonius*	98	40	138	0.79
mimic shiner	*Notropis volucellus*	12	0	12	0.07
tadpole madtom	*Noturus gyrinus*	0	10	10	0.06
Chinook salmon	*Oncorhynchus tshawytscha*	1	0	1	0.01
pugnose minnow	*Opsopoeodus emiliae*	0	1	1	0.01
yellow perch	*Perca flavescens*	1338	340	1678	9.56
logperch	*Percina caprodes*	16	0	16	0.09
troutperch	*Percopsis omiscomaycus*	0	2	2	0.01
bluntnose minnow	*Pimephales notatus*	62	40	102	0.58
fathead minnow	*Pimephales promelas*	466	83	549	3.13
white crappie	*Pomoxis annularis*	3	40	43	0.25
black crappie	*Pomoxis nigromaculatus*	8	161	169	0.96
brown trout	*Salmo trutta*	0	2	2	0.01
walleye	*Sander vitreus*	8	1	9	0.05
rudd	*Scardinius erythrophthalmus*	1	3	4	0.02
central mudminnow	*Umbra limi*	114	3	117	0.67

Comparison of fish assemblages between open and closed wetlands

Overall fish abundance was compared between open and closed wetlands (Figure 8.18A). In five of the six wetland pairs, the open wetland had greater species abundance than the paired closed wetland. Although Canard River and Holiday Beach appear to have equal abundances, Canard River had three individuals more than Holiday Beach. Amherst Island at the eastern end of Lake Ontario was the single wetland with the greatest overall fish abundance. In three of the wetlands (Martindale Pond, Parrott's Bay, and Amherst Island) only three quadrats, rather than four, were sampled because of limitations of navigable water. Area-corrected abundances (per quadrat) were compared across wetlands pairs (Figure 8.18B). The same trend persisted when the corrected abundance values were compared.

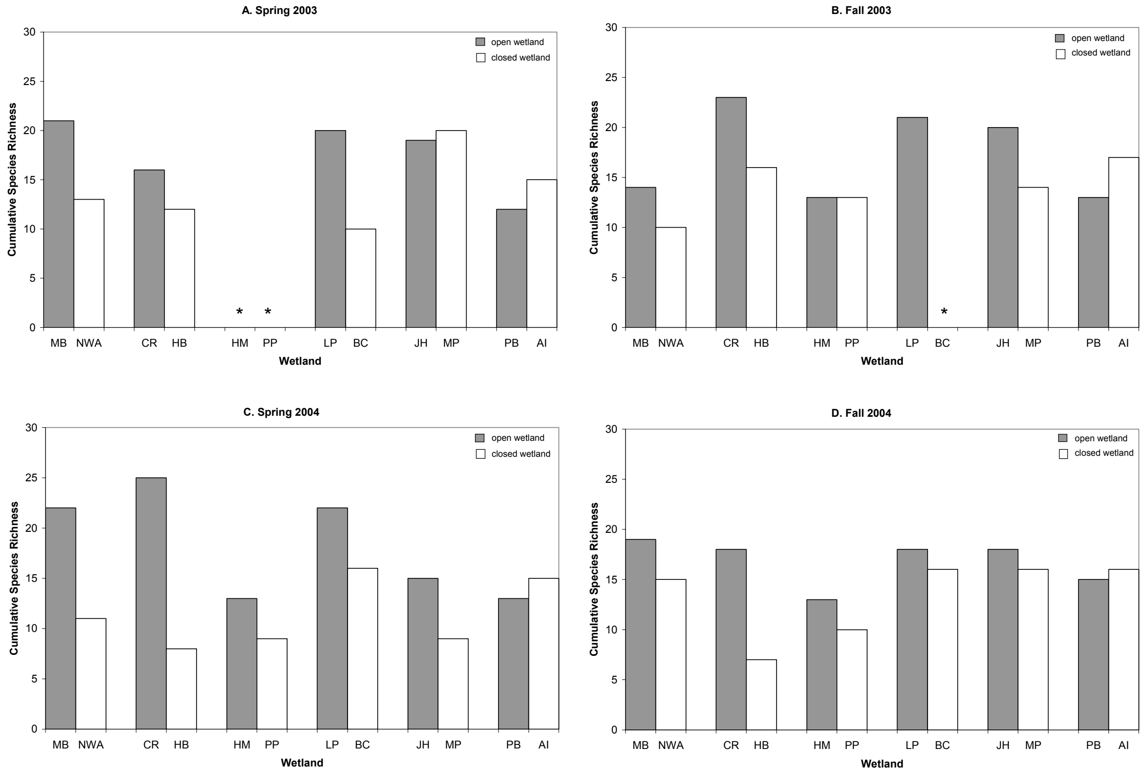


MB - Mitchell's Bay
NWA - St. Clair NWA
CR - Canard River
HB - Holiday Beach
HM - Hillman Marsh
PP - Point Pelee (Lake Pond)
LP - Long Point (Inner Bay)
BC - Big Creek Marshes
JH - Jordan Harbour
MP - Martindale Pond
PB - Parrott's Bay
AI - Amherst Island

Figure 8.18 Actual (A) and corrected per quadrat (B) overall fish species abundance values for open and closed wetlands. Data was pooled from all four sampling visits (i.e. spring and fall 2003 and spring and fall 2004).

Cumulative species richness was compared between open and closed wetlands for each sampling period (Figure 8.19). During the spring 2003 sampling period, the open wetland in three of the five wetland pairs (HM-PP pair is excluded because of sampling inconsistency) had greater cumulative species richness than the closed wetland (Figure 8.19A). The two wetland pairs sampled from Lake Ontario (JH-MP and PB-AI) had greater cumulative species richness in the closed wetland. This trend persisted during the fall 2003 sampling period (Figure 8.19B), except for Jordan Harbour (JH) and Martindale Pond (MP), in which Jordan Harbour had greater cumulative species richness.

In 2004, in five of the six wetland pairs, the open wetland had greater cumulative species richness than the associated closed wetland. For this sampling period only the PB-AI wetland pair showed a greater cumulative species richness for the closed wetland. The same trend was noted in both the spring and fall 2004 sampling periods (Figures 8.19C, 8.19D). Species lists were pooled across all sampling periods to obtain an overall species list for each wetland (Figure 8.20). When comparing the spring average species diversity between open and closed wetlands in four of the six wetland pairs, the closed wetland had a greater species diversity index (Figure 8.21A). The fall average species diversity showed somewhat similar results with the closed wetland in three of the six wetland pairs having greater species diversity (Figure 8.21A). These ratios were also present when yearly averages were calculated (Figure 8.21B). In 2003, half of the closed wetlands had greater species diversity than their open counterparts and in 2004, four of the six closed wetlands had greater species diversity. Results were similar when the data from all four sampling seasons were pooled (Figure 8.22). Although only one closed wetland (MP) showed significantly greater ($p<0.05$) overall species diversity than its corresponding open wetland, four of the six closed wetlands had greater overall species diversity than their paired open wetland.



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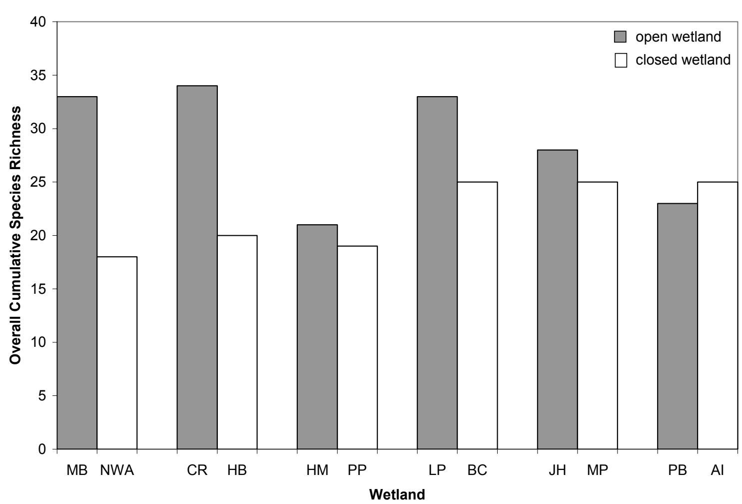
MP - Martindale Pond
NWA - St. Clair NWA
HM - Hillman Marsh
BC - Big Creek Marshes
PB - Parrott's Bay
CR - Canard River
PP - Point Pelee (Lake Pond)
JH - Jordan Harbour
AI - Amherst Island

Figure 8.19 Cumulative fish species richness for open and closed wetlands for all four sampling periods; spring 2003 (A), fall 2003 (B), spring 2004 (C) and fall 2004 (D); *represents a site that was not sampled



MB - Mitchell's Bay
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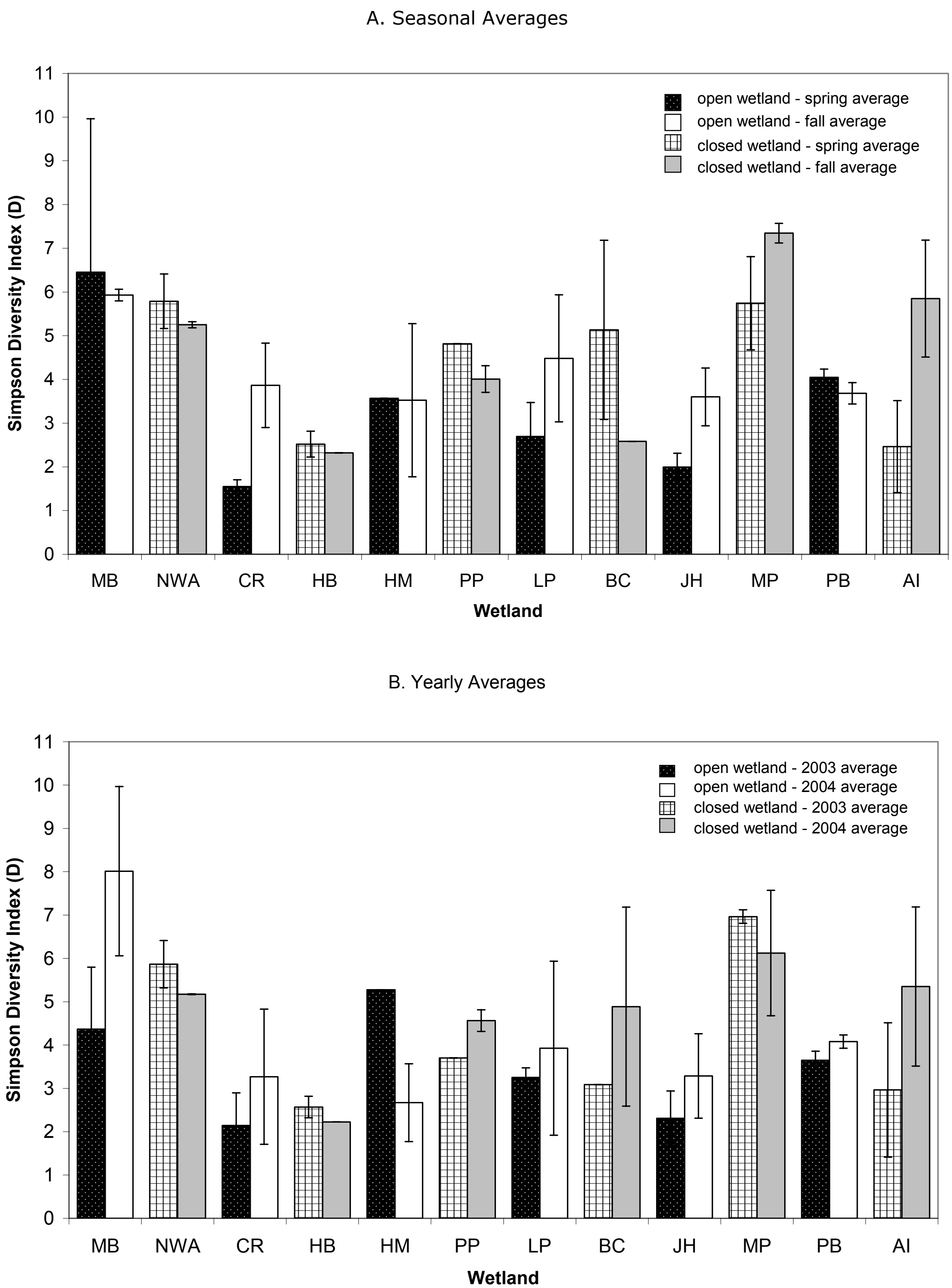
BC - Big Creek Marshes
JH - Jordan Harbour
MP - Martindale Pond
PB - Parrott's Bay
AI - Amherst Island

Figure 8.20 Overall cumulative fish species richness for open and closed wetlands



MB - Mitchell's Bay
HB - Holiday Beach
LP - Long Point (Inner Bay)
MP - Martindale Pond
NWA - St. Clair NWA
HM - Hillman Marsh
BC - Big Creek Marshes
PB - Parrott's Bay
CR - Canard River
PP - Point Pelee (Lake Pond)
JH - Jordan Harbour
AI - Amherst Island

Figure 8.21 Simpson diversity index values (± standard error) for fish assemblages seasonal averages (A) and yearly averages (B)

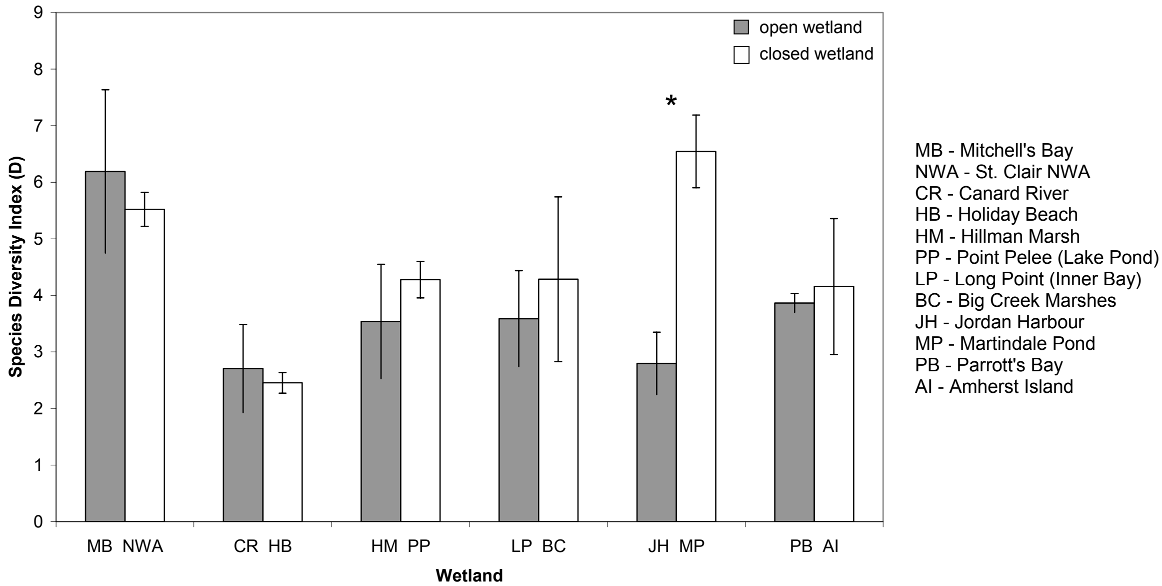


Figure 8.22 Overall Simpson's diversity index (± standard error) values obtained by averaging values from all sampling visits ± standard error; * represents a significant difference between open and closed wetland

Habitat comparison

Habitat characteristics were not statistically tested but patterns were observed and noted here between open and closed systems. In four of the six wetland pairs, the closed wetland had a greater average water depth than its open pair (Figure 8.23). The site with the greatest average depth was Point Pelee (closed wetland) with an average depth of 1.30 m. Conversely, Big Creek Marsh had the lowest average depth of 0.40 m. Average secchi depth values followed the same pattern as the water depth values with the exception of one wetland pair (Figure 8.23). Although Amherst Island had a greater water depth (0.64 m) than its pair Parrott's Bay (0.60 m), Parrott's Bay had a greater secchi depth (0.55 m, Amherst Island 0.48 m).

