Quantifying submerged aquatic vegetation using aerial photograph interpretation: Application in studies assessing fish habitat in freshwater ecosystems

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ABSTRACT: Use of aerial photograph interpretation (API) in resource inventory projects recently has increased, and this reflects benefits like established protocols, high spatial resolution, readily available photography, and limited cost. Application of API to quantify features of aquatic habitats used by fishes, like submerged aquatic vegetation (SAV), has been advocated for decades but a paucity of use suggests inadequate awareness of the methods. This article reviews a protocol that guides the use of API to quantify features of aquatic habitats, and then uses examples from contrasting habitats in the Lake Ontario watershed from 1972–2003 to illustrate this protocol. Even though we used photographs originally collected for other purposes, API identified the change in minimum area and depth distribution of SAV over time. These observations reinforce how API can contribute information to resource inventories, and why investigators should expand use of API in studies of aquatic ecosystems.

INTRODUCTION

Assessment and management of fishes requires knowledge of the spatial and temporal dynamics of aquatic habitats. Such knowledge can be obtained from field surveys or from use of remotely-sensed data to quantify current conditions. In combination with historical surveys, it is possible to achieve sufficient understanding of habitat dynamics to facilitate the management of the fish species and habitat(s) of interest. This strategy of linking habitats with fishes has been successfully used to assess resident or migratory species over spatial and/or temporal scales in freshwater (e.g., Ward and Ward 2004), estuary (e.g., Cooke et al. 2004), and marine (e.g., Coleman et al. 2004) environments. However, many investigators lack the financial or other resources to collect comprehensive field data on current habitat conditions and/or lack the historical surveys needed for suitable analyses. An alternative and lower cost approach for such analyses involves the use of aerial photograph interpretation (API) to quantify aquatic habitat features such as wetlands, emergent vegetation, and submerged aquatic vegetation (SAV) across spatial and temporal scales. Because API can be time-efficient and cost-effective, this methodology has been frequently advocated for habitat assessments (e.g., Orth 1983). Despite this support, API apparently is still infrequently used to assess fish habitat. For example, from 1993–2004, reports of API use in articles varied from 0–6 per year (mean=1.6) in the *North American Journal of Fisheries Management* and from 0–3 per year (mean=0.8) in *Transactions of the American Fisheries Society*; the highest number in both journals occurred during 2004. To promote the use of API, we review the protocol that guides the application to assess aquatic habitats. To illustrate this protocol, we use examples of the application of API to quantify SAV in contrasting habitats of the Lake Ontario watershed from 1972–2003. These examples detail the methods used to quantify and validate habitat assessments, and identify sources of photographs for API available in each county of the United States.

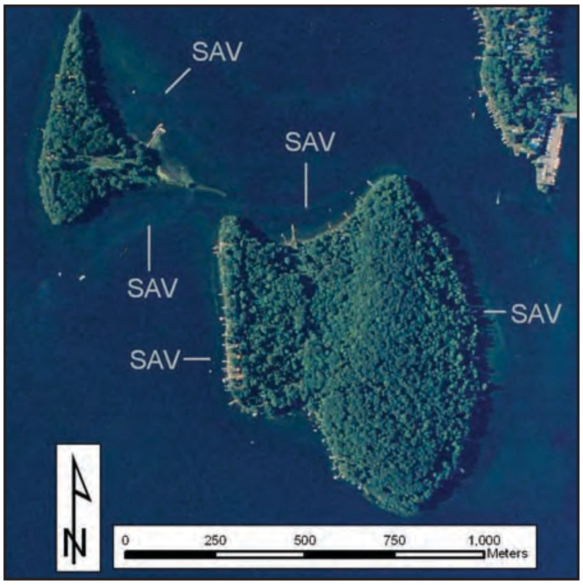
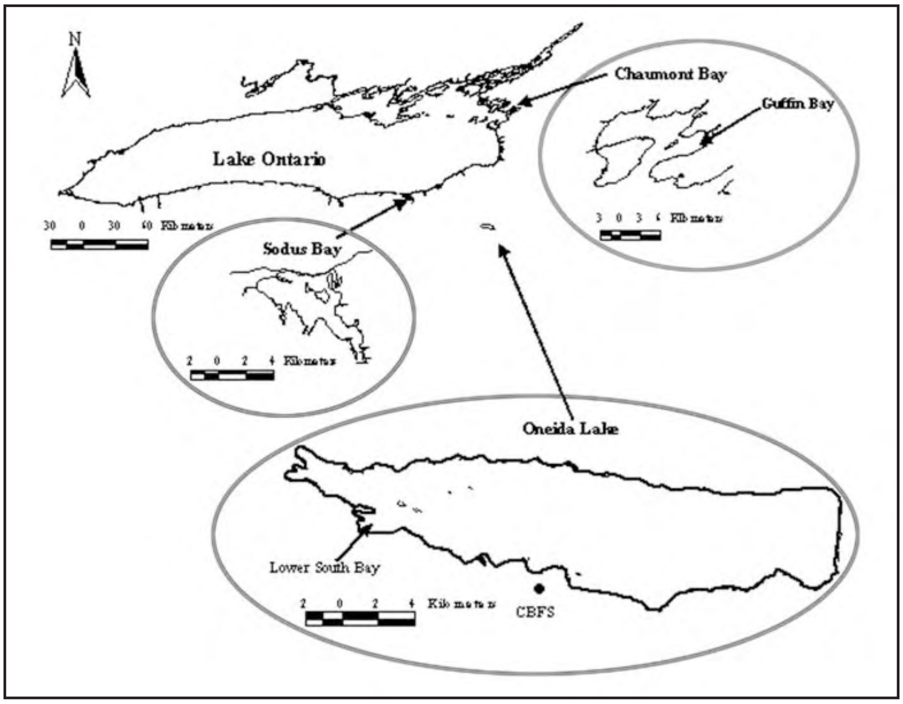


Figure 1. Representative aerial photograph of Sodus Bay in August 2001, obtained from the Wayne County Farm Service Agency of the U.S. Department of Agriculture. Examples of submerged aquatic vegetation (SAV) visible in the photograph are noted.