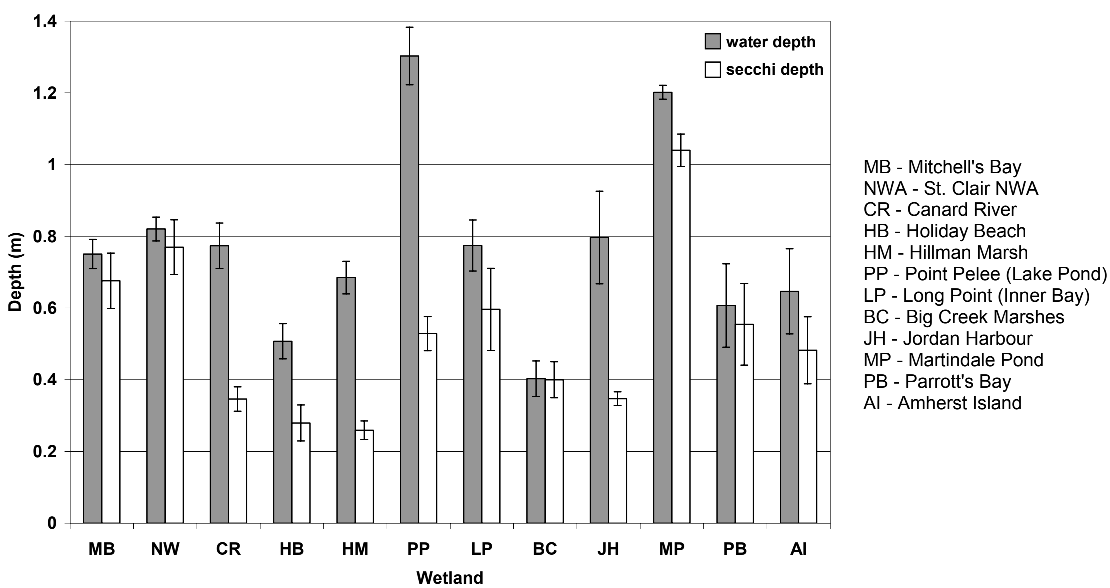


Figure 8.23 Average water depth (m) and secchi depth (m) (± standard error) for all the wetlands included in this study

The same pattern was observed when average temperature, turbidity, and pH values were compared across open and closed wetlands (Figures 8.24A, 8.24B, 8.24C). These three habitat characteristics were higher in the open wetland than the closed wetland in five of the six wetland pairs. The only closed wetland with higher average temperature, turbidity, and pH than its open pair was Big Creek Marsh. The same ratio was observed for average nitrogen values, although the one closed wetland with higher average nitrogen value was Holiday Beach (Figure 8.24D).

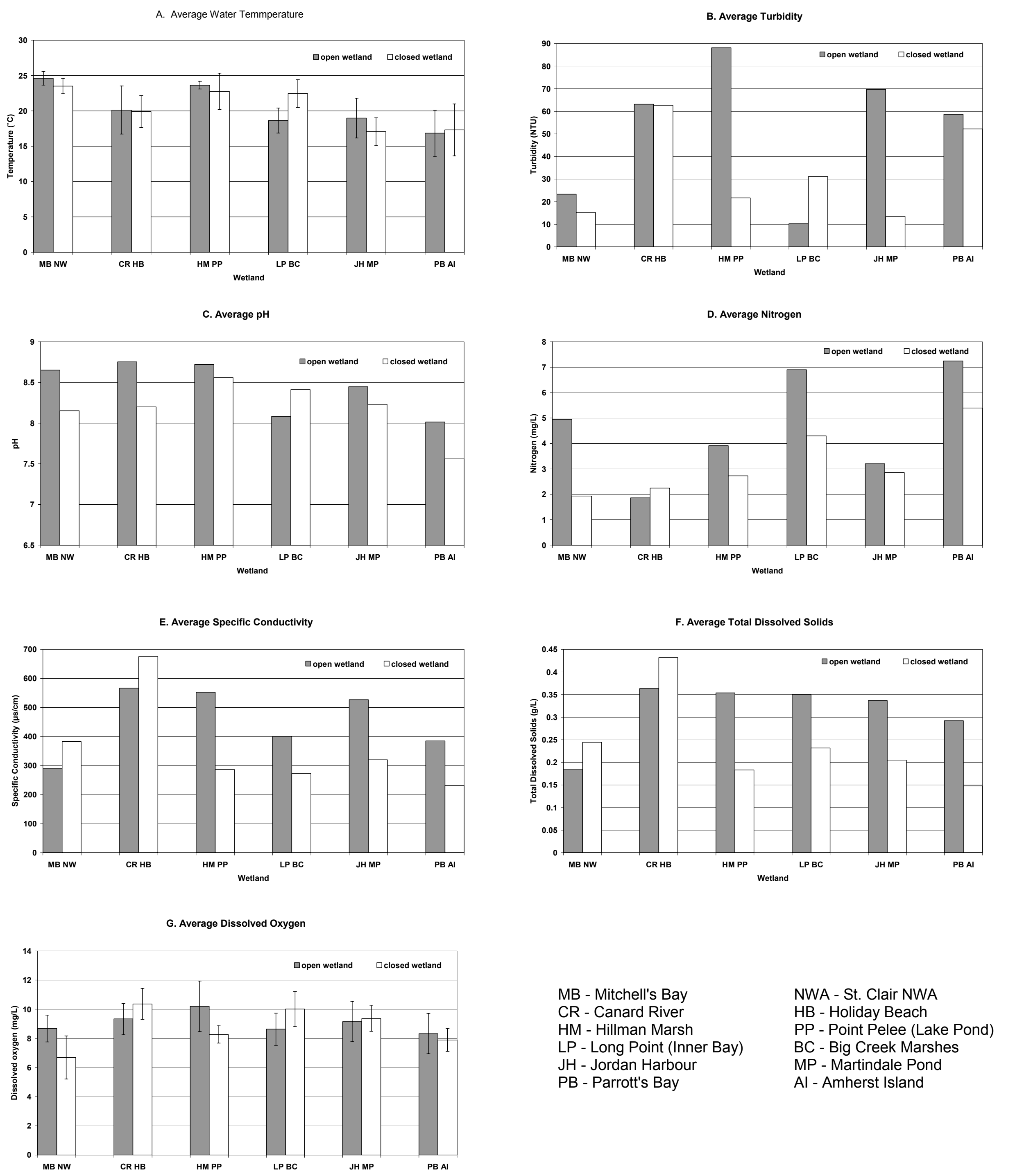


Figure 8.24 Average water temperature (A), turbidity (B), pH (C), nitrogen concentration (D), specific conductivity (E), total dissolved solids (F) and dissolved oxygen (G) for all wetlands in this study

Average specific conductivity and average turbidity values, which are highly related, also held the same pattern (Figures 8.24E, 8.24F). For both of these habitat characteristics four of the six open wetlands had higher values than their closed pair. The two wetland pairs that demonstrated the reverse trend were Mitchell's Bay - St. Clair NWA and Canard River - Holiday Beach.

No differences between open and closed wetlands were observed is dissolved oxygen (Figure 8.24G). The range for this habitat characteristic was also very small. The values ranged from 6.70 mg/L (St. Clair NWA) to 10.36 mg/L (Holiday Beach).

Sediment comparison

The average percent sediment composition was calculated from qualitative estimates recorded in the field (Figure 8.25). Based on visual inspection of the data, three of the wetland pairs (Mitchell's Bay - St. Clair NWA, Canard River - Holiday Beach, Long Point - Big Creek Marshes) had very dissimilar sediment composition, while the remaining three wetland pairs had somewhat similar composition. Hillman Marsh and Point Pelee had similar levels of organic content and sand, but there was almost a complete lack of clay recorded in the sediment at Point Pelee (0.16%). Jordan Harbour and Martindale Pond also had similar sediment composition although Martindale Pond had greater than four times the amount of sand recorded in Jordan Harbour. The third wetland pair with similar sediment composition was Parrott's Bay and Amherst Island. The difference between this wetland pair was that the open wetland (Parrott's Bay) had slightly more silt, while the closed wetland (Amherst Island) had a greater amount of clay.

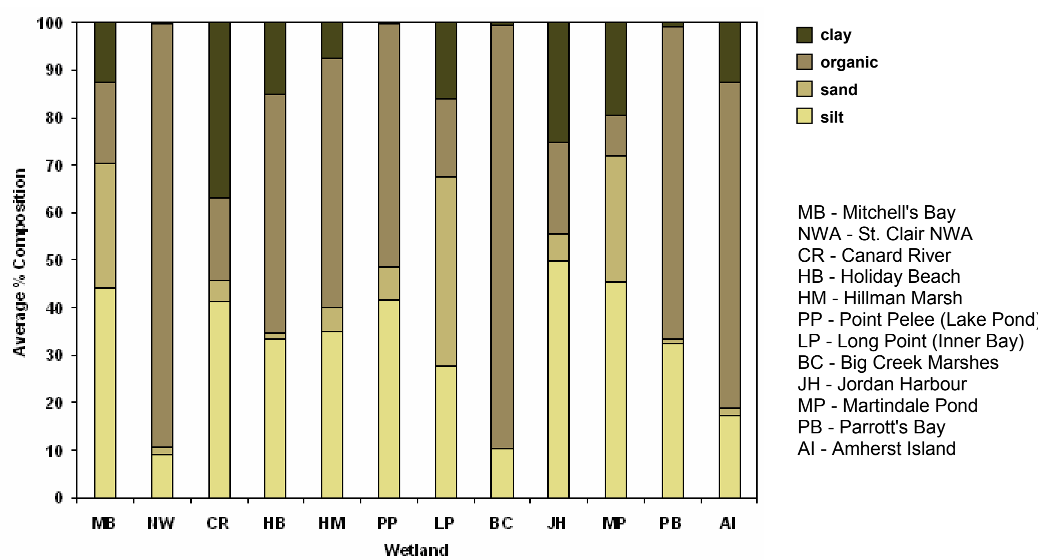


Figure 8.25 Average percent sediment composition for wetlands included in this study

Macrophyte comparison

A much clearer trend was observed when the average percent macrophyte coverage was compared between open and closed wetland pairs (Figure 8.26). There was similarity within pairs in the percent composition of macrophytes. Three of the wetland pairs (Canard River - Holiday Beach, Hillman Marsh - Point Pelee, Jordan Harbour - Martindale Pond), both open and closed wetlands, were dominated by open water. One of the wetland pairs (Parrott's Bay - Amherst Island) was dominated by floating macrophytes, while one pair (Long Point - Big Creek Marsh) was dominated by submergent macrophytes. None of the wetland pairs were dominated by emergent macrophytes. This would be expected since all surveys were completed from a boat and therefore navigable water was a necessity.

Temperature comparison

The temperature logger data enabled temperature regime comparisons between paired open and closed wetlands. In 2003, data were available for comparison from Mitchell's Bay - St. Clair NWA, Canard River - Holiday Beach, Jordan Harbour - Martindale Pond, and Parrott's Bay - Amherst Island (Figures

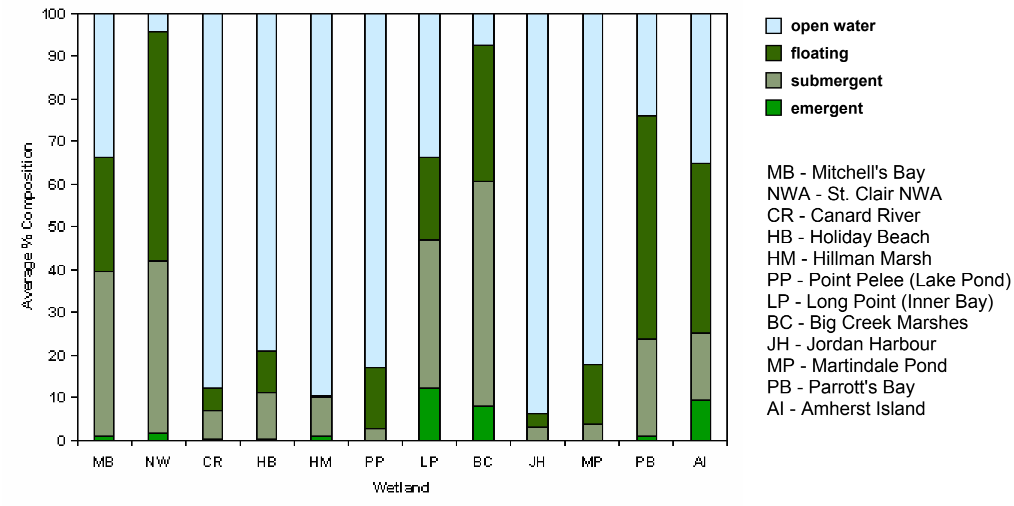


Figure 8.26 Average percent macrophyte composition for all wetlands included in this study

8.27A, 8.27B, 8.27C, 8.27E). In 2004, temperature data were available for three wetland pairs including Canard River - Holiday Beach, Parrott's Bay - Amherst Island and Long Point - Big Creek Marsh (Figures 8.27D, 8.27F, 8.27G). Further statistical analysis will be required to investigate the trends present in the temperature time series data.

8.2.5.5 Discussion

Overall, there were very few statistically significant differences between the fish assemblages observed in open and closed wetlands. In all but one wetland pair, the open wetland had greater overall fish abundance than the closed wetland. However, Amherst Island (closed wetland) had more than three times the abundance of any other wetland. One explanation for the high abundance of fishes in Amherst Island may be its connectivity with Lake Ontario. The wetland on Amherst Island was connected to Lake Ontario during high water level periods allowing fishes to use the wetland as spawning and nursery grounds. When water levels declined this connection was lost and the YOY were trapped within the wetland until water levels increased again. If the sampling visit corresponded with a low water level period, it would explain the high abundance of YOY observed and higher overall abundance.

These fluctuating water levels may also explain the increased species richness observed at the Amherst Island study wetland. In all other wetland pairs, the open wetland yielded higher overall species richness. Closed wetlands with uncontrolled water levels often have monotype emergent stands (Wilcox *et al*. 2003). Less macrophyte species diversity may result in decreased fish species richness because of decreased habitat complexity and heterogeneity (Eklov 1997). A greater number of fish species may be able to thrive in open wetlands because of increased macrophyte diversity and connectivity to the lake.

Although species richness in general was greater in open wetlands, the opposite was true when species diversity was considered. When species diversity was averaged across all four sampling visits, a greater number of closed wetlands had higher species diversity values than their corresponding open wetland. This was mainly due to the more uniform distribution of abundances across species in closed wetlands than a skewed distribution in open wetlands where a few species dominate. Although this was noteworthy, only one of the closed wetlands (Martindale Pond) had significantly higher species diversity. Therefore, dyking or closed systems may be beneficial to species diversity although richness and abundance may decrease (Figure 8.22). The effect of lake connectivity may be confounded by upstream connectivity, which for some of these systems creates a much larger functional area for species movements than is implied by dyking on the lakeside.

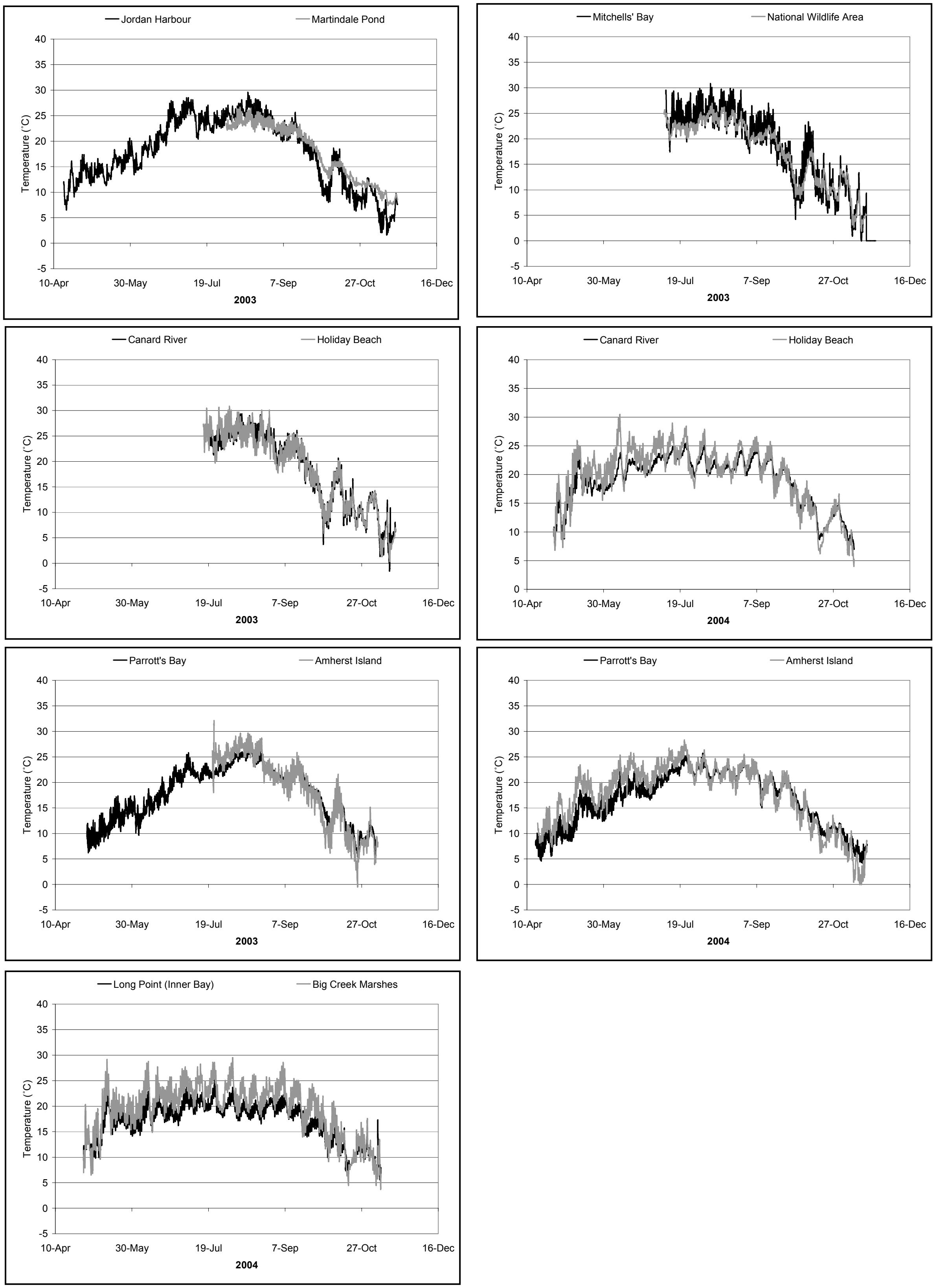


Figure 8.27 Temperature comparisons between paired sets of open and closed wetlands

Potential bias

Results from this study were based on a maximum of four sampling visits per study wetland over a two-year period, which introduced sources of bias. The first consideration should be the frequency of sampling. All fish sampling was conducted during a one-day period. This can only provide a snapshot of the fish assemblages present in the wetland. Also the time of year that the wetland was sampled was not consistent. An attempt was made to visit each wetland in the spring and the fall of each year, but, due to uncontrollable factors some sampling trips were postponed till the summer. This may have affected the number and species of fishes recorded. A third potential source of error would be the electrofishing experience of the person netting the fishes. An experienced netter will undoubtedly be able to capture a greater number of fishes, yielding greater fish abundance and perhaps greater fish species richness. Additional sampling should be completed within, as well as between, years to obtain supplemental information supporting the trends noted in this study and in the literature, which usually involve short-term studies as well, with a few exceptions.

A final thought

Numerous questions must be considered on a site-by-site basis before wetland dyking can be considered a suitable adaptation strategy to maintain fish habitat in coastal wetlands. Relating directly to climate change, it is predicted that water levels will decrease, but at what rate? Would the loss of some wetlands through climate change equal the number of new wetlands being created by the same process? If it was decided that a wetland should be dyked to maintain fish habitat, should an attempt be made to preserve the wetland in its pre-dyking condition? Finally, further studies on the use of fishways or fish ladders should be investigated as a possible adaptation strategy.

8.2.6 Modelling of the Wetland Vegetation and Bird Communities in Dyked and Undyked Coastal Wetlands

Another component of assessing the value of wetland dyking as an adaptation strategy requires an understanding of what the dyked wetland community may look like under climate change if the area was not dyked (i.e. water level manipulated independent of the lake). To complete this assessment, wetland vegetation and bird models were run using elevation models created for both the dyked and undyked wetland study sites. Models and outputs follow those detailed in Sections 4.3 and 5.3. In addition, the warm & dry and warm & wet climate change scenarios, and a high (1978) and low (1965) water years were used for the assessment of the wetland study sites. Site-specific vegetation and bird model results were pooled by lake basin.

8.2.6.1 Results and Discussion for the Predictive Modelling of Wetland Vegetation and Bird Communities

Lake Ontario

Wetland model results under the climate change scenarios predicted that vegetation in the dyked study site wetlands would consist primarily of treed/shrub vegetation if the dyke and water pumping infrastructure did not exist (Figure 8.28A). The occurrence of bird communities were also predicted to shift with a decrease in marsh nesting birds and a large increase in the abundance of treed/shrub nesters (Figures 8.29A, 8.30A).

There were some limitations in the elevation data collected at some dyked wetland locations, especially at the Amherst Island wetland, that affected these modelling results. The emergent vegetation in the dyked wetland on Amherst Island consisted of a dense and extensive floating mat of cattail with some meadow marsh habitat. The floating mats prevented a good estimation of the true basin profile, and a higher average elevation was ascribed to the wetland for modelling. The dyked wetland on Amherst Island contributed a large proportion of the total area of dyked wetland habitat on Lake Ontario, and the higher elevation model also likely resulted in an over-estimate of the meadow marsh and treed/shrub vegetation and bird communities pooled modelling results.

With the projected lower water levels on Lake Ontario under the climate change scenarios (-0.5 to -0.75 m decrease), the undyked wetland vegetation and bird modelling results also predicted a shift from emergent vegetation to meadow and treed/shrub habitats (Figures 8.28B, 8.29B, 8.30B).

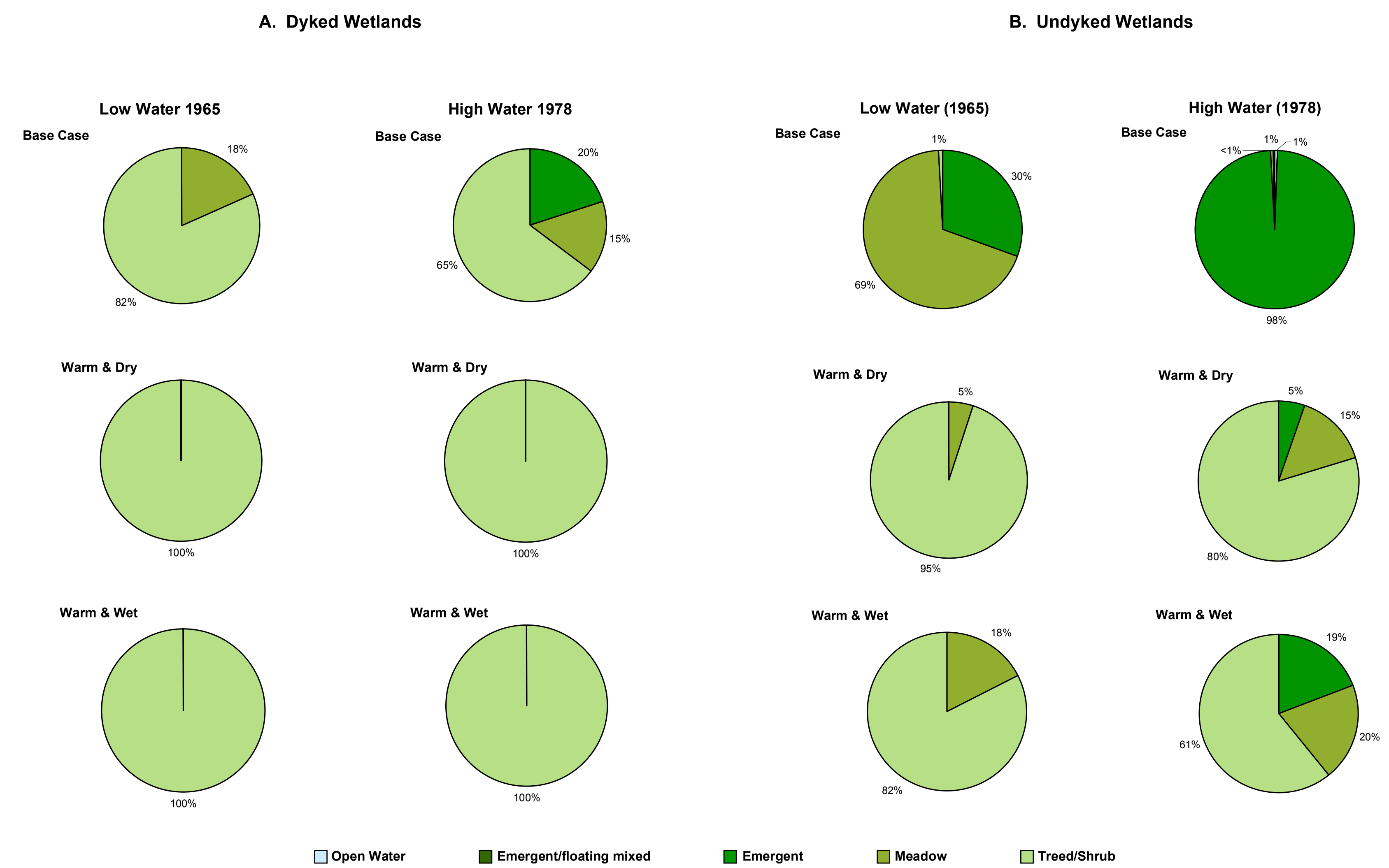


Figure 8.28 Predicted changes in wetland vegetation under low and high water cycles for the base case, warm & dry, and warm & wet climate change scenarios in dyked (A) and undyked (B) wetland study sites, Lake Ontario

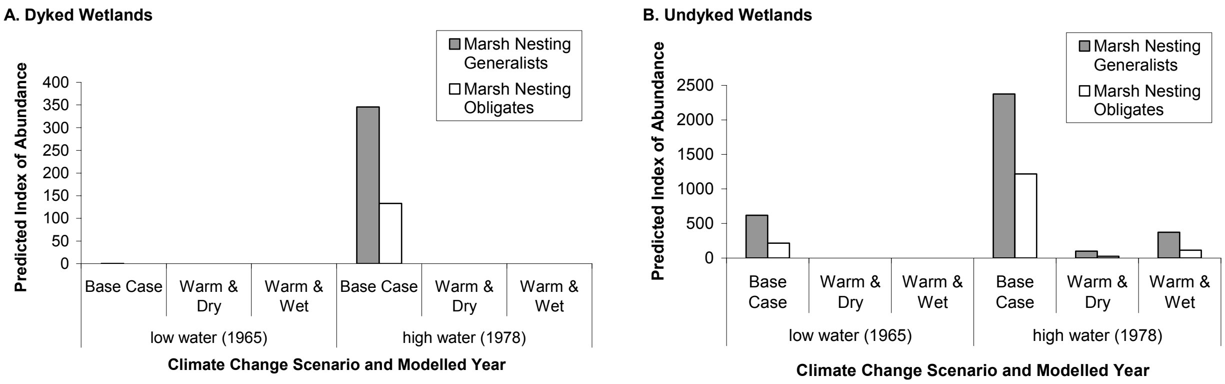


Figure 8.29 Predicted changes in indices of the abundance for marsh nesting birds under low and high water cycles for the base case, warm & dry, and warm & wet climate change scenarios in dyked (A) and undyked (B) wetland study sites, Lake Ontario

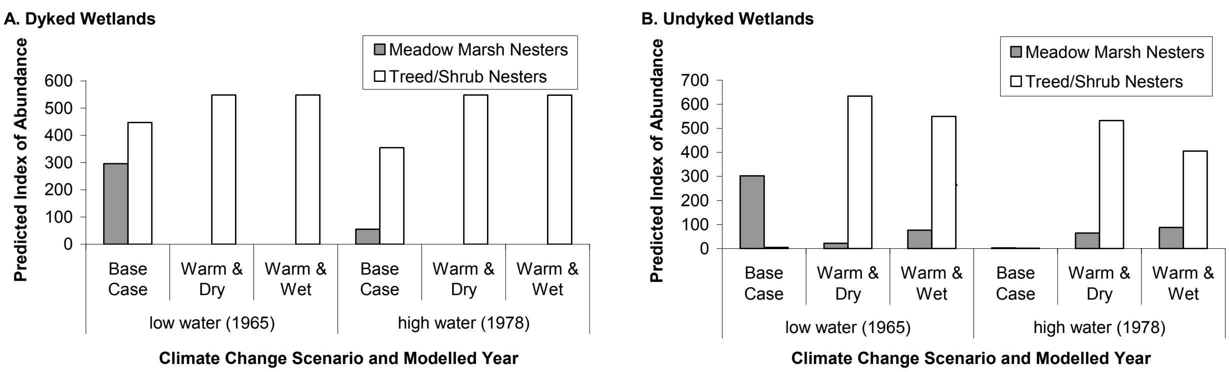


Figure 8.30 Predicted changes in indices of the abundance for meadow marsh and treed/shrub nesting birds under low and high water cycles for the base case, warm & dry, and warm & wet climate change scenarios in dyked (A) and undyked (B) wetland study sites, Lake Ontario

Lake Erie

Vegetation modelling of the dyked wetland study sites on Lake Erie also suggested a reduction in emergent marsh habitat, and a shift to treed/shrub habitat under the climate change scenarios if the dyke did not exist (Figure 8.31A). The predicted area of open water habitat did not change significantly under the climate change scenarios due to the presence of low elevation areas in the wetland basin profile of the dyked wetlands. Predicted water depths within these areas were sufficient to maintain open water habitat even with the projected drop in water level due to climate change. Some of the low elevation areas were likely created during construction of the dyke and nesting islands, which required dredging within the wetland. Under the climate change scenarios, these areas were typically where open water was predicted to occur.

When comparing between predicted vegetation response in the low and high initial water level scenarios, it was the base case scenarios that showed the greatest differences. Emergent marsh was a significant component of the wetland in the high water, base case scenario only (40%) (Figure 8.31A). In all other scenarios, emergent marsh accounts for < 20% of the modelled area. The base case, low water scenario was also unique due to the prominence of meadow marsh that was predicted (29%). These shifts indicated a very flat basin profile in which hydrologic thresholds for the emergent and meadow marsh communities were being exceeded within the various water level and climate change scenarios. This change was also evident when comparing the response of the marsh bird communities to the low and high water conditions during the base case and each climate change scenario (Figure 8.32A). Generally, the predicted index of abundance for the marsh bird communities became more stabilized with changing water levels under climate change in comparison to the base case.

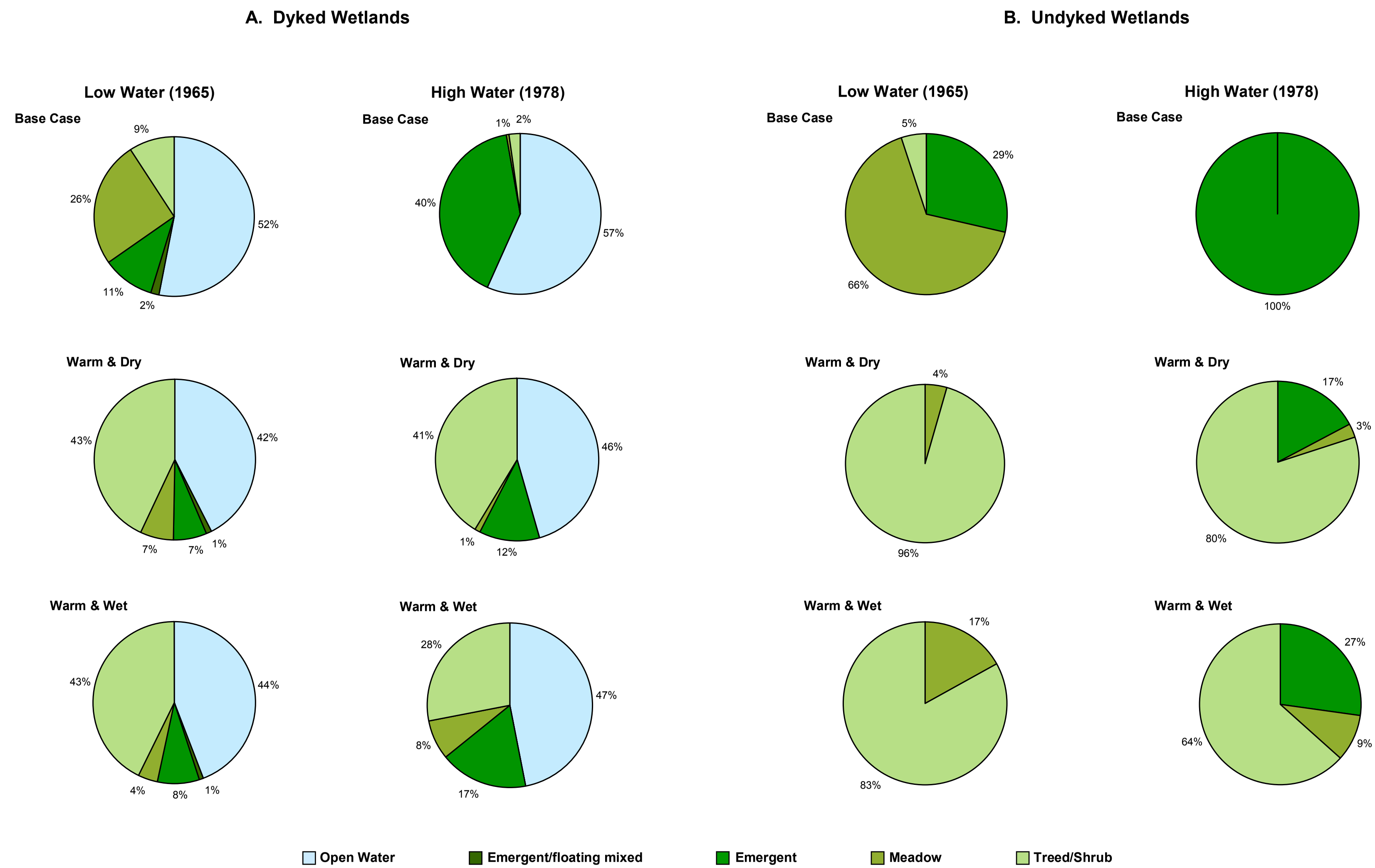


Figure 8.31 Predicted changes in wetland vegetation under low and high water cycles for the base case, warm & dry, and warm & wet climate change scenarios in dyked (A) and undyked (B) wetland study sites, Lake Erie

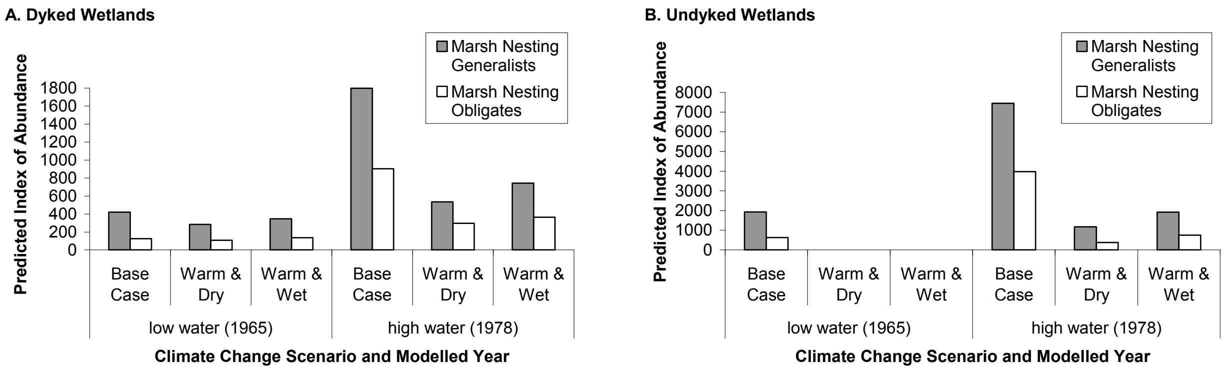


Figure 8.32 Predicted changes in indices of the abundance for marsh nesting birds under low and high water cycles for the base case, warm & dry, and warm & wet climate change scenarios in dyked (A) and undyked (B) wetland study sites, Lake Erie

Similar to the undyked wetlands on Lake Ontario, the area of emergent vegetation habitat in undyked wetlands study sites on Lake Erie were also reduced under the climate change scenarios in comparison to the base case (Figure 8.31B). Under base case scenarios, there was a large shift in wetland habitat between emergent vegetation and meadow marsh habitats between the low and high water conditions. As a result, marsh nesting and meadow marsh bird species showed a strong positive response in their abundance to an increase in the availability of associated breeding habitat (Figures 8.32B, 8.33B). As both the emergent and meadow marsh hydrologic thresholds were exceeded within the basin model under the climate change scenarios, habitats shifted to treed/shrub vegetation and bird communities (Figures 8.31B, 8.33B).