Figure 2. Map of study locations (Sodus and Chaumont Bays, and Oneida Lake) in the Lake Ontario watershed.



To date, API has been used to assess diverse features of aquatic habitats, including the distribution of plant species in wetlands, rivers, lakes, and estuaries (e.g., ASPRS 1997). In general, the interest in aquatic vegetation like SAV stems directly from the role they play across boundaries in ecosystems, and from the important ecosystem services these species perform. Macroscopic vegetation like SAV shapes aquatic primary production rates, acts to immobilize or ameliorate toxic chemicals, stabilizes substrates and therefore reduces turbidity, and provides a home for numerous organisms, including fish. In addition, SAV is also consumed by invertebrates and vertebrates ranging from fishes to muskrats to ducks, and is used as a physical habitat by invertebrates and vertebrates (Wetzel 1983; Covich et al. 1999; Håkanson and Boulion 2002). Thus, the assessment of SAV across spatial or temporal scales is warranted because it can offer unique insight on habitat condi-

Figure 3. Distribution of submerged aquatic vegetation (SAV) in Lower South Bay estimated with aerial photograph interpretation (API) for photographs collected during 2002 and the hydroacoustic transect survey during 2003. The SAV identified from the API is represented by the hatched area and the SAV along the hydroacoustic survey by the heavy line, showing the presence (green, >66.67% of area of transect) and absence (red, <33.33% of area of transect) of plants. As a complement, 6 rings each of 25-m width delimit the SAV estimated from API.

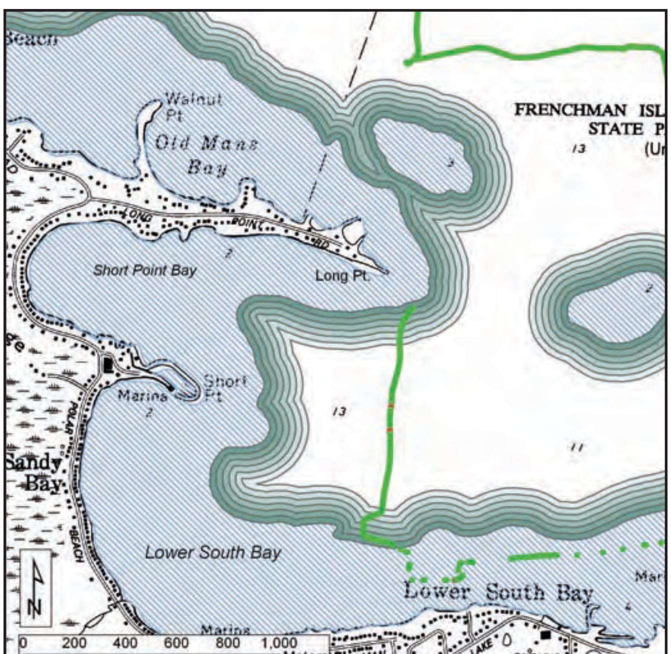
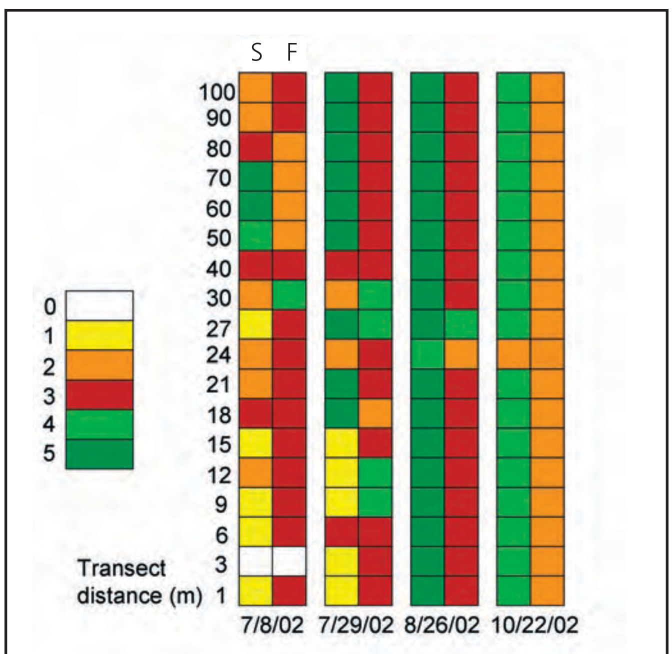


Figure 4. Schematic representation of the stem density of submerged aquatic vegetation (SAV) and filamentous algae (F) in Oneida Lake, along a 100-m transect from the shoreline of Lower South Bay, a habitat protected from heavy wave action. Observations start at the bottom of the figure, in 1 m^2 quadrats along the transect, measured by distance in meters from the shore. The SAV are represented by plant stem density (S) and F represented as total coverage on SAV. Observations are presented with the following six categories:

0: 0 stems in quadrat, and 0% coverage of F on SAV;
1: 1–10 stems, and 1-15 % area;
2: 11–25 stems, and 16-25% area;
3: 26–50 stems, and 26-50% area;
4: 51–75 stems, and 51-75% area;
5: >76 stems, and 76-100% area.



tions and food webs in aquatic ecosystems (e.g., Bettoli et al. 1991).

Use of API to assess aquatic vegetation has revealed wide-scale applicability of these techniques across habitats (e.g., bay of a lake, reach of a river, entire lake) to inventory geographic extent, depth distribution, functional groups, and species distributions, based on the signatures in the photographs being interpreted (ASPRS 1997). Frequently, large-scale inventories represent initial habitat assessments although they can be used to answer specific questions. For other questions, additional activities are typically required to confirm the findings and identify details. Options for conventional or

Figure 5. Same as Figure 3 except distribution of submerged aquatic vegetation (SAV) in Chaumont Bay for photographs collected during 2002 and hydroacoustic survey during 2002.