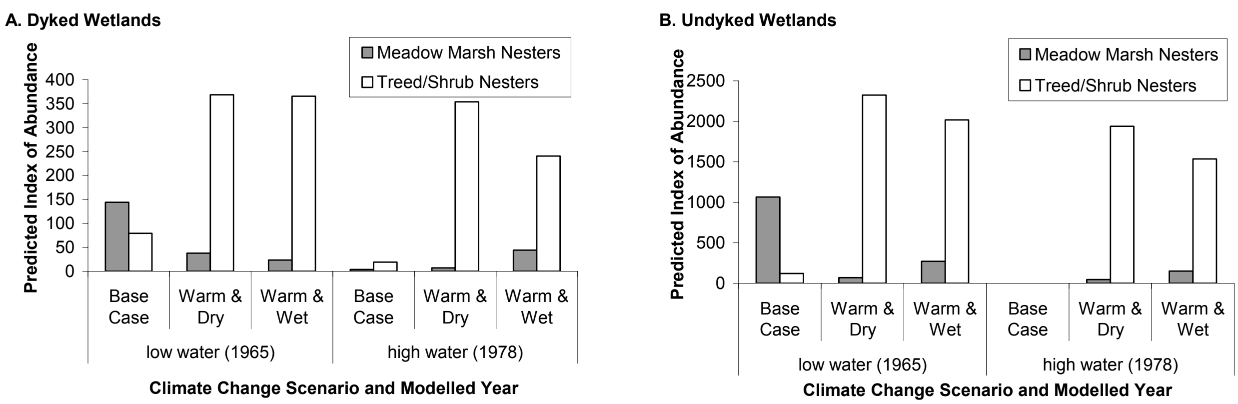


Figure 8.33 Predicted changes in indices of the abundance for meadow marsh and treed/shrub nesting birds under low and high water cycles for the base case, warm & dry, and warm & wet climate change scenarios in dyked (A) and undyked (B) wetland study sites, Lake Erie

Lake St. Clair

The Lake St. Clair dyked wetland study sites currently support almost exclusively emergent marsh and open water habitats. Based upon the vegetation and bird modelling results, it was predicted that the area of emergent vegetation and meadow marsh habitat, and associated bird communities in the Lake St. Clair dyked wetland study sites would be significantly reduced under all water level and climate change scenarios if the dyke and its pumping infrastructure did not exist (Figures 8.34A, 8.35A). The predicted shift in habitat under an undyked scenario was very similar to predictions on Lake Ontario dyked wetlands study sites.

Under the base case scenario, the undyked wetland study sites on Lake St. Clair were predicted to have a diverse community of wetland vegetation that was significantly affected by water level fluctuations (Figure 8.34B). These results also indicated that a gradual sloping basin profile allowed for the expansion or retreat of wetland vegetation as water levels changed in the high and low water level scenarios. A parallel fluctuation was observed in the predicted index of abundance for the marsh nesting generalists and obligates bird communities in response to the changing area of emergent vegetation (Figure 8.35B).

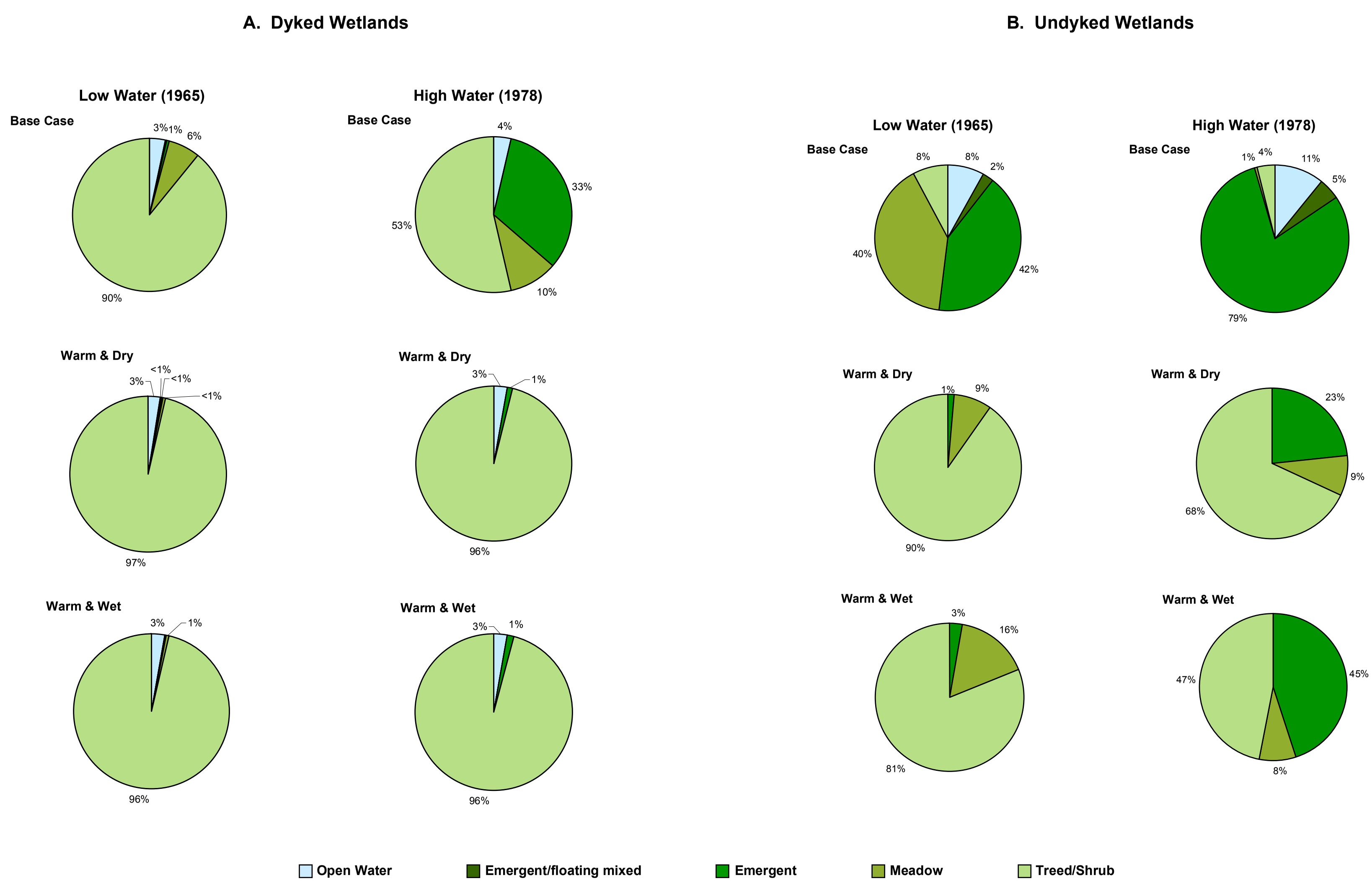


Figure 8.34 Predicted changes in wetland vegetation under low and high water cycles for the base case, warm & dry, and warm & wet climate change scenarios in dyked (A) and undyked (B) wetland study sites, Lake St. Clair

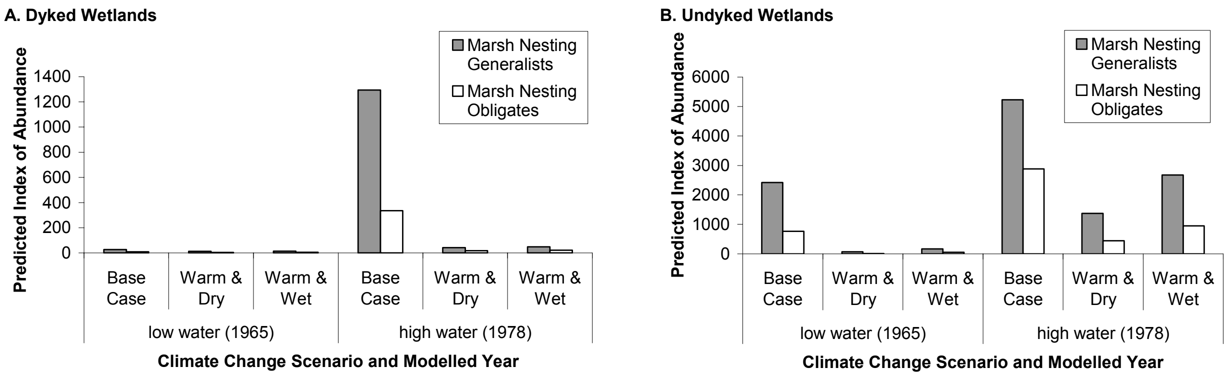


Figure 8.35 Predicted changes in indices of the abundance for marsh nesting birds under low and high water cycles for the base case, warm & dry, and warm & wet climate change scenarios in dyked (A) and undyked (B) wetland study sites, Lake St. Clair

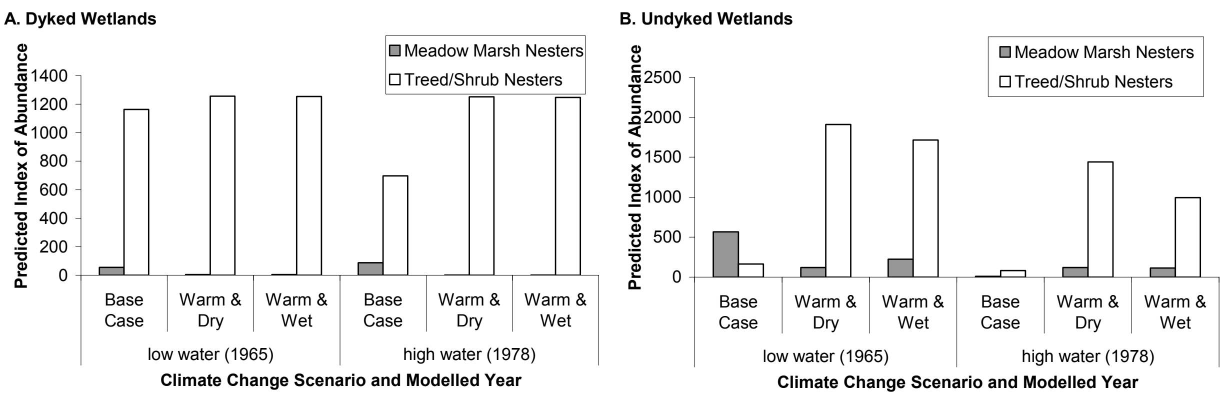


Figure 8.36 Predicted changes in indices of the abundance for meadow marsh and treed/shrub nesting birds under low and high water cycles for the base case, warm & dry, and warm & wet climate change scenarios in dyked (A) and undyked (B) wetland study sites, Lake St. Clair

Under the climate change scenarios, the predicted area of emergent vegetation was reduced and open water habitat disappeared (Figure 8.34B). The predicted index of abundance for marsh nesting birds also declined, with a shift to more treed/shrub bird species in response to the predicted increase in area of this wetland habitat (Figure 8.36B).

Summary

The dyked wetland study sites were predicted to support less emergent marsh habitat in an undyked state under any water level or climate change scenario. These wetland areas were predicted to change to a predominantly treed/shrub habitat under lower water levels associated with the climate change scenarios. The primary purpose of the dyked and undyked comparison was to evaluate the ability of dyked wetlands to maintain emergent marsh plant and bird communities in an undyked state. For this evaluation, modelling was restricted to very specific wetland areas that were within the dyked and associated undyked study area. Unlike the wetland modelling represented in Chapter 7, these results do not evaluate the potential for movement of plant and bird communities up or down slope outside of the specific study site boundaries. As a result, in wetlands that are currently dyked, emergent habitats may be maintained or possibly expand into new areas with climate change.

It is clear from the model predictions that the wetland areas currently dyked in Lakes Ontario, Erie, and St. Clair would shift to predominantly treed/shrub habitats under lower water levels associated with the climate change scenarios. Wetland dyking and pumping infrastructure would be required to maintain emergent habitats within these specific areas if that is the management objective. However, the ability of the wetland vegetation and bird communities to naturally adapt to water level fluctuations, and long-term reduced levels should be important considerations when evaluating the use of dyking as a climate change adaptation strategy within a coastal wetland.

8.2.7 Assessment of Current Wetland Dyking as an Adaptation Strategy to Climate Change - Wetland Dyke Design and Potential Infrastructure Problems in Relation to Predicted Changes in Water Levels in each of the Great Lakes due to Climate Change

Dyking a wetland involves modifying the existing hydrological connection between the wetland and its water source by a human-made barrier, designed to alter the inflow or outflow of water and to protect the wetland from lake forces (Environment Canada 2002). Although the hydrology of the wetland becomes primarily regulated through the use of pumps, underground flumes, gravity-flow gates, or stop logs (Sherman *et al.* 1996), water levels in some dyked wetlands may also be affected by runoff from adjacent land and seepage through dyke barriers. For example, Cranberry Marsh is a dyked wetland, where connectivity to the lake is impeded by stop logs but it is not encompassed by a dyke. Instead, it is isolated from the lake by a barrier beach and is naturally refilled by runoff from over 126.5 ha of adjacent land (Central Lake Ontario Conservation Authority 1999). Similarly, the dyked wetland on Amherst Island receives some runoff from adjacent land but it is also refilled with water from pumps and has a partial, or U-shaped, dyke. Consequently, use of natural barrier beach bars and U-shaped dykes maintain some level of natural hydrological connectivity and variability. In contrast, ringed dykes, such as the National Wildlife Areas at Lake St. Clair and Long Point, are fully enclosed wetlands that have water diversion channels (i.e. ditches) around their periphery. The hydrology in these wetlands is manipulated by anthropogenic regulation with limited natural hydrological variability (i.e. direct precipitation).

Changes in precipitation and lake water levels, due to climate change, will affect barrier beach, U-shaped, and ring-shaped dykes differently. Because precipitation and the resulting runoff are the main factors in reflooding and maintaining water levels in barrier beach and U-shaped dykes without pumping structures, any change in precipitation will affect the hydrology in these wetlands. The frequency and intensity of extreme meteorological events, such as flash floods and drought, are predicted to increase due to climate change (see Chapter 2). Subsequently, the variability of water levels in these wetlands will likely increase.

Ring-shaped and U-shaped dykes with pumping structures will likely be affected less by changes in precipitation but more by changes in lake water levels due to problems associated with the pump infrastructure. Historically, dyke construction, pump infrastructure, and associated water intake channels were designed to operate under "normal" water level ranges for specific lake basins. Dyke design and management has given little consideration to a permanent drop in lake water levels due to climate change. If water levels on the Great Lakes drop according to climate change scenarios (see Chapters 2 and 7), two water intake problems may arise. First, water pump intake pipes and/or dredged channels may dry, making pumps inoperable. Second, shifts in shoreline processes and vegetation communities may reduce water flow into intake channels resulting in reduced water pumping capacity.