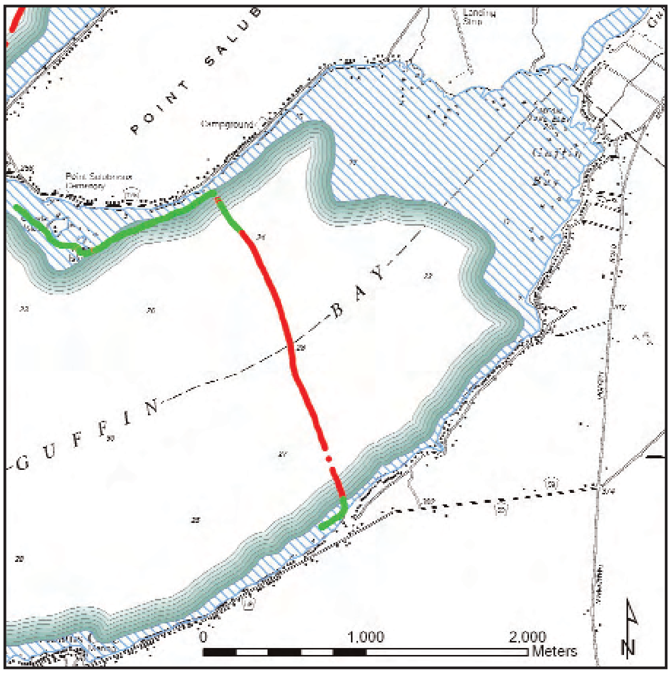


Figure 6. Same as Figure 3 except distribution of submerged aquatic vegetation (SAV) in Sodus Bay, Lake Ontario for photographs collected during 2002 and hydroacoustic survey during 2002. In addition, the distance from the shoreline to the bay edge of the SAV estimated from the API is represented by the arrow marked A and the distance between the API and bay edge estimated from the hydroacoustic survey represented by the arrow marked B.

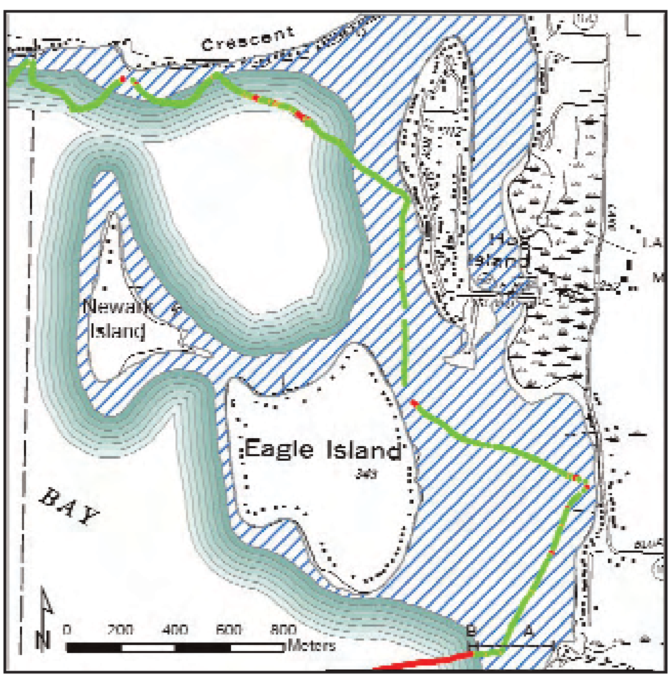


Figure 7. Distribution of submerged aquatic vegetation (SAV) in Sodus Bay, Lake Ontario, observed between 1972 and 2002, based on aerial photo interpretation and hydroacoustic surveys. The upper panel shows the distribution of SAV during 1972 (sage yellow) and 1980 (moss green) whereas the lower panel shows the distribution of SAV during 2000 (dark blue) and 2002 (light blue). The hydroacoustic survey of SAV during 2002 is represented by the heavy line, showing the presence (green, >66.67% of area of transect) and absence (red, <33.33% of area of transect).

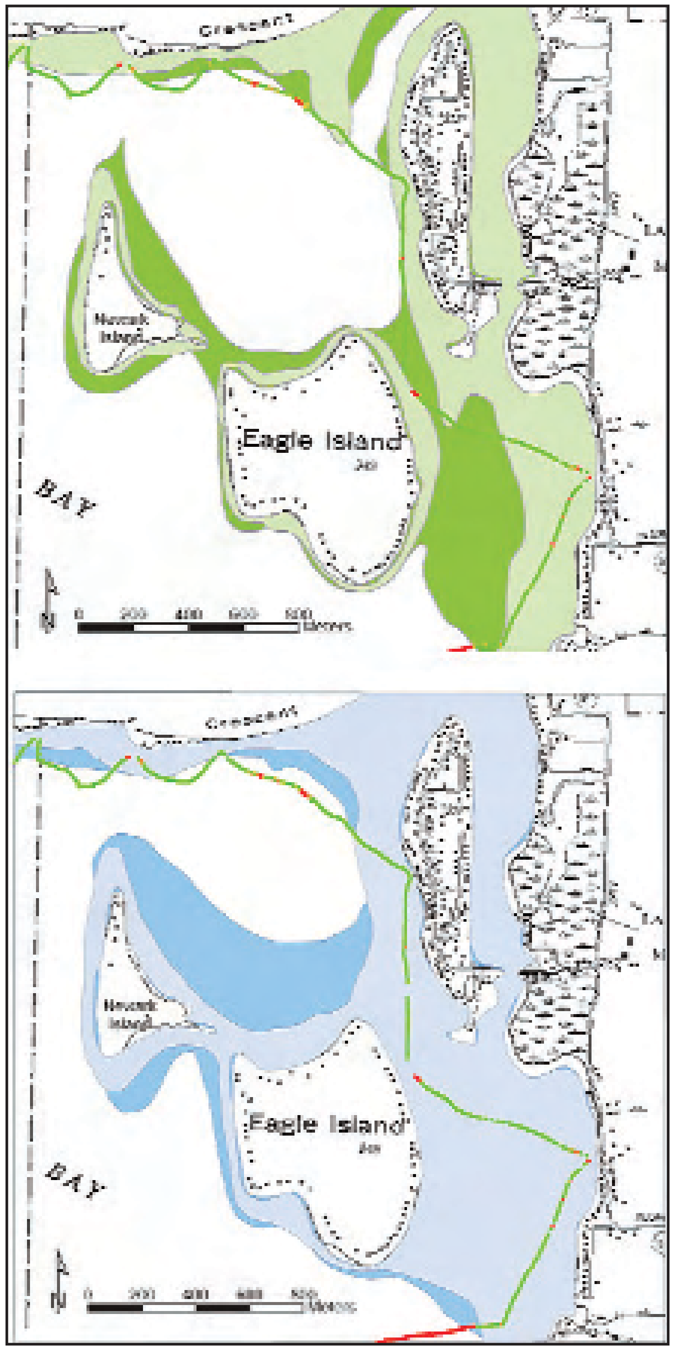


Figure 8. Distribution of depth of submerged aquatic vegetation (SAV) by depth along the south east corner of Sodus Bay, Lake Ontario estimated with a hydroacoustic survey during 2002. This portion of the hydroacoustic transect is noted by the A and B Figure 6. The vertical line identifies the bay edge of SAV identified from the API for photographs collected on 21 August 2002. The distance between the bay edge of SAV from the two methods was 25-m.