8.2.7.1 Lake St. Clair NWA – A Case Study of Vulnerability to Water Level Decrease

The Lake St. Clair NWA dyked wetland complex was chosen as a case study to illustrate possible infrastructure problems resulting from projected drops in lower water levels of up to one metre due to climate change. It is the oldest dyked wetland of the study sites, and was built during relatively high water levels (Table 8.16; Figure 8.37) and potential problems with dyke design relative to current and future low water levels should emerge.

Table 8.16 Characteristics of pump infrastructure and associated operating thresholds for dyked wetland sites

Wetland	Date constructed	Wetland size (ha)	Dyke type	Infrastructure	Pumping schedule	Water level elevation (m) required for pump intake pipes to function	Elevation range (m) of water intake channel	Percent of 50-year hydrological cycle that pumping infrastructure is inoperable under Climate Scenario				
								Base Case	Warm & Wet	Warm & Dry	Not as Warm & Dry	Not as Warm & Wet
Amherst Island	1997	143.52	U-Shaped	Pump system / Stop logs	July – Sept	74.20	74.23 – 75.25	0√	0√	0√	0√	0√
Lynde Shores (Cranberry Marsh)	2001	47	Natural barrier	No pump / Stop logs	None	None	None	-	-	-	-	-
Long Point (Big Creek NWA)	1985	86.63	Ringed	Pump system / Stop logs	July – Sept	173.75	173.75*	0	26	46	20	2
Lake St. Clair NWA	1980/81	232.62	Ringed	Pump system / Stop logs	July – Sept	Main pump: 175.10 Auxiliary Pump: 173.86	174.60 – 175.07	14	94	98	78	26
Hillman Marsh	1988	66.46	Ringed	Pump system / Stop logs	July – Sept	173.60	173.70 – 174.26	0	24	40	14	0

* Channel was dredged in 2003
√ Base Case and Climate Change Scenarios for Lake Ontario were calculated from a 25-year hydrological data set. Lake level projections for Lake Ontario include Base Case = 75.11 m, Warm & Wet = 74.50 m, Warm & Dry = 74.36 m, Not as Warm & Dry = 74.65 m, and Not as Warm & Wet = 75.17 m; Lake Erie include Base Case = 174.46 m, Warm & Wet = 173.79 m, Warm & Dry = 173.65 m, Not as Warm & Dry = 173.92 m, and Not as Warm & Wet = 174.32 m; and Lake St. Clair include Base Case = 175.38 m, Warm & Wet = 174.57 m, Warm & Dry = 174.40 m, Not as Warm & Dry = 174.75 m, and Not as Warm & Wet = 175.18 m.

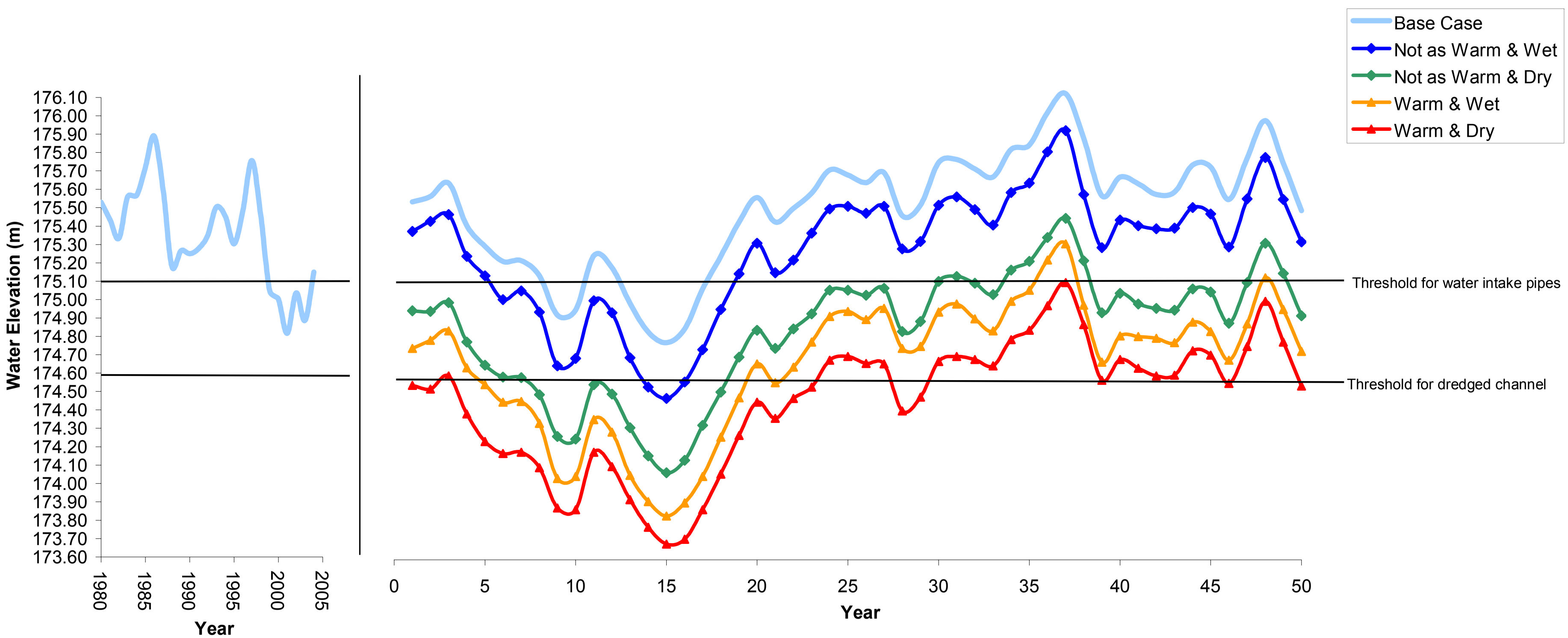


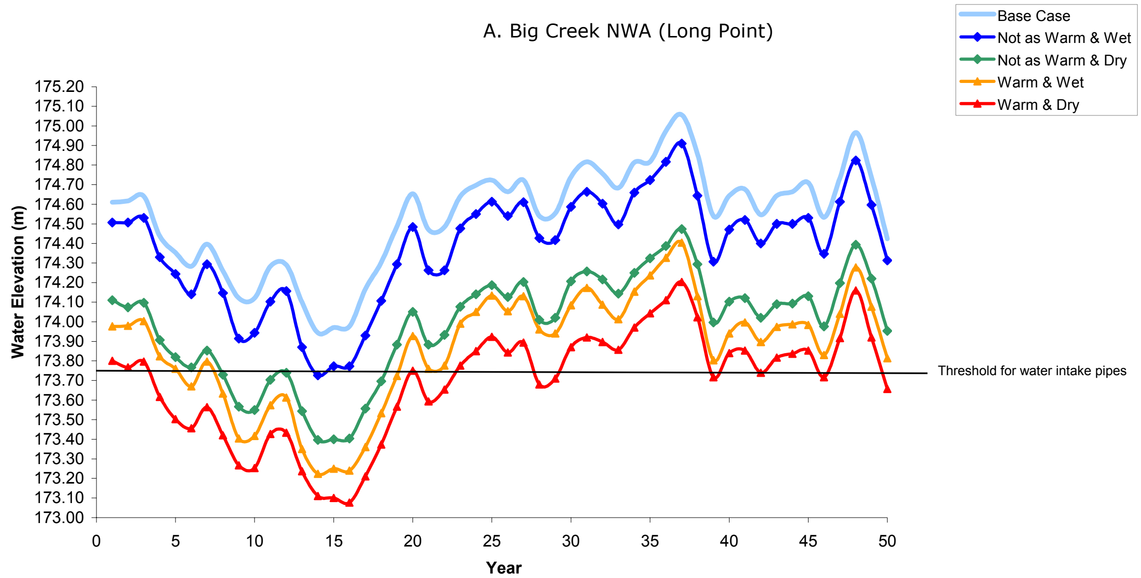
Figure 8.37 Lake St. Clair average water level for the 3-month pumping period (July–September) from 1980 to 2004, and modelled 50-year time series of average 3-month pumping period under the base case, not as warm & wet, not as warm & dry, warm & wet, and warm & dry climate change scenarios. Estimated elevation thresholds for the water intake pipes and channel.

Lower water levels on Lake St. Clair may compromise the capability to pump water into the NWA dyked wetlands. Water intake pipes for the main dyke pump at Lake St. Clair were inoperable in 2004 because of insufficient supply of water to the pumping system. This problem first occurred in 1999, when water levels in Lake St. Clair averaged 175.05 m during the 3-month pumping period (i.e. July-Sept.). In the 19 years prior to 1999, the lowest average water level for Lake St. Clair during the pumping period was 175.18 m (Figure 8.37) and pumping was possible throughout this period. The threshold water level elevation for functioning of the water intake pipes at the NWA is likely around 175.10 m. It was estimated that the current pumping design would be inoperable for greater than 75% of the 50-year water level time series associated with the warm & dry, warm & wet, and not as warm & dry climate change scenarios (Table 8.16). Pumping infrastructure upgrades would be required for the main water intake pipes to function at the NWA under these scenarios. Even under the base case scenario without climate change, the pumping system was expected to be inoperable for 14% of the 50-year water level time series.

Low water levels on Lake St. Clair also affect the supply of water to the pumps through a channel from the lake. Water pumps will not operate when there is insufficient water supplied to the intake pipe to keep the pump submerged. To examine the potential for this problem, the elevation range of the intake channel for the NWA main and auxiliary pumps was estimated from bathymetric data (Table 8.16). This range was compared to historical water levels and the projected 50-year water level time series for each climate change scenario for Lake St. Clair (Table 8.16; Figure 8.37). The intake channel has maintained an operable water depth since 1980, and was predicted to maintain an operable water depth without climate change (i.e. the 50-year hydrological cycle for the base case). Water depth in the intake channel would be significantly shallower for the 50-year hydrological time series in three of the four climate change scenarios. Climate change could result in higher maintenance costs associated with dredging the intake channel than what has occurred historically or would have occurred without climate change as represented by the 50-year base case. Consideration must also be given to a reduction in water flow through intake channels because of shoreline processes and shifts in vegetation communities. Climate change may increase the frequency and intensity of extreme meteorological events and resulting storm surges, winds, and wave action may alter shorelines causing more infilling of dredged channels. Recent low water levels on Lake St. Clair have resulted in the lakeward expansion of the emergent marsh vegetation, with some areas now having bands of emergent vegetation that are hundred of metres wide. This expansion may create additional challenges for maintaining the intake channels as they are choked by plant growth. Shallower water levels in the channel may also promote submerged aquatic vegetation growth. Extensive vegetation growth may impede water flow through the intake channels and require more frequent dredging to maintain a sufficient flow of water to intake pipes.

8.2.7.2 Summary

Undoubtedly, lower water levels on the Great Lakes will pose new challenges for the maintenance of water levels within dyked wetlands. While it is difficult to generalize potential problems within all dyked wetlands on the Great Lakes due to differences in dyke infrastructure (Table 8.16; Figure 8.38), one management aspect is apparent – the costs of maintaining current habitats within existing dyke wetland sites will increase. Therefore, consideration should be given to the potential effects of climate change on the original objectives for the dyked wetland projects when contemplating future investments in maintaining and upgrading dyke infrastructure as an adaptation to climate change.



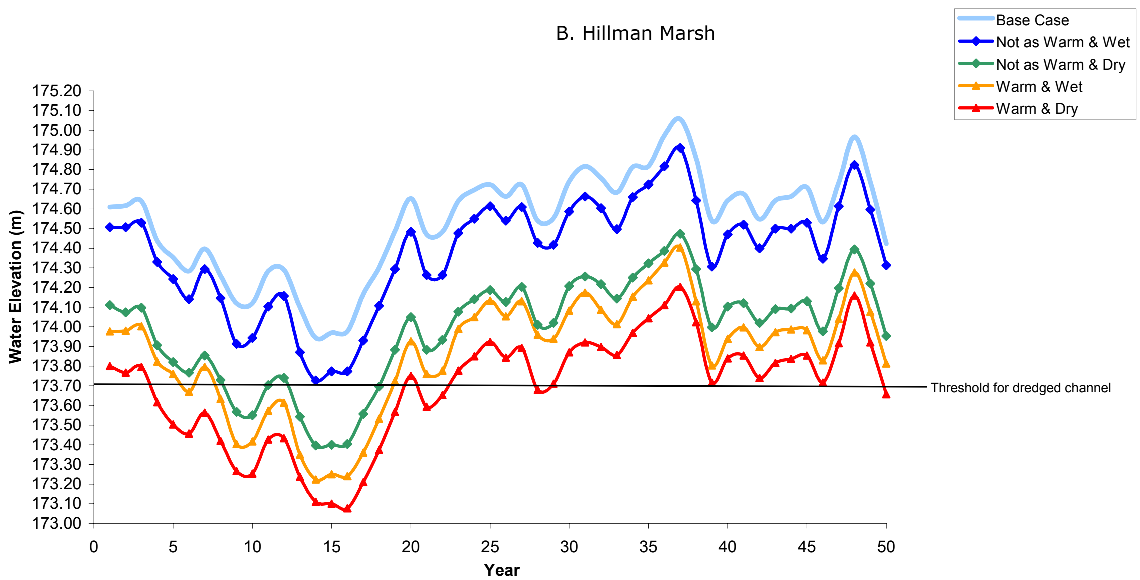


Figure 8.38 Lake Erie average 3-month pumping period (July-September) water level for the projected 50-year time series under the base case, not as warm & wet, not as warm & dry, warm & wet, and warm & dry climate change scenarios, and the water intake pipe threshold elevations at the Big Creek NWA (Long Point) (A) and Hillman Marsh (B)

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8.3 LAND USE PLANNING

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Land use planning can be a tool to help wetlands adapt to climate change (Bruce *et al.* 2000) but to date reports offer either very broad overviews (e.g. Kling *et al.* 2003; Easterling *et al.* 2004) or specifics applicable to ocean coasts where the issue is rising rather than falling water levels (e.g. Titus 1998). Other land use planning reports discuss mitigation rather than adaptation, citing measures such as urban intensification to reduce greenhouse gas emissions (e.g. Silsbe 2003). From an Ontario perspective, Smith *et al.* (1998) list adaptation options of habitat protection, expanded refuges, landscape corridors, and flexible land use zoning around nature reserves to allow movement as climate changes. In this section, these options are explored more deeply, starting the discussion of how they might assist Ontario's Great Lakes coastal wetlands' adaptation to climate change.

Many policies and programs call for coastal wetlands to be secured and restored. Some like Ontario's Provincial Policy Statement require consideration of effects of immediately adjacent uses. A few recognize the severe habitat fragmentation of parts of the Great Lakes basin and the need for better linkage. But climate change shifts the picture from static to moving, and the scale from local to international. Planning must account for broader dimensions of space and time. This adjustment has some challenges. Land use planning is largely the responsibility of local jurisdictions that have no direct influence on upstream or on neighbouring jurisdictions' uses and management. No one has had to deal with the concept of land uses that move. The recent provincial planning reform documents (Ontario Ministry of Municipal Affairs and Housing 2004, 2005a,b), however, can be interpreted to offer some support for a new policy direction of moderating the impacts of climate change through supportive land-use patterns.

8.3.1 Ontario's Land Use Planning Context

Land use planning for coastal wetlands should be considered in light of other trends affecting land use. By 2031, over four million more people are predicted in Ontario (Ontario Ministry of Public Infrastructure Renewal 2004) – most in the Great Lakes basin, concentrated in the Golden Horseshoe within easy reach of Lakes Ontario and Erie This growth will bring: pressures for recreation especially along lake shores as temperatures rise (Wall 1998); escalating water demand; increased need for protection of already stressed and fragmented natural areas; and a greater urgency to develop citizens' ecological understanding and sense of place to minimize their ecological footprint (Dempsey 2004).

Recent shoreline land use trends include shoreline sprawl with bigger homes, less agriculture, and less public access (Zuzek and Warbach 2002). Positive directions include: a growing awareness of the problems and expense of the traditional built shore protection approach, a shift toward staying out of nature's way – restoring shores and dunes and slowing erosion by managing water (Keillor 2002), and a greater recognition of the services natural systems provide society (Costanza *et al.* 1997).

Land use planning in Ontario is controlled primarily through the *Planning Act*; the Provincial Policy Statement under Section 3 of the Act provides policy direction. Municipalities implement the *Planning Act* and associated policy through Official Plans. The land use policies of Official Plans are put into regulation through the adoption and approval of zoning by-laws. At report writing, the provincial government is in the midst of extensive planning reform.

8.3.2 Planning Criteria for Coastal Wetland Adaptation to Climate Change

Coastal wetland planning, confronted with the double threat of climate change and population growth, must meet several conditions for ecological health and public acceptance:

1. Protect existing wetlands,
2. Expand wetland protection lakeward over time as wetlands shift,
3. Prepare for changes in shoreline processes, storms and ice protection,
4. Manage watersheds to mitigate upstream stressors,
5. Connect coastal wetlands to the greater coastal landscape,
6. Raise public awareness,
7. Involve shoreline owners,
8. Increase public access to the shoreline,
9. Minimize costs, and
10. Provide an implementation structure.