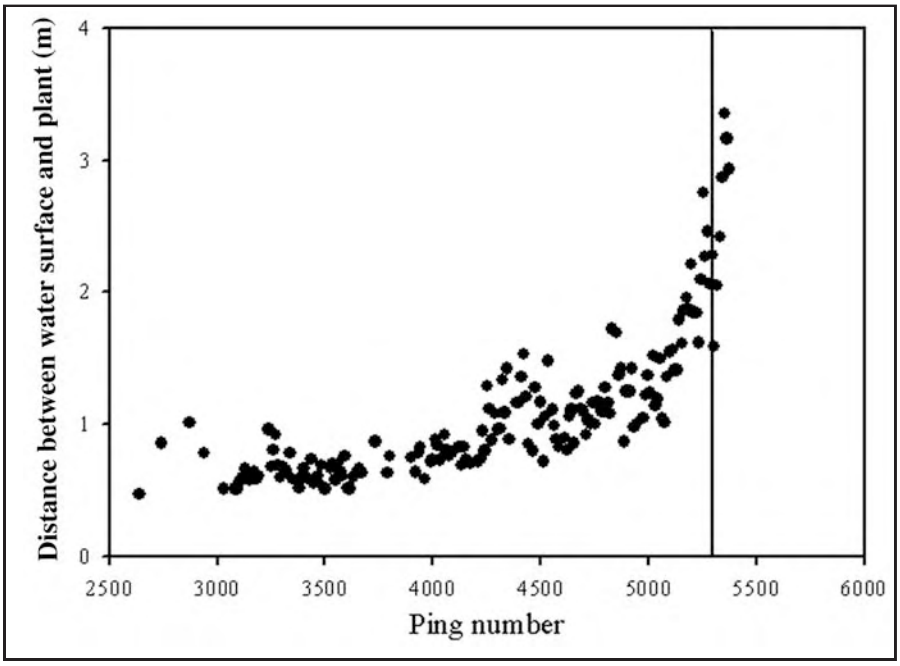
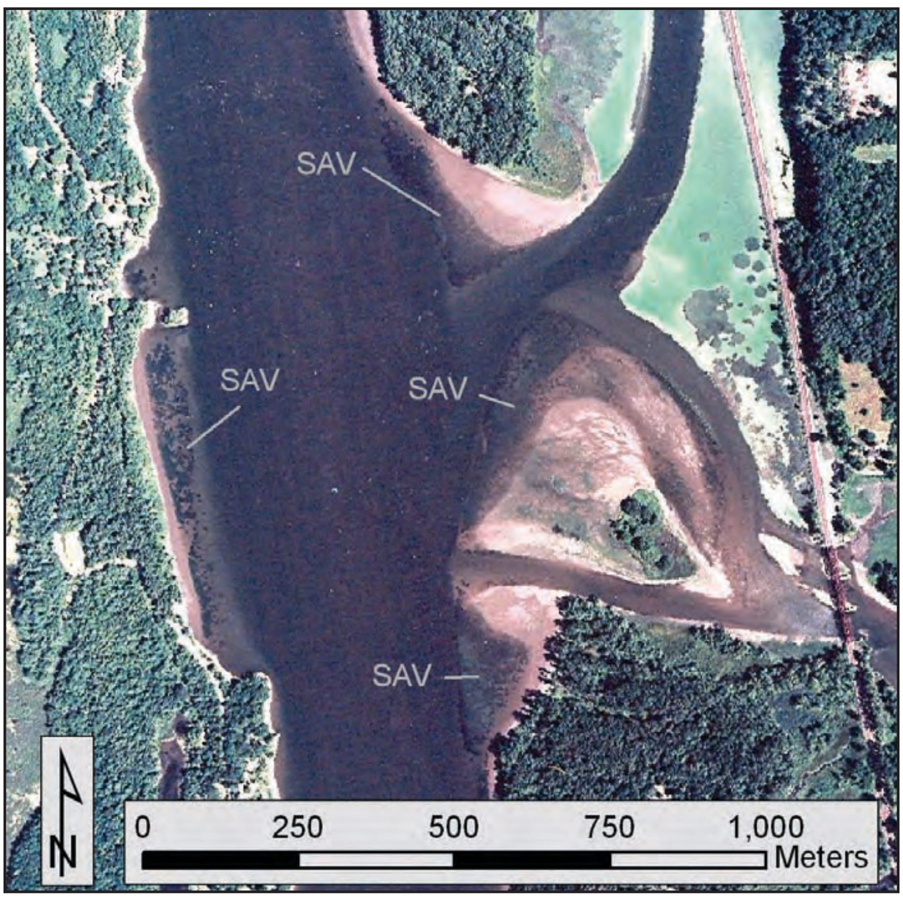


Figure 9. Representative aerial photograph of the Hudson River near Stockport Flats in August 2002, collected for Institute of Resource Information Sciences of Cornell University by Aerographics Inc., Bohemia, NY. Examples of submerged aquatic vegetation (SAV) visible in the photograph are noted, and river flows from the north to south.



digital image analysis and storage also make the application of API both feasible and flexible for use with different species across scales of investigation. However, all studies that use API require some field validation of the findings (ASPRS 1997).