Rationale for each criterion is summarized under "Why?" in Section 8.3.3.

8.3.3 Existing Policies and Planning that Fulfill the Ten Planning Criteria

No instance was encountered of land use planning or policy directed specifically to adaptation of Great Lakes coastal wetlands or coasts to climate change (Barron pers. comm.; Baskerville pers. comm.; Donnelly pers. comm.; Hayman pers. comm.; Hill pers. comm.; Mortsch pers. comm.; Moulton pers. comm.; Puddister pers. comm.; Taylor pers. comm.). The following discussion outlines Ontario's existing land use planning and policy developed without stated climate change considerations but applied to the Ten Planning Criteria for coastal wetland adaptation. Some issues to consider are listed under each criterion. Issues include knowledge gaps, implementation hurdles, and public misconceptions. Appendix 8.3 provides a glimpse of coastal wetland planning elsewhere in the basin ecosystem with a few examples of American state and federal planning.

8.3.3.1 Protect Existing Wetlands

Why? *The first line of defence against climate change effects on wetlands should be protection of existing wetlands (Ramsar Convention on Wetlands 2000).*

Many policies and programs call for wetlands to be secured and restored (e.g. Federal Policy on Wetland Conservation, Canada - Ontario Agreement Respecting the Great Lakes Basin Ecosystem - Lake-wide Management Annex, *Species At Risk Act*, Great Lakes Wetlands Conservation Action Plan, North American Waterfowl Management Plan).

The Provincial Policy Statement prohibits development in provincially significant coastal wetlands and on lands immediately adjacent to those wetlands if development would have a negative impact on wetland features and functions (Ontario Ministry of Municipal Affairs and Housing 2005b). Many coastal wetlands, however, particularly along the upper lakes, have not been evaluated for their significance (Environment Canada and OMNR 2003) and some may be undelineated. In addition, some existing evaluations may be outdated as wetlands change (Environment Canada and OMNR 2003) and will become more outdated with the effects of changing climate. Wetlands smaller than two hectares are excluded from the evaluation.

Each municipality must integrate the applicable parts of the Provincial Policy Statement into its Official Plan. The 2005 Provincial Policy Statement strengthens the wording to the mandatory “shall be consistent with” over the previously weaker “shall have regard to”. Agriculture, although historically responsible for the majority of southern Ontario wetland conversions (Snell 1987), is exempt from the Provincial Policy Statement. Significant coastal wetland conversion to agriculture has occurred in the vicinities of Point Pelee and eastern Lake St. Clair.

Through mechanisms such as Environmentally Sensitive Areas (ESAs), municipalities can protect natural areas beyond the scope of provincial policies. Where ESA inventories exist, however, they vary greatly in quality and comprehensiveness (Reid 2001).

Under the new generic regulation of the *Conservation Authorities Act*, Conservation Authorities can prohibit development in wetlands with no qualifier on the size or significance of the wetland (Ontario Gazette 2004). Conservation Authority shoreline jurisdiction, however, with the exception of the shorelines near Thunder Bay and at Sault Ste. Marie, is limited to the lower lakes below Wasaga Beach. Even in the lower lakes, jurisdictional gaps occur for the top half of the Bruce Peninsula, parts of the St. Lawrence River shoreline and islands, and Kettle Point area.

Protection also occurs through public ownership. In some cases, land purchases target sites representative of particular habitats. If climate change shifts continental biomes, the level of representativeness of such publicly owned lands could change (Scott and Lemieux 2003).

Other legislation protects wetlands against the effects of specific types of projects:

- The federal *Fisheries Act* prevents damage or destruction of wetlands that contribute to a fishery and includes provisions for pollution prevention. Penalties include fines, imprisonment, and orders requiring restoration of damaged fish habitat (WetKit 2004);
- The provincial *Public Lands Act* requires permits for any work such as dredging or filling that involves the bed of a water body including nearshore areas;
- The provincial *Lakes and Rivers Improvement Act* requires a work permit for any activity that alters flow (Fisheries and Oceans Canada 2003);
- The provincial *Environmental Protection Act* includes requirements regarding contaminant emissions; and
- Environmental Bill of Rights provides a set of participatory tools for the public to hold the government accountable for protection, conservation, and restoration of the natural environment for present and future generations. The Act’s goals include protecting and conserving ecological diversity, ecological systems and ecological sensitive areas and processes (Environmental Commissioner 2003).

Voluntary wetland protection includes projects by Stewardship Councils, land trusts, private non-profit organizations, Conservation Authorities, community groups, and individual landowners. A major project is the Eastern Habitat Joint Venture of the North American Waterfowl Management Plan, a public and private partnership. Tax incentives such as provincial Conservation Lands Tax Reduction Program and federal Ecological Gifts Program shift conservation stimuli from restrictions to financial incentives and raise owner awareness of wetlands’ valuable ecological services. Although these programs are gaining recognition in Ontario, they remain relatively under-utilized (Reid 2001).

Protection includes reduction of current stressors. Wetlands most threatened by climate change are those currently under the most stress from other sources. Easing of other stressors will increase wetlands’ ability to deal with climate change (McCarthy *et al.* 2001; Poff *et al.* 2002). Planning measures such as habitat linkage or buffer creation can enhance the resilience of ecosystems and their capacity to respond to climate change

(Easterling *et al.* 2004). Many regulations deal with control of individual stressors but few tackle cumulative or potential long-term effects on wetlands and other habitats.

Existing federal measures against harmful alien organisms include controlling the trans-boundary movement of specifically identified organisms through the *Wild Animal and Plant Protection and Regulation of International and Interprovincial Trade Act*, mapping alien species through the Biodiversity Mapping Program, monitoring species expansion, and raising awareness of the threat (Environment Canada 1998). A strategy to manage invasive plants in southern Ontario proposes a number of broad actions and policy reviews, and as a protective measure, suggests wide corridors and development setbacks to protect natural areas (Havinga 2000).

Issues to Consider

- Undelineated and unevaluated coastal wetlands (particularly in the north).
- Inadequate protection from invasive species.
- Lack of regulation over loss of wetlands to agriculture.
- Shoreline gaps in Conservation Authority jurisdictions.
- Lack of public awareness of tax reduction opportunities of wetland protection.
- Lack of acknowledgement to wetland owners for the ecological services their lands provide.
- Need to consider wetland protection from influences outside protected boundaries.
- Need to understand and respond to cumulative effects.

8.3.3.2 Expand Wetland Protection Lakeward Over Time

Why? *As lake levels recede with climate change, coastal wetlands may shift location, making current protection boundaries inadequate.*

No practical examples were found that specifically address flexible wetland boundaries which are applicable to a receding shoreline. For example, in the nine pages of references for Ramsar's report on wetlands and climate change (2002), the only studies of falling water levels were the relatively few Great Lakes examples and none dealt with adaptation. All remaining studies appeared to be related to rising sea levels and ocean coasts.

For ocean coasts, rolling easements have been proposed to help wetland retention in the face of rising water levels. A government-purchased easement would allow the owner to keep development rights, excluding structures to hold back the sea. Once inundated, the property would go to the easement holder. The easement's proactive nature minimizes cost (Titus 1998).

In Ontario, jurisdiction over newly exposed lands has been unclear. For a recent complaint about dredging through land exposed by low lake levels among Lake Huron's Fishing Islands, OMNR claimed no jurisdiction to regulate on the basis that anything exposed over a year is accreted land belonging to the shoreline property owner. The Conservation Authority had no jurisdiction; its regulations applied to construction of structures but not to excavation along the shoreline. Although DFO can have jurisdiction on these areas that are periodically flooded, it is not staffed to deal with the potential of many instances along the Great Lakes shoreline. The result was a no-man's-land between high water mark and the water and, in the Fishing Islands instance, landowner lawsuits (Hill pers. comm.).

Recent changes should help clarify the situation by extending Conservation Authority jurisdiction to the international boundary (McColl 2005). The new Provincial Generic Regulation mandates Conservation Authorities to prohibit development in wetlands (Ontario Gazette 2004). Several hurdles remain. All Conservation Authorities must obtain approval for regulations consistent with the new Provincial Generic Regulation by May 2006 when the old regulations expire. Compliance requires new shoreline mapping and schedules. Some Conservation Authorities lack the staff or funding to conduct this mapping, risking no compliance in May 2006 and, with the expiry of the old regulations, no regulation at all. Furthermore, given the increase in responsibility and extent of the new jurisdiction, few rural Conservation Authorities have the staff capacity for implementation (Hill pers. comm.). In addition, most of the upper lakes shoreline and small parts of the lower lakes and St. Lawrence River shorelines have no Conservation Authority. Finally, if shoreline conditions and coastal wetlands shift as lake levels recede with climate change, on-going protection of the wetlands will require on-going revisions to mapping.

Issues to Consider
- Lack of clarity regarding ownership and agency responsibility for regulating newly exposed lands due to water level decline.
- Lack of shoreline mapping and schedules.
- Lack of funding and staff to meet regulation criteria and enforce them.
- Shoreline gaps in Conservation Authority jurisdictions.
- Lack of public and decision-maker perception of the dynamic nature of the Great Lakes shoreline and coastal wetlands and of the associated need to adjust policies and boundaries accordingly.

8.3.3.3 Prepare for Changes in Shoreline, Storms, and Ice

Why? *Projected climate change includes more extreme storms and less ice protection from storm effects. Coastal wetlands could be directly damaged and suffer indirect impacts from storm-altered shoreline sediment processes. Areas with coastal armouring, dams, and harbours may lack beach building material as shorelines seek to re-establish themselves (Keillor 2002) hindering the successful lakeward shifts of affected barrier-protected wetlands.*

Conservation Authorities' Shoreline Management Plans include consideration of shore processes but often in response to high lake levels with little discussion of climate change effects and response (Donnelly pers. comm.).

The 2005 Provincial Policy Statement directs development outside hazard lands adjacent to the shorelines of the Great Lakes-St. Lawrence River and impacted by flooding, erosion, and/or dynamic beach hazards. The generic regulation (Ontario Gazette 2004) defines flooding hazards as extending to the 100-year flood level plus an allowance, determined by the Conservation Authority, for wave uprush and other water-related hazards such as boat wakes and ice piles. The erosion hazard limit uses the 100-year erosion rate. Dynamic beach hazards go beyond the flooding hazard to include highly unstable shorelines, dune protection, and areas where development would alter sediment transport or cause other adverse environmental impacts. Uncertainties include whether limits based on past rates will be appropriate for future conditions altered by climate change. As shoreline processes change, Conservation Authorities (where they exist) will need to recalculate and re-map hazard area extent, and then alter their regulated area (McColl 2005).

Issues to Consider
- Lack of public and decision-maker perception of the dynamic nature of the Great Lakes shoreline and associated need to adjust policies and boundaries accordingly.
- Lack of public and decision-maker awareness of the ecological services provided by undeveloped and naturally functioning shorelines.
- Shoreline gaps in Conservation Authority jurisdictions.

8.3.3.4 Manage Watersheds to Mitigate Upstream Stressors

Why? *River-mouth coastal wetlands are particularly susceptible to watershed stressors imposed by upstream residents. These stressors both reduce wetland resilience to climate change effects and will themselves intensify with climate change.*

Most of southern Ontario, the area of the province with the most intense land use and upstream effects, is subject to watershed-based management through Conservation Authorities. These agencies work closely with municipalities to conserve, restore, develop, and manage natural resources on a watershed basis. For half a century, these agencies have conducted projects on:
- Watershed strategies and management,
- Flooding and erosion protection,
- Water quality and quantity,
- Reforestation and sustainable woodlot management,
- Ecosystem regeneration,
- Environmental education and information programming,
- Land acquisition,

- Outdoor recreation,
- Soil conservation,
- Environmental land use planning,
- Habitat protection,
- Agricultural and rural landowner assistance, and
- Sensitive wetlands, flood plains, valley lands protection.

The types of projects that benefit overall watershed water quantity and quality also reduce negative stressors on wetlands near watershed outlets to the Great Lakes. Some specific initiatives benefiting downstream wetlands include:

- Rural Water Quality Protection Programs where downstream urban areas benefiting from upstream water protection measures contribute to their cost, thereby saving money from the more expensive water treatment alternative;
- Natural Channel Design where streams are restored to their natural form resulting in naturally moderated flows and sedimentation processes; and
- Source Water Protection Planning where action will focus on protecting water quality at upstream sources.

The 2005 Provincial Policy Statement requires the use of watershed units for water resource planning; minimization of negative impacts within and across watersheds; identification, protection, and restoration of natural heritage features that provide hydrological functions; maintenance of linkages among features; and promotion of conservation and stormwater management that minimizes volumes and contaminant loads (Ontario Ministry of Municipal Affairs and Housing 2005b). Implementation capacity challenges include: improving background data, adequate on-going funding, building linkages to Official Plans, strong local leadership, and broad citizen participation (de Loë 2004).

Lake-wide Management Plans are an initiative of the 1987 Protocol to the Great Lakes Water Quality Agreement. They are bi-national for each Great Lake basin and aim to protect and restore habitat. They work with the public to set and implement targets (Environment Canada 2003) but are much less locally active than the river basin-specific Conservation Authorities.

Several coastal wetlands are located within the boundaries of the 17 Areas of Concern in Canada that are subject to Remedial Action Plans. Wetlands in these areas benefit as remedial actions are implemented throughout their watersheds.

Many other initiatives, although not explicitly watershed-based, benefit all downstream uses including coastal wetlands. Examples include: government initiatives such as sewage treatment plants, non-government projects such as Ducks Unlimited Canada's Ontario Land Care assistance to farmers for soil and water conservation measures, and individual landowner stewardship such as the highly successful Environmental Farm Plans.

Upstream pressures are projected to grow with climate change both from direct effects such as greater flooding and sedimentation and from climate-induced land use change. Given the soils distribution in the Great Lakes basin, however, it seems unlikely that vast new agricultural zones will be cleared, at least on the Canadian side. On the other hand, population growth will place increasing pressure on all natural resources.

Issues to Consider

- Lack of citizen awareness that they live in a watershed and their role in its health.
- Conservation Authority funding issues including long-time certainty.
- Need to link watershed concerns to local planning.
- Need for integration, overcoming jurisdictional silos.

8.3.3.5 Connect Coastal Wetlands to the Greater Coastal Landscape

Why? *In intensively used urban and agricultural areas dominant in the lower Great Lakes basin, habitat fragmentation is increasingly recognized as a serious threat to natural heritage maintenance as well as to resiliency to climate change (McCarthy et al. 2001; Reid 2001; Easterling et al. 2004). Wetlands are often so fragmented that wetland plants and animals cannot naturally "migrate" to new locations in response to climate changes (Kusler 1999).*

Section 2.1.2 of the 2005 Provincial Policy Statement, states that connectivity of natural features should be maintained, restored, or where possible, improved, recognizing linkages between and among natural heritage features and areas, surface water features, and groundwater features. It also requires maintaining linkages and related functions among water features, hydrologic functions, and natural heritage features (Section 2.2.1e, Ontario Ministry of Municipal Affairs and Housing 2005b).

A number of coastal linkage and planning initiatives are occurring:

- The Lake Ontario Greenway Strategy, coordinated by the Waterfront Regeneration Trust, includes protection of shoreline features, increased public understanding and shoreline access, and encouragement of Greenway-compatible economic activities through public/private cooperation (Reid 2001). In a survey of Canadian and American coastal land uses of Lake Ontario and the St. Lawrence River above Cornwall, the highest proportion of parks and open space corresponded to areas where the Waterfront Regeneration Trust – a non-profit organization dedicated to promoting the regeneration of the Lake Ontario waterfront and Greenway – was actively involved (Christian J. Stewart Consulting 2004).
- The Great Lakes Heritage Coast is a project launched by the Province of Ontario in 2000. It covers 2,900 km including the Canadian coastline of Lake Superior, St. Mary's River, northern Lake Huron, and Georgian Bay to the Severn River. The project's goal is the long-term protection of the Heritage Coast's significant values and natural resources, while promoting the tourism benefits of its scenic and cultural heritage (Petersen 2000).
- Some coastal inventory and classification have been assembled as a possible basis for future protection initiatives. The Great Lakes coast has been rated into six categories for biodiversity based on species and communities of special interest, diversity of habitats, productivity, and integrity (SOLEC 2001). Regions of shoreline ranked in the top two categories for biodiversity cover 22% of the shoreline, with a higher proportion found on the Canadian side. They include Prince Edward County vicinity, Long Point Bay, and the east and north shores of Georgian Bay to Sault-Ste Marie. These designations were intended to raise awareness of important areas.
- The Nature Conservancies of Canada and United States are establishing a conservation blueprint of the Great Lakes basin to map key conservation locales at a landscape and ecoregion levels (Environment Canada 2003). Shoreline areas are included.

Inland-focused natural heritage network initiatives could connect to coastal corridors. The OMNR Natural Heritage Information Centre conducted the Big Picture project for Carolinian Canada and Bigger Picture project for southern Ontario that mapped both existing natural areas and proposed optimum routes for connecting corridors (McMurtry *et al.* 2002). Some areas such as Essex and Chatham-Kent have so few existing natural areas that proposed connecting corridors are also very sparse. Several regional municipalities (e.g. Waterloo, Halton, Ottawa-Carlton) have incorporated natural heritage networks into their planning. Natural network and corridor concepts, including shoreline components, are being actively promoted within Conservation Authorities.

Issues to Consider

- Urgency of current planning issues prevents consideration of long-term trends and opportunities.
- Terrestrial-based natural heritage networks can overlook some aquatic links required in a coastal environment.
- Jurisdictional mandates mask mutually beneficial opportunities and create unregulated gaps in natural heritage protection.

8.3.3.6 Raise Public Awareness

Why? *Widespread understanding of important issues such as climate change improves decision-making capacity at every level and increases society's collective capacity to adapt (Easterling et al. 2004). As governance shifts from state-centric to more citizen involvement, citizen awareness is essential (de Loë 2004). Some planning decisions overlook current knowledge relevant to natural heritage protection because it is not presented in an accessible format. Over the last 200 years, many Great Lakes decisions regarding fisheries, water pollution, and shoreline preservation have been delayed through lack of political will, to the detriment of the ecosystem (Dempsey 2004). Fisheries have deteriorated, waters have been degraded, and ecological areas paved over despite studies, warnings and even proposed legislation. Often significant response to environmental issues occurs only when the public's attention is caught by extreme events or by a highly effective champion for a particular issue.*

Effective public information on ecosystem values is produced by national and provincial coastal parks, Conservation Authorities, partnerships such as the Great Lakes Wetlands Conservation Action Plan and the Lake Ontario Greenway Strategy, and non-government groups such as Ducks Unlimited Canada and Ontario Nature (formerly Federation of Ontario Naturalists). Citizens less connected with these agencies and possibly less knowledgeable of ecology may have less access. Even with access, rushed lifestyles and immediate concerns divert attention elsewhere.

Issues to Consider

- Need to inform decision-makers of shoreline issues in brief, effective, understandable formats.
- Culture, entertainment, emotional, or spiritual appeal to raise public awareness of ecosystem values often not used.
- Lack of public understanding of the value of ecological services combined with too much faith in technological solutions.
- Public sees immediate problems like point sources but rarely appreciates the severity of long-term more intangible problems like climate change nor their own role in causing and solving problems.
- Lack of time to read information.
- Issues overlooked due to lack of monitoring or to lack of publicly accessible monitoring results.
- Terminology (e.g. "environment" over "ecosystem") can obscure citizen role.

8.3.3.7 Involve Shoreline Owners

Why? *For successful implementation, shoreline planning must involve shoreline owners. Owner involvement will raise their awareness of the issues, clarify their needs, and bring their commitment to the results.*

Property owners often react to lower water levels by dredging, adjusting docks, restricting boat size, relocating boats, and establishing floating docks (Wall 1998). In 2000, DFO initiated the Great Lakes Water Level Emergency Response Program providing $15 million in dredging assistance to marinas severely affected by low water levels. But in many situations, very high costs and contaminated sediments limit the applicability of this program (Warren with Lemmen 2004). Shoreline owner involvement occurred in Conservation Authority Shoreline Management Plans.

Issues to Consider

- Lack of participatory processes that involve shoreline users and owners in shoreline planning and issue resolution.
- Lack of compensation by society for the ecological services shoreline owners can provide.
- Lack of financial and technical assistance to help shoreline owners adapt to change.
- Lack of long-term thinking to appreciate the dynamic nature of the shoreline.
- Lack of shoreline owner awareness of natural shorelines' ecological services.

8.3.3.8 Increase Public Access to the Shoreline

Why? *Personal experience of an area is important in building awareness and commitment to its future. As climate warms and gas prices rise, nearby shoreline recreation demand will grow, offering the opportunity for that personal experience. Access must be managed to provide an enriching visit in touch with nature.*

Current public access to the Canadian Great Lakes shoreline includes National Parks, Provincial Parks, Conservation Areas, and Municipal Parks. A few public lands (e.g. some National Wildlife Areas) have restricted access to help maintain ecosystem integrity. For some shorelines, like Lake St. Clair, private ownership minimizes public access. In non-regulated areas with accessible shoreline, motorized vehicles are increasingly disruptive.

The *Public Lands Act*, administered by OMNR, directs the Ministry to "have charge of the management, sale, and disposition of the public lands", and to regulate Crown Lands regarding development, management, use, and rehabilitation. Although the Act authorizes MNR to define "shore lands", ownership (private or public) of a dynamic shore does not appear to be clear (Ausable-Bayfield Conservation Authority 2000; Hill pers. comm.). Ontario property law at changing water boundaries is a very complex subject beyond the scope of this report and includes conditions like perceptibility of accretion, and the effect of the change (Lambden and de Rijcke 1996).

Issues to Consider

- Lack of clarity regarding ownership and agency responsibility for regulating newly exposed lands.
- Need to instil in all public users a sense of responsibility to adjoining lands, lake, wildlife, and people.
- Need to regulate ATV and dirt bike users.

8.3.3.9 Minimize Costs

Why? *Pro-active adaptation to climate change can avoid many future costs by minimizing injury and property damage, reducing eventual protection costs, and maximizing continued ecological health of coastal resources.*

The Provincial Policy Statement (Ontario Ministry of Municipal Affairs and Housing 2005b) requires development be directed outside hazardous lands adjacent to the shorelines of the Great Lakes-St. Lawrence River System impacted by flooding, erosion, and/or dynamic beach hazards. This preventative approach protects public health and safety while supporting financial well-being over the long-term by minimizing cost, risk, and social disruption.

Coordinated cross-jurisdictional planning can also save money. Although planning is largely a municipal responsibility, the Provincial Policy Statement requires coordination for effective planning of cross-boundary features. Shorelines are one such feature. Any short-term funding constraints that curtail coordination within and among Conservation Authorities or municipalities ultimately deprive those regional communities of long-term savings.

Conservation Authorities' Shoreline Management Plans completed in response to high water levels have no formal authority but are intended as input to municipal planning documents (e.g. Official Plans, comprehensive zoning by-laws, Secondary Plans) and as assistance to other agencies dealing with shoreline issues (Ausable-Bayfield Conservation Authority 2000).

Issues to Consider

- Short memories for and lack of public awareness of lake level changes.
- Need to consider issues in light of future generations and of long-term costs and benefits.
- Need to understand the huge cost efficiencies of prevention over repair.
- Need to understand and raise awareness of the costs and benefits of different adaptation strategies.

8.3.3.10 Provide an Implementation Structure

Why? *Implementation of adaptive measures is strongly dependent on institutional capacity – financial, human resources, and political will. The 1990s cuts to natural resource agencies like the Ministry of Natural Resources and Conservation Authorities together with public cynicism about government have eroded the capacity of existing institutions. Further, effective responses to water and habitat issues have often been hampered by overlapping or competing jurisdictions as well as lack of inter-agency cooperation and interdisciplinary integration (Bruce et al. 2000). Citizen involvement is a growing necessity and an implementation challenge.*

Monitoring measures progress and provides the basis for program modification (Ramsar Convention of Wetlands 2002). Coastal areas including wetlands can be monitored directly; since basin processes affect wetlands, basin monitoring can also be helpful.

Structure

The Planning Reform documents (Ontario Ministry of Municipal Affairs and Housing 2004) called for a coordinated approach among municipalities for ecosystem, shoreline, and watershed-related issues to optimize the long-term availability and use of land, resources, infrastructure, and public service utilities. The many functions of coastal wetlands (e.g. nutrient cycling, sediment retention) and of coastal hazard lands (e.g. storm protection) could justify their inclusion under coordination of public infrastructure as well as of natural resources.

The coordination imperative has emerged in the 2005 Provincial Policy Statement, the Greenbelt Plan (Ontario Ministry of Municipal Affairs and Housing 2005a,b) and Places to Grow (Ontario Ministry of Public Infrastructure Renewal 2004). So while no structure or proposal has appeared which is specific to the complete Great Lakes coast, recent planning initiatives offer support and models. Some structure components and examples are:

- **Policy**: Examples include *Niagara Escarpment Act* and Plan, *Oak Ridges Moraine Conservation Act* 2001 and Oak Ridges Moraine Conservation Plan. The coast could be subject to a similar plan or be a component of a broader green corridor legislation that includes Niagara, Oak Ridges Moraine, other greenbelts, as well as coastal zones. Other options for coastal policy could be an area-specific provincial policy statement under the *Planning Act*, regulations for Conservation Authorities, or guidelines like the Foodland Guidelines.
- **Administration**: Possible structures include: a designated commission; a provincial ministry like Natural Resources; municipalities and Conservation Authorities with a coordinating body like the Waterfront Regeneration Trust, Coastal Stewardship Council, Areas of Concern Remedial Action Plan committees; or even a non-government organization to arrange easements like the Bruce Trail Association.
- **Incentives**: Shore owner cooperation could be encouraged through programs offering assistance for water access, and recognition – including financial compensation – of ecological services their land is providing.

Monitoring

The Marsh Monitoring Program (MMP) is a long-term volunteer-based program coordinated by Birds Studies Canada with funding from Environment Canada, the U.S. Great Lakes Protection Fund, and the U.S. Environmental Protection Agency. It has standard protocols to monitor marsh birds and amphibians, and applies to marshes throughout the Great Lakes basin including coastal ones. Launched in 1995, involvement of volunteers has helped its survival despite widespread trends to cut monitoring.

The State of the Lakes Ecosystem Conference (SOLEC) has proposed a suite of indicators to monitor and report on the health of the Great Lakes basin ecosystem, including coastal wetlands. Coastal wetland indicators include coastal wetland area by type, wetland-dependant bird diversity and abundance, and amphibian diversity and relative abundance. Other SOLEC indicators that relate to the impacts of climate change include ice duration on the Great Lakes, and water level fluctuations.

These and other coastal wetland indicators are being assessed and refined by the bi-national Great Lakes Coastal Wetlands Consortium funded by the Environmental Protection Agency. Plans call for the development of a bi-national, multi-jurisdictional long-term monitoring program for Great Lakes coastal wetlands (Environment Canada 2003).

The OMNR is developing the Southern Ontario Land Resource Information System (SOLRIS), a natural resource and land use database for south of the Canadian Shield. It will use satellite imagery to track change and create an inventory based on the Ecological Land Classification (Environment Canada 2003).

A number of Conservation Authorities are developing Watershed Report Cards or State of the Watershed Reports that could have a wetland component and shed light on processes affecting their shoreline wetlands.

Issues to Consider

- Need for a lead agency to be responsible for shoreline protection and development.
- Lack of funding for current shoreline agencies.
- Need for more monitoring in such a dynamic, climate change-influenced zone when monitoring budgets are being cut.
- Need for monitoring results to be accessible and publicized regularly.
- Need to involve many stakeholders for broad ownership of coastal zone concept.
- Need for valid but simple indicators suitable for volunteer participation.

8.3.4 Opportunities

Although climate change has extremely serious global implications, Ontario has a number of factors that offer opportunities for coastal land use adaptation.

- Water level decreases create new land that can increase protection of near-shore investment, in contrast to the loss of land and decreased protection on ocean coasts subject to sea level rise.
- In intensively developed southern Ontario, new land presents the unique opportunity for a natural corridor that can offer many ecological services without displacing existing uses.
- Great Lakes coastal wetlands require water level fluctuations to maintain species and habitat diversity. Compared to many ecosystems, this adaptation may offer coastal wetlands more resiliency to the larger water level changes that climate change may bring.
- Recent Ontario government planning reforms lend support to land-use planning adaptation options, and to protection and restoration of ecological features, functions, and systems.
- Ontario has several examples of collaborative ecosystem-driven planning involving experts and an active public (e.g. Oak Ridges Moraine, Living Legacy).
- Unlike many other Great Lakes issues such as fisheries and water pollution, effective action on coastal wetlands need not await American cooperation nor integration with eight states to be effective. Indeed, local action in one jurisdiction can spark interest in others.

8.3.5 Recommendations

The main recommendation resulting from the review of planning needs and the current situation is development of a natural coastal corridor. Ideas are presented related to its design, implementation, administration, and benefits. Additional recommendations applicable to the Ten Planning Criteria are made. Given the preliminary nature of some of the concepts and the numerous actors potentially involved, responsibility for implementation of the recommendations is not assigned.

8.3.5.1 Main Recommendation: Development of a Coastal Corridor

A Coastal Corridor could encourage coastal adaptation to climate change through natural processes and protection of property while enhancing public access, at low long-term cost.

Suggestions for design

The designated belt of land could begin at the 2006 extent of hazardous land and wetlands, according to Conservation Authority specifications. The 2006 date refers to the deadline for Conservation Authorities to conduct shoreline mapping to comply with the new generic regulation. Rather than the current situation of the upland boundary being fluid, shifting lakeward with climate change as hazards shift lakeward, it is recommended that the upland boundary be permanent.

As lake levels decrease and new land is exposed, the protected coastal zone would add to its original extent to include the new land. The protected Coastal Corridor would become widest where nearshore slopes are shallow and substrates are hospitable to wetland plants – conditions most conducive to wetland migration. Shoreline regions with a steep nearshore slope would likely experience little or no newly exposed area nor Coastal Corridor expansion lakeward. Existing coastal wetland boundaries and adjacent lands would remain protected, persisting as natural area whether or not the wetland shifts lakeward. If a wetland was not able to shift, this protection would ultimately create an upland ecosystem along the shoreline.

Wetlands interact with adjoining landscapes. In coastal regions, biota migrate among wetlands, and the wetlands themselves are dependent on shoreline sediment transport processes. It is proposed that the Coastal Corridor could have sub-zones, which could include:

- **Core natural areas** that would include wetlands and their adjacent land as well as other significant natural coastal systems like beaches and dunes where few activities beyond approved access and restoration would be allowed;
- **Buffer zones** along the shore on either side of core areas that, for wetlands, could be available and expandable if the wetland shifts in that direction and would include barrier sediment source areas; and
- **Coastal transition zones** that would encompass a less restrictive but still protected and regulated area corresponding to all the remaining current shoreline hazard land plus newly exposed land not in the other two zones. Permits might be considered for very restricted beach clearing or dredged boat access.

The proposed use limitations, except for the wetlands themselves, are stricter than current limitations on coastal hazard lands. Now, outside wetlands, Conservation Authorities can approve development if it does not affect hazard control (McColl 2005). The proposed limitations would incorporate protection of potential connectivity functions, a direction supported by the 2005 Provincial Policy Statement (Clause 2.1.2).

Such a 3-zone classification is compatible with the UNESCO system for Biosphere Reserves and has been proposed by the Government of New Brunswick to protect the province's coast from storm surges while maintaining access (New Brunswick Department of Environment and Local Government 2002). The classification also meets Ontario's recent planning criterion that the functionality of the ecosystem should define the plan, exemplified in the Oak Ridges Moraine Conservation Plan (Miller 2004). It could help planning progress from patch protection to a systems approach, towards planning natural infrastructure as other critical infrastructure is planned (Pearson 2004). It is a very low risk measure, linking highly fragmented wetlands and affecting little land if climate change effects do not occur. It would meet the criteria for a good adaptive system – minimizing negative effects of the unexpected while maximizing opportunity (Holling 1978 in Smith *et al.* 1998).

In southern Ontario, establishment of a corridor on the coast would be much easier than inland. There are minimal land use conflicts because, for currently exposed shoreline land within the hazard lines, the use is already restricted; for land not yet exposed by receding lakes, the pre-emptive designation would prevent most conflict from ever arising. Given southern Ontario's need for natural corridors but the intense competition for land that will only intensify as the population grows, this Coastal Corridor could be a unique opportunity: low-cost, low-conflict, and extremely valuable ecologically. If established soon, when owners have nothing to lose, it may overcome the public's difficulty to think a few decades hence. As a linear feature, monitoring would be straightforward – involving one flight line.

The ownership of newly exposed land is currently unclear. If it should prove to belong to the current shoreline owner, a protection option could be to consider a downslope rolling easement, i.e. the government buy a very low cost easement that allows the full development of the property subject to current or 2006 limits but no further lakeward as levels go down. This would involve no loss of rights, simply a small payment not to extend them. Some provisions may be necessary for boat channels, water access paths, and lake views for existing development with the possibility of stricter limits or joint access arrangements for future development. As climate change occurs, return sales of easement rights to shoreline owners could be considered where critical ecosystem and processes can be shown to be lacking and are unlikely to regenerate and the Coastal Corridor conditions are met. An easement would also raise awareness of the value of a coastal corridor and improve the understanding of it as an important form of public infrastructure.