OVERVIEW OF THE API METHOD

Background

Assessment of features on the landscape with API dates back to the mid-1800s, just after the advent of modern photographic methods. Early uses of API included the assessment of military activities during the American Civil War and the quantification of timber inventory in forests (ASPRS 1997). The basic tenets of the use of API in forestry are stated by Spurr (1948:3): "With the proper photographs, instruments, and techniques, the forester can obtain much information in less time, at a lower cost, and with greater precision than he could in the past." Over the last six decades, the API method has been applied to other arenas, including the assessment of habitat features across the land-water ecotone. This expansion is due, in part, to a greater appreciation of the benefits of API, like availability of relatively inexpensive photographs from archives, the ability to create thematic maps, and the opportunity to evaluate estimates of habitat features with repeated measures and/or complementary field investigations (ASPRS 1997). Lastly, the ease of converting photographs to a digital format can be used to create inexpensive storage options that concurrently simplify data exchange for collaboration or dissemination (e.g., Pavlidis 1988).

Process

The process of API is predicated on the availability of suitable aerial photography relevant to the question(s) of interest to the investigator. Guidelines for monitoring coastal land use and land cover with API are published as the Coastal-Change Analysis Program (C-CAP) of the National Oceanic and Atmospheric Administration (NOAA). These methods were developed to standardize the interpretation and analyses by different investigators in upland, wetland, water, and submerged habitats (Dobson et al. 1995; Finkbeiner et al. 2001; Tables 1 and

Table 1. Typical procedure required to complete regional coastal-change analysis program (C-CAP) classification, to quantify water and submerged land information, using API (Dobson et al. 1995; Finkbeiner et al. 2001).

Step 1.	Identify regional problem or changing habitat or species of interest. Area(s) of interest needs to be delineated, along with the spatial extent and frequency of assessment that can be completed with the available photography, and then identify classes of land-cover (see Table II) to be assessed.
Step 2.	Account for factors that may influence analysis. Methodological factors include spatial and temporal resolution, flight-line, and film. Environmental factors include atmospheric conditions, turbidity conditions in water, tidal stage, water surface conditions, and vegetation phenological cycle.
Step 3.	Interpret aerial photograph(s). Obtain appropriate map(s) and photography of area and time periods of interest. Preprocess photography if it will improve color balance, and then register photography with planimetric maps. Identify suitable change-detection approach for interpretation of photography. Complete image analysis, using monoscopic or stereoscopic methods, to delimit habitat polygons. Transfer habitat polygons to planimetric basemap. Digitize habitat polygons. Assess habitat polygons for change through time using GIS-based methods. Represent habit change using maps or spatial statistics.
Step 4.	Confirm accuracy and precision in data collection methods. Assess spatial data and statistical accuracy of habitat polygon boundaries.
Step 5.	Distribute findings. Digital and print products produced with comprehensive background information.

Table 2. Land-cover classification relevant to the C-CAP classification system of aquatic habitats (after Dobson et al. 1995).

1. Wetland
- Marine/estuarine rocky shores, composed of either bedrock or rubble.
- Marine/estuarine unconsolidated shores, beach or bar of mud, gravel, or sand.
- Marine/estuarine emergent wetland, including saline and brackish marshes.
- Estuarine woody wetland, classed as forest or scrub by type (deciduous, conifer, dead).
- Riverine unconsolidated shore, beach or bar of mud, gravel, or sand.
- Lacustrine unconsolidated shore, beach or bar of mud, gravel, or sand.
- Palustrine unconsolidated shore, beach or bar of mud, gravel, or sand.

2. Water and Submerged Land
- Water, as marine, estuarine, riverine, lacustrine (>20 acres), or palustrine (<20 acres).
- Marine/estuarine reef, formed by sedentary invertebrates.
- Marine/estuarine aquatic bed, containing rooted and floating plants, classed by salinity.
- Riverine aquatic bed, containing rooted and floating plants.
- Lacustrine aquatic bed, containing rooted and floating plants.
- Palustrine aquatic bed, containing rooted and floating plants.

2). The C-CAP protocol for mapping of submerged habitats includes technical specifications that guide photograph acquisition, and identify preferred film type, photograph scale, angle of the photograph relative to the landscape, and environmental conditions during acquisition. For example, acquisition should coincide with a high sun angle to minimize sun glare and shadows on the landscape, as they can obscure the identification of the landscape feature(s) in a photograph.

Other physical and photogrammetric factors shape the process of API, and will determine the feasibility and extent of potential analyses that can be completed relative to the question(s) of interest. These factors include the resolution of the species and/or habitat under study in the photograph(s), the season(s) of photograph collection, and the extent of geographic coverage required to fully assess a species in a habitat (Finkbeiner et al. 2001; Tables 1 and 2). The characteristics in a photograph will determine the suitability for interpretation, and shape the accuracy and precision of the measurements of interest. Ideally, photographs will show high resolution, clear patterns of texture in the landscape feature(s), fully represent shape of the feature(s), and contain minimal confounding environmental conditions like cloud cover. Also, localized or episodic events like rain can act to temporarily increase runoff and turbidity levels and obscure habitat features otherwise readily visible in a clear-water state. The visibility and resolution of the species or habitat will be dependent on the depth of penetration of light in the water column as captured by the film (e.g., Lillesand and Keifer 2000). In addition, color photographs are preferred but not essential for API. Collectively, the investigator considers these factors when previewing aerial photographs for suitability of interpretation (Finkbeiner et al. 2001). Such technical considerations need to then be integrated with the ecology and habitat used by a species, and this will dictate the minimum geographic zone that needs to be represented in the study. Also, if sampling extends across latitudes, the combined consequences of temperature and day length need to be considered, as they shape time when SAV maximum biomass and autumn senescence occurs (Spence 1982; Wetzel 1983), and should be considered in the study design.

Realistically, the extent of API completed for any study will be shaped, at least in part, by the cost per photograph; the economics of this methodology need to allow for sufficient analysis of the species and/or habitat, across both the temporal and spatial scales of interest.