Additional features of the proposed Coastal Corridor might include:

- New public trails to allow shoreline access but avoid core natural area impacts;
- Linkages to river systems and inland natural heritage plans;
- Incorporation into provincial policy, regional and local land use plans, and disaster avoidance plans responding to the possibilities of more intense storms; and
- Provisions for local differences in shoreline characteristics. For example:
  - Canadian Shield coastal areas are often less degraded, have less extensive wetland development, and a smaller potential Coastal Corridor area because of steep slopes. The Canadian Shield geographic boundary is already incorporated into the Provincial Policy Statement and could also be used in the current proposal to define guidelines specific to this area;
    The Oliphant area on the western side of the Bruce Peninsula has very large expanses of exposed land and interspersed islands as well as unique shoreline fens at least partially dependent on groundwater and therefore need controls on development inland of the coastal hazard land boundaries;
  - Eastern Lake St. Clair has extensive dyked marshes and dyked farmland where special consideration may be required to prevent agricultural encroachment into newly exposed areas and to provide joint management of new marshes with existing dyked ones; and
  - Urban areas have hardened shorelines and harbour requirements.

Suggestions for implementation and administration

Effective implementation of any type of large-scale habitat conservation mechanism depends on:

a) Development of a broad coalition, including experts but also coastal stakeholders and concerned non-governmental groups;
b) Consideration of the action as an economic opportunity or boost to long-term economic well-being of the coastal zone, i.e. as valuable natural infrastructure offering critical services and damage protection;
c) The existence of innovative policy entrepreneurship, i.e. some government officials adept at building coalitions around innovative ideas; and
d) Ecosystem functionality as the key principle (Rabe 2002; Miller 2004; Whitelaw 2004).

Key Partners to Administration and Implementation

- Federal Government
- Provincial Government
- Conservation Authorities
- Municipalities
- First Nations
- Non-governmental organizations
- Public representation
- Land owners
- Universities

As part of the Living Legacy, the Ontario government created the Great Lakes Heritage Coast extending over 2,900 km above the Severn River in Georgian Bay. Its establishment is still in its formative stages but it does provide a framework to consider for administering an all-lakes Coastal Corridor. Administration of a Coastal Corridor will require significant study; the extent of the area in question is large. The Great Lakes coastline (Canadian and American) length is equal to almost 45% of the circumference of the Earth (Ontario Ministry of the Environment and Energy, no date).

Both a lead organization and strong partnerships of various coastal stakeholders are essential. Successful structures for corridor administration include the Lake Ontario Waterfront Regeneration Trust and the Niagara Escarpment Commission. In addition, an independent advisory board could be effective. Members would be experts but for their term would contribute as concerned citizens rather than as agency representatives. The International Joint Commission advisory board experience has shown this model elicits high quality advice and effective consensus (Dempsey 2004).

More Coastal Corridor Related Recommendations

- Clarify jurisdiction and ownership of newly exposed lands.
- Increase funding to Conservation Authorities to cover additional shoreline responsibilities.
- Consider extending jurisdiction of existing Conservation Authorities over the shoreline gaps and islands south of the Shield.
- Revisit Shoreline Management Plans and shoreline regulations in light of possible effects of climate change.
- Consider whether climate change and associated storms will alter wave and ice setbacks when setting shoreline setback boundaries.
- Complete cost/benefit analyses of coastal adaptations including the Coastal Corridor.
- Dedicate selected revenue sources to land securement or easements (Reid 2001).

Additional Benefits

Beyond coastal wetland protection, a Coastal Corridor could provide other benefits.

- Contributions to **climate change mitigation**:
    - Support for sustainable transit if trails are included for walking and bicycling;
    - New recreation options close to millions of people, minimizing travel; and
    - Provision of wind-rich shoreline sites (in the proposed Coastal Transition Zone) for wind-power generation as lower lakes reduce hydro power and fossil fuel access declines.
- Support for **other corridor and natural heritage network initiatives**:
    - Complement the Niagara Escarpment, Oak Ridges Moraine, and Algonquin to Adirondack corridors, enhancing the value of each;
    - Addition of a shore-related layer to the inland-focussed Big Picture;
    - Addition of much-needed corridors in areas where natural land cover is very low and existing land use makes restoration of inland corridors challenging, e.g. Lake St. Clair counties where flat topography could offer especially wide coastal zones; and
    - If appropriately managed and allowed to naturalize, a possible control mechanism for invasives that thrive in human disturbed areas (McNicol and Russell 2004).
- **Urban options**: the possibility of rejuvenating the waterfront at relatively low cost by providing ecological support systems, recreation, avoidance of hazard costs, lake access, and lake buffering.
- **Storms**: the corridor would provide protection from extreme storms, for which the risks may increase as climate changes.
- **Public acceptance**: increased public access to the shoreline would encourage growth of a constituency supporting shore protection and on-going implementation and management of the Coastal Corridor.
- **Financial benefits**: major government savings in disaster relief and boost to the insurance industry viability as development avoids the zone of risk from climate change, extreme storms, and intense overland runoff. The set boundary would relieve the Conservation Authorities of the costs of re-mapping hazard lands as lake levels recede. The highly significant natural corridor would be attained at relatively little cost within southern Ontario's context of high and rising land prices.

The Coastal Corridor could contribute significantly to all ten planning criteria with the sole exception of the fourth criteria to Manage Watersheds to Anticipate Upstream Stressors.

8.3.5.2 Other Suggestions specific to the Ten Planning Criteria

Protect existing wetlands

- Use bigger tax breaks and/or payments for conservation easements to reflect the value of ecological services such as protection against more intense storms, filters of upstream eroded sediment to help protect lakes, carbon uptake, and fish nursery.
- Strengthen and enforce rules against ballast dumping and exotic introductions. Control of invasives is necessary both to relieve current wetland stressors for greater resiliency to climate change but also because climate change will encourage the spread of invasives (McNicol and Russell 2004).
- Identify and address any policies that work against maintenance and creation of wetlands.

Expand wetland protection lakeward over time

- Investigate instances elsewhere in the world of water level decreases, reviewing effects and adaptations for ideas applicable to Great Lakes coastal wetlands.

Prepare for changes in shoreline, storms and ice

- Consider tax breaks or payments for shoreline landowners who naturalize shores even if no wetland area is present. Shore land provides storm protection, ecological linkage, recreation trail options, water quality buffer, space for wind generators.
- Limit hardened shorelines that eliminate shoreline sediment processes.

Manage watersheds to mitigate upstream stressors

- Expand rural water quality programs including buffer development and soil conservation techniques; pay farmers to naturalize marginal, highly erodible and riparian land (see the Conservation Reserve Program in Appendix 8.3); provide bigger tax breaks for natural areas under Farm Land Taxation Program; support Environmental Farm Plan (Reid 2001).
- Discourage out-of-watershed water transfers.
- Reduce urban sprawl and impervious surfaces.

Connect coastal wetlands to the greater coastal landscape

- Establish refuges, parks, and reserves along corridors, especially north/south shorelines.
- Link coastal corridors to watershed riparian corridors.

Raise public awareness

- Regularly publicize monitoring results in highly accessible formats.
- Publish success stories of local communities' adaptation and mitigation actions.
- Move and entertain the public about the Great Lakes coast – its beauty, balance of nature, identity, history.
- Involve First Nations who bring traditional ecological understanding.
- Expand opportunities for the public to experience the coast, including school outings, additional parks, numerous public access points, shoreline trails.
- Involve the public in shoreline planning activities.
- Re-think terminology. Does use of the term natural "heritage" in planning policy suggest value as a relic and contribute to the lack of awareness of the ecological services that natural areas provide? Does "environment" contribute to the damaging perception that humans are separate from our "ecosystem" rather than part of it?

Involve shoreline owners and users

- Raise awareness among potential investors in lake-associated recreational facilities regarding possible climate change effects and any Coastal Corridor restrictions.
- To avoid beach bulldozing, assist with portable docks and public access routes.
- Discourage permanent docks, assist with localized clearing for community beaches, raise awareness of the value of expanding wetlands including for shore property protection, consider compensation for the wetlands' services to society.
- Recognize and address concerns about Lyme's disease, West Nile virus, and other possible diseases including investigation of methods to control problem species and vector populations.
- Provide regulator staff funding adequate for fast reviews of permits.

Increase public access to the shoreline

- Address the issue of motorized vehicles like ATVs and dirt bikes. Consider licensing and use in restricted areas only.
- Seek alternatives to private cottage sprawl. Consider options such as cottage clusters with controlled beach access.

Minimize costs

- Involve the insurance industry since as it suffers more and more losses from climate change effects, it is likely that insurance rates will rise and insurance availability will be restricted (Berz 1999).

- Study costs and benefits of various adaptation options and policies.
- Map shoreline and wetland bathymetry and substrate to rate wetland, coastal zone, and water access opportunities as levels recede.
- For effective responses to water level change, develop an understanding of the relative effects of isostatic rebound, channel dredging, diversions, in-basin water consumption, and climate change on water levels throughout basin.

Provide implementation structures/monitoring

- Identify and target information needs of decision-makers and make results accessible.
- Monitor wetland response to lower levels in various wetland types and management regimes.
- To inform adaptive management, monitor wetland response to higher temperatures, both of water and air.
- Continue to support volunteer monitoring programs such as the Marsh Monitoring Program.

8.3.6 Summary

Over the next century, water level decreases induced by climate change will affect the extent and value of Great Lakes coastal wetlands. Their natural ability to adapt will depend on shoreline slope, substrate, the rate and magnitude of water level changes, and the lack of adverse human interference. In Ontario, these ecological changes will occur simultaneously with enormous population growth. Proactive mechanisms such as innovative development and implementation of land use policy instruments could respect natural processes of wetland adaptation as well as the needs for recreation access and restored natural connectivity while minimizing ecological, social, and financial costs. A Coastal Corridor is one possible instrument.

The Ten Planning Criteria, the Coastal Corridor concept, and additional recommendations are preliminary concepts intended to launch discussion among stakeholders with expertise and interest in climate change impacts, coastal wetlands, and land use planning.

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9.0 FINAL SYNOPSIS

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Climate is a key determinant of the distribution, productivity, and functioning of wetland ecosystems. As such, wetland vegetation and wetland-dependent bird and fish communities are expected to be impacted by projected changes in climate acting in concert with other wetland stressors. Through an integrated assessment approach and using water level change as a surrogate for climate change, this project explored:

- the projected natural changes in Great Lakes coastal wetland ecosystems (wetland vegetation and associated wetland-dependent birds and fish) to identify any potential vulnerabilities, and
- infrastructure adaptation and management policies and strategies that may be used to maintain ecosystem function and values.

For the Great Lakes region, climate change projections based on global climate models indicate warming temperatures in all seasons to varying degrees and increasing annual precipitation by the year 2050. Winter and spring warming will shift a proportion of winter precipitation from snowfall to rain resulting in reduced snowcover and earlier spring melt. Great Lakes water levels reflect a critical balance between the timing and amount of water supply and water loss, including evaporation. Historic variation in water level records demonstrates the impact of small changes to this water budget on lake levels. Projecting 50 years into the future, Great Lakes impacts from changes in climate include warmer water temperatures and lower water levels occurring with increased frequency and duration. Shifts in the seasonal pattern of water levels are also projected, as runoff increases in the winter and snowmelt occurs earlier. These hydrologic alterations occur at several temporal scales, and can impact wetland ecosystem functions in a variety of ways.

Wetlands are located at the land-water interface, which are dynamic environments along the Great Lakes shoreline. By their very nature, wetlands are continually responding to changing water level regimes, and require hydrologic variability to maintain diversity and function. A long-term significant decrease in water levels would result in significant changes to the current distribution and abundance of wetlands. Several variables will influence the ability of current wetland communities to naturally respond to lower water levels. Nearshore bathymetry, coastal processes, geomorphology, and other human-related stresses and alterations will interact and affect the natural succession process.

In addition to site-specific and landscape level influences on wetland responses, species-specific life history traits will also govern how higher level communities (e.g. birds and fishes) within a wetland respond to the lowering of water levels. The ecological plasticity, or the ability of an organism to respond to changes in the environment, varies greatly for coastal wetland species. It is expected that some species will readily adapt their behaviour to hydrologic changes, while others with narrow hydrologic tolerances and limited reproductive capacity are at high risk for negative impact or greater natural selection pressure. Similar to other types of wetland stresses, these hydrologic stresses will likely result in reductions to the distribution and abundance of rare, specialist species and the expansion of generalist and invasive species. Potential outcomes include a reduction in species diversity and biological integrity.

Wetland community modelling within this study has confirmed that projected water level reductions, under some climate change scenarios, can have a considerable impact on the current distribution and abundance of wetland vegetation, bird, and fish communities. In particular, scenarios indicate that major shifts are likely as long-term average water levels decline. Model results for all wetland communities indicate that coastal wetlands have the potential to naturally shift in response to water level changes projected under climate change scenarios. A decrease in water levels favours the expansion of drier vegetation types, like treed/shrub and meadow marsh vegetation. Although modelling was limited to a small number of sites in the lower Great Lakes, the results indicate that hydrogeomorphology will influence the extent and type of vegetation that develops in response to water level decreases.

Lacustrine, protected wetlands appear the most resilient to climate change where drier vegetation types, such as treed/shrub and meadow marsh vegetation, are capable of expanding into emergent vegetation areas. The slow migration of emergent and other aquatic vegetation lakeward is also possible in these areas due to suitable growing conditions at lower elevations (i.e. protection from wind and waves, suitable soils, slope). Conversely, riverine, drowned river-mouth wetlands typically have limited suitable growing conditions downslope of current floodplains. Expansion of emergent and aquatic vegetation within these wetlands will be limited to the river channel and potentially within deltaic features should shoreline conditions allow for these features to be formed as deposition of riverine sediment loads shift from the existing floodplain lakeward as water levels decline.

Great Lakes coastal wetland bird and fish communities have the ability to move in response to potential changes in vegetation community distribution. Guild-specific modelling and species-specific vulnerability assessments indicate that certain bird and fish guilds will be impacted to varying degrees by changes in the timing, duration, and depth of flooding within specific vegetation communities. In particular, over-water nesting bird species and spring spawning warmwater fish species, especially their nursery habitats, that require flooded marsh vegetation for reproduction are most affected.

In this evaluation, human-directed adaptations to climate change in coastal wetland areas can take several forms. Wetland modelling and stakeholder input indicate that land use planning and policy actions are necessary to protect natural processes of wetland succession and function and should be a top priority in the suite of potential adaptation strategies. Current land use planning and policy in southern Ontario does provide protection to existing wetlands; however, proactive mechanisms are required to better incorporate potential future changes in wetland distributions and their relative importance to system function due to climate change.

Lake-wide water level regulation and preliminary evaluations of wetland dyking as potential human-directed adaptation strategies have been shown to sustain certain coastal wetland biotic communities and impact others. For example, emergent marsh is maintained but area of meadow marsh is decreased or removed. The faunal communities that are associated with these vegetation types respond directly, as will obligate nesting birds, but fish habitat suitability and population responses are dampened by interactions with other site-specific, physical features, and lag times in population response. Given the costly, labour-intensive, and long-term resource requirements of water level manipulation through engineered structures (e.g. dykes), a better understanding of multiple impacts on wetland function across wetland types is required prior to their use as climate change adaptation strategies. The wetland dyking analysis demonstrated the potential vulnerability of current dyke infrastructures to decreasing water levels and the need to consider future water level scenarios within proposed engineering designs. Stakeholders generally agreed that wetland dyking should only be a secondary consideration after the protection of natural processes, and should only be considered in special circumstances (e.g. protection of species at risk critical habitat) and as temporary measures under rapid loss.

Overall, the integrated assessment focused on understanding responses of wetland vegetation, bird, and fish communities and utilized literature review, stakeholder engagement, and modelling to characterize vulnerabilities and potential responses to climate change. This improved understanding of Great Lakes coastal wetland community responses to climate change-induced hydrologic alterations will hopefully support proactive wetland conservation planning. An important outcome would be the consideration of long-term variability and climate projections as well as sustainable human-directed adaptations into planning and decision-making processes. This will help ensure important Great Lakes coastal wetland functions are protected and allow natural succession to occur.

Recommendations for areas of research and investment required to further advance the scientific understanding of coastal wetland vulnerabilities to climate change and the application of adaptation strategies include the following:

- verification of wetland species vulnerabilities and their water level thresholds through experimentation and pattern analysis;
- improved understanding of the effects of substrate types and other environmental variables on vegetation colonization processes when water levels decline;
- fish and bird regional population assessments (monitoring and modelling) that include spatial and temporal evaluations at the scale at which the metapopulation functions;

- add temperature increases to water/land effects as thermal rises in some wetlands may mean coolwater, and some warmwater, fishes' upper limits are encountered;
- consistent field and remote sensing measurements (e.g. bathymetry, elevation, and substrate) in nearshore coastal areas at the land-water interface for improving current digital elevation models;
- completion of basin- or lake-scale, integrated assessments of climate change impacts;
- identification of important coastal wetland transitional areas and important functional processes for conservation and protection; and
- consideration of climate change projections and new scientific advice and findings in Great Lakes coastal wetland habitat conservation policy and planning to maintain healthy and diverse fish and wildlife communities.