Fortunately, the recent increased access of many private and public photograph collections (e.g., local, Internet-based) has reduced the cost of acquisition of existing photographs. Typically, single sets of photographs will not contain all elements required for a study, and multiple sources will need to be assessed to determine the feasibility of using the API method. Further, the assessment of a species typically requires the overlap of adjacent photographs across a focal or entire habitat zone. For example, API of low-elevation photographs could be used to map largemouth bass (*Micropterus salmoides*) nests in select SAV beds while lake-wide analyses could involve all nests and SAV beds. Such analyses completed across scales could be used along with traditional approaches to identify the habitat feature(s) shaping SAV beds, nest location, and/or vegetation-fish abundance relationships (e.g., chapters in Philipp and Ridgway 2003). For example, API could have possibly been used by Pothoven et al. (1999) to more fully evaluate the spatial relationships in small lakes between vegetation removal and fish habitat use. It is also possible to reveal more information in the field of view of the photographs if they are interpreted with stereoscopic techniques, a process that involves viewing two adjacent photographs simultaneously for a three-dimensional view. Identification of adequate aerial photography to complete a study from a suite of sources is analogous to locating the pieces of a puzzle and then resolving how they best fit together to create a coherent picture over the desired scale(s).

Varied sources of aerial photography exist (Table 3), and these extend across county (e.g., county planning departments), state (e.g., New York State Department of Transportation), and federal (e.g., NOAA's National Ocean Service) levels of government; photographs are available also from other independent sources (e.g., The Nature Conservancy). These photographs are often present in the public domain and can be accessed by any individual or group with relative ease. Typically, communication with the source is required to arrange access to preview photographs.

Table 3. Potential sources (county, state, federal, other) of aerial photographs for use during C-CAP classification of land cover in the U.S. (Dobson et al. 1995).

1. **County**
   - County planning departments
   - County real estate assessors
   - Farm service agencies
   - Local historical societies
   - Environmental management councils
   - Soil Water Conservation Districts
2. **State**
   - Departments of Environment Quality and Natural Resources
   - Department of Transportation
   - State Geological Survey
   - State GIS Clearinghouse
   - State land-grant college campuses
3. **Federal**
   - Environmental Protection Agency
   - National Aerial Photography Program
   - National Archives and Records Administration
   - National High Altitude Photography
   - National Imagery Mapping Agency
   - National Oceanic and Atmospheric Administration (NOAA)
   - National Ocean Service of NOAA
   - U.S. Army Corps. of Engineers
   - U.S. Department of Agriculture Aerial Photography Field Office
   - U.S. Fish and Wildlife Service
   - U.S. Forest Service
   - U.S. Geologic Survey
4. **Other**
   - City and regional planning departments
   - Local colleges and universities
   - Not-for-profit groups
   - Private and public aerial photography collections
   - Private industry with large land holdings (e.g., electric utility like National Grid)

FIELD VALIDATION

Use of API, like other remote-sensing methods, or image analysis in general, is commensurate with the need for complementary methods to validate and evaluate the accuracy and precision of the interpretation. The approach used to validate API will depend on the habitat and/or species under study. As expected, the C-CAP protocol offers specific guidance for field testing the findings from API (e.g., Finkbeiner et al. 2001; Table 1). Findings from API should be compared with in-field assessments, ideally within two calendar years, of aerial photograph acquisition. Thus, it is preferable to minimize the time between aerial photograph collection and field surveys, as landscape features can be dynamic but aquatic vegetation is typically present in the same place for years unless severe disturbance(s) occur (e.g., shoreline dredging). Environments like estuaries that are characterized by predictable cycles (and corresponding flux in water clarity) or reservoirs that undergo rapid hydraulic cycles, require additional considerations for assessment and validation of API compared with more static environments like lakes (Finkbeiner et al. 2001). For example, API of a reservoir will identify a SAV distribution that reflects current pool depth. To evaluate the relationships among fish, hydrology, and SAV in reservoirs (Durocher et al. 1984; Bettoli et al. 1993; Sammons et al. 1999), application of API may be required across seasons and pool depths.

Field assessment of the findings from API can use small- or large-scale surveys. Small-scale surveys include point-sample collections in SAV beds with rakes, quadrats, etc., or fixed-distance transects that assess substrates, SAV density, and species composition across different substrates and depths. Large-scale surveys can be done with hydroacoustic methods that rely on the characteristic features of echograms produced from the vegetation and substrates (e.g., Duarte 1987). Recently, automated algorithms for hydroacoustic data have been developed to facilitate efficient large-scale assessment of SAV coverage, distribution, and height (e.g., Sabol et al. 2002). Early analyses with hydroacoustics quantified the SAV spatial coverage, depth distribution, height in the water column, and substrate type by direct inspection of the acoustic echograms (Greenstreet et al. 1997; Sabol and Burczynski 1998). Complementary field surveys are used to validate assessments completed with remote-sensing techniques like hydroacoustics. With this method, it is possible to directly associate the landscape feature(s) with signatures visible in a photograph.

APPLICATIONS AND LIMITATIONS

The process of API must consider both the attributes evident in the aerial photography and the ecology of the species under study. Comparative studies have revealed the subtle differences evident in an image cannot be readily distinguished by automated methods

(Pavlidis 1988; Dobson et al. 1995; ASPRS 1997; Finkbeiner et al. 2001). For example, an analyst who completes API must identify the key indicators of color, pattern, shape, shadow, texture, and geographic location of the species or habitat under study (Lillesand and Keifer 2000; Finkbeiner et al. 2001). Finkbeiner et al. (2001:24) also recognized: "This type of mapping also requires experience at ground level in the study area since the photographic images of habitat features vary in ways that cannot readily be modeled, described, or communicated." It is for these reasons that many universities still offer hands-on courses with a focus on API methods.

The limitations of API are varied, as indicated in some of the earlier examples (e.g., costs for photograph acquisition, temporal changes in the aquatic habitat including water clarity, hydraulic cycles, etc.). Beyond these considerations, a major limitation of aquatic habitat assessment concerns large areas of water that lack points for geographic rectification (i.e., ground-control point references). Such geographic rectification is required to resolve the degree of error within the interpretations. One solution to this situation would be to add buoys set at known locations before acquiring the photographs. In addition, the precision is proportional to the scales of photographs used for API. Typical photograph range is from 1:10,000 to 1:48,000; a scale of 1:24,000 is considered a good balance between detection of landscape features and area coverage for aquatic habitats (Finkbeiner et al. 2001). With such scales, the thickness of a 12 pt. line (pencil width) on a map will represent ~15–30 m of SAV in the water. Thus, the characteristics of the particular photographs typically do not allow for discrimination of SAV species but can identify mono-generic or mono-specific patches of plants (e.g., Nieder et al. 2004). To detect change over time in the distribution of SAV with API, the magnitude of change must be greater than the inherent errors involved with both the photograph interpretation and map production.

If suitable photographs exist, API can facilitate intra-annual and inter-decade assessment of species or habitats across spatial and temporal scales. Retrospective analyses can provide information on habitat conditions by season, response characteristics to habitat modification, and large-scale patterns that may not be apparent from traditional small-scale field assessments. For example, API can be used to quantify the direct and indirect effects of natural processes like floods and anthropogenic processes like shoreline modification, on SAV distribution (e.g., depth, area) and abundance (Finkbeiner et al. 2001; Table 4). Strategies that integrate API with traditional habitat assessments continue to be advocated (e.g., McMahon et al. 1996).

Table 4. Topics that can be addressed with API for a species or habitat characteristics over spatial and temporal scales, if suitable photographs exist.

1. Chemical limnology (e.g., water clarity, chemistry).
2. Physical limnology (e.g., wave action, water-level regulation, temperature).
3. Biological limnology (e.g., species reintroduction, habitat rehabilitation)
4. Consequence(s) of nonnative species invasion or intentional species introduction.
5. Consequence(s) of climate change.

APPLICATION OF API TO ASSESSING FISH HABITAT CHANGE

Anthropogenic change in waters of the Great Lakes watershed

Documentation of the degradation of surface waters in North America due to anthropogenic activities received broad attention after World War II (e.g., Hasler 1947), and was well known in the Laurentian Great Lakes (e.g., Beeton 1969). This awareness led to government initiatives like the Great Lakes Water Quality Agreement (GLWQA) that were intended to improve chemical limnology (e.g., reduce nutrient loading), and promote native species restoration (IJC 1988). This management initiative increased water clarity and facilitated an expansion of benthic vegetation like SAV in lakes and rivers. Such a scenario was actually a predicted outcome of this program. Ryding and Rast (1969:81) state: "One must be careful, therefore, that control measures designed to reduce phytoplankton biomass do not inadvertently produce light conditions favorable for the excessive growth of phytobenthos." In other words, reduction of phosphorus (P) can promote a shift between a turbid state dominated by phytoplankton to a high water clarity state dominated by attached plants (e.g., Sheffer et al. 1993).

The GLWQA mandated the reduction of both point and non-point sources of pollution and had the expected effect of restoring habitats for native species across the watershed. For example, in Lake Erie, Ludsin et al. (2001) and Krieger et al. (1996) confirmed the recovery of native fish and invertebrates after this lake was previously characterized as the "Dead Sea of North America" (Sweeney 1993). But these reductions in pollution levels also have been associated with the unexpected effect of making habitats in the Great Lakes more vulnerable to invasion by nonnative species (Mills et al. 1994; MacIsaac et al. 2001; Holeck et al. 2004). In the last decades, one of the most pervasive nonnative species to invade the Great Lakes basin has been the filter-feeding dreissenid mussels (*Dreissena* sp.; Vanderploeg et al. 2002). Dreissenid mussels alter habitat by increasing structural complexity through the deposition of inorganic and organic materials at the local scale, and increase water clarity at the lake-wide scale (Vanderploeg et al. 2002). The synergistic consequences of nutrient reduction and dreissenid mussels in lakes act to increase water clarity and the importance of benthic food webs, a process termed "benthification" (Mills et al. 2003). Expansion of the photic zone and SAV to deeper habitats in aquatic ecosystems has created a need to assess these habitats to understand current use by fishes and invertebrates (Mills et al. 2003; Sheffer and Carpenter 2003).

Before the GLWQA, cultural eutrophication and intensive agriculture led to high P and sediment loading across the Lake Ontario watershed (Beeton 1969; Mills et al. 2003). The degradation of habitats, water clarity, and distribution of SAV was comparable for inland lakes like Oneida Lake and large bays like Sodus Bay in central New York state (Mills et al. 1978, 2003). For example, up to the early 1950s, Oneida Lake showed high water clarity (average annual Secchi depths >3.5 m) and SAV dominated habitats with suitable substrate, slope, and fetch (Mills et al. 1978). An increase in shoreline development during the late 1950s increased nutrient and sediment loads, and this rapidly reduced water clarity (mean <2.5 m, Mills et al. 1978). These conditions shifted the lake to a plankton-dominated state, reduced benthic primary production, and limited the SAV to patches in water <2-m deep (Mills et al. 1978; Idrisi et al. 2001; Mayer et al. 2002).

After the GLWQA, improvements were made to sewage treatment plants, farming practices, erosion management in riparian zones, and this collectively reduced the P and sediment loads to the Oneida Lake watershed. The reduction in P and sedimentation was followed by a gradual increase in water clarity (average Secchi >3.5 m), and lower standing algal biomass (e.g., chlorophyll *a*) per m^3 (Idrisi et al. 2001). Analyses revealed the >2-m increase in mean water clarity between 1976 and early 1990s still left a majority of the benthic plant community light limited (e.g., Idrisi et al. 2001). In response to the combined effects of GLWQA, and invasion of dreissenid mussels in 1991, the P and sediment loads to Oneida Lake have continued to decline, and resulted in water clarity resembling the period prior to cultural eutrophication (i.e., average Secchi >4.5 m). High water clarity has increased the biomass, diversity, and distribution of benthic algae and SAV (Mayer et al. 2002; Zhu et al. in press).

To illustrate how to use API to quantify the spatial distribution of SAV, we consider examples from physically-different habitats in the Lake Ontario watershed (Figures 1 and 2). The analyses assess SAV in open and closed bays with annual average water clarity that ranged from low to high during 2002 and 2003. The high water clarity habitat is the open Guffin Bay in Chaumont Bay of Lake Ontario, the moderate water clarity habitat is in the closed Sodus Bay of Lake Ontario, and the low water clarity habitat is the open Lower South Bay of Oneida Lake. This gradient in water clarity is due to differences in the standing algal biomass, substrate composition, and degree of openness to wave action. For example, Chaumont Bay has low algal biomass and limited fine sediments, so the water clarity is high despite being open to the lake and to wave action. On the other hand, Lower South Bay of Oneida Lake shows high algal biomass, is dominated by fine sediments, and is open to wave action from the east, all factors that can contribute to low water clarity.

Application of API in this study directly reflected the guidance provided by the C-CAP protocol (Dobson et al. 1995; Finkbeiner et al. 2001) to interpret the distribution of SAV in Lower South Bay, Oneida Lake; the area around Eagle, Newark, and Leroy islands of Sodus Bay, Lake Ontario; and Guffin Bay of Chaumont Bay, Lake Ontario (Figure 2). Modifications to this protocol included the use of aerial photograph(s) collected for other purposes (see below: Photograph selection and sources). The photographic key indicators of color, texture, pattern, shape, and geographic location were used to identify the presence and extent of beds of SAV, independent of species composition. This interpretation was delineated on acetate overlays (0.5 mm, Steiner Paper Corporation, Irvington, NJ) and then redrawn onto Mylar™ overlays (Charette ProPrint film 9104, Charrette Corp. Woburn, MA) registered to U.S. Geological Survey 7.5' (1:24,000 m) topographic maps using landmark ground control points. The Mylar™ overlays were digitized with a large scanner (Scangraphics CF 1000/44, monochrome) and then vectorized using ArcGIS software (ESRI, Inc. Redland, CA). As a complement to aid map interpretation, 6 bands of 25-m width were added to the maps around the perimeter of the SAV from API for the 2002 photographs. Use of set-width bands on these maps allow for an objective way to assess the spatial extent of SAV.

PHOTOGRAPH SELECTION AND SOURCES

Because the SAV distribution in lakes generally peaks during late summer (Wetzel 1983; Chambers and Kalff 1985), aerial photography that was collected between early August and early October are preferred for interpretation even though this time frame limits the consideration of the role of species that peak in distribution during the spring or early summer (e.g., nonnative curly-leaf pondweed *Potamogeton crispus;* Spence 1982). In addition to this preferred time collection, other factors considered in the photograph preview and selection process included the scale of the photograph, angle of the photograph relative to the landscape, and presence of confounding environmental conditions in the photograph (e.g., glare on water).

In this analysis, all images identified for interpretation were collected during the August–October period. All recent images were available as 35-mm color slides and obtained from the Farm Service Agency county offices of the U.S. Department of Agriculture (USDA) that contracted land-use inventories from aerial photographs collected at low elevation and also included lake shorelines. These slides sometimes did not allow for a clear and/or focused view of the SAV and/or water surface, so they needed to be previewed prior to printing, at a scale of 1:10,000. The slides of Oneida Lake were from 10–21 August 2002, of Sodus Bay were from 15–31 August 2000–2002, and of Chaumont Bay were from 10 September–30 October 2002. Images of Sodus Bay representing the period of cultural eutrophication and the early period of abatement were available as color photographs collected by the National Ocean Service on 20 August 1972 and 29 September 1980, respectively. These color photographs had a scale of 1:30,000.

FIELD VERIFICATION

Multiple methods were used to verify the findings from API. Hydroacoustic transect surveys and rake grab samples were used during 2002 and 2003 to delineate the spatial-depth distribution of SAV in all habitats. In Oneida Lake, hydroacoustic transect surveys were completed on 31 July 2002 and 5 August 2003. In Sodus Bay, hydroacoustic surveys were completed on 9 September 2002 while the Chaumont Bay hydroacoustic surveys were completed on 18 September 2002. All hydroacoustic surveys were completed with a 420 kHz transducer connected to a laptop computer running the program Eco-SAV with default settings, and maximum depth identified as appropriate for each habitat (Sabol et al. 2002). During each hydroacoustic survey, rake grab samples were collected (e.g., Weaver et al. 1997), to simultaneously confirm the depth distribution of plants visible on the laptop, and to complete plant identification.

To assess intra-annual variation in SAV in Oneida Lake during 2002, we completed nearshore surveys to estimate SAV stem density and filamentous algae coverage along a 100-m transect perpendicular to the shoreline of geographic center of Lower South Bay. Nearshore transect surveys were completed on 2 July, 25 July, 26 August, and 22 October. For each transect survey, a 1-m^2 quadrat was used to count the SAV stem density and estimate the filamentous algae coverage on the SAV at set distances from the shoreline, extending from 1 to 100 m. The SAV densities during each survey were represented by total number of stems per quadrat, grouped in six categories, as: 0, 1–10, 11–25, 26–50, 51–75, >76 stems. The filamentous algae on the SAV during each survey were represented as percent coverage, grouped in six categories, as: 0, 1–15, 16–25, 26–50, 51–75, and 76–100% coverage. Identification of SAV and filamentous algae was completed on 26 August 2002, and used available taxonomic descriptions (Wetzel 1983; Borman et al. 1999).

ASSESSMENT OF SAV ACROSS A GRADIENT OF WATER CLARITY

Oneida Lake

The API of Lower South Bay, Oneida Lake (with low water clarity), revealed that SAV during 2002 was represented by a continuous distribution along the shoreline, with an area of 2,583 ha, and extended to a maximum of ~3-m depth (Figure 3). Comparison of the SAV distribution estimated from API with a hydroacoustic survey during 2002 revealed the bay edge of the SAV distribution was underestimated, and extended to ~4 m, the maximum depth of the bay, a difference of ~1.0 m. This observation was confirmed by the rake samples that showed SAV along the entire bay (Figure 3). Thus, an additional hydroacoustic survey was completed during 2003, and this effort resolved the distribution of SAV across the bay, and showed plants were present to ~4 m (Figure 3).

The 100-m nearshore transect surveys identified intra-annual variability of the SAV distribution in Lower South Bay, Oneida Lake, during 2002 (Figure 4). Density of SAV increased between 2 July and 26 August, and then declined afterwards (Figure 4). The exception to this pattern was curly-leaf pondweed that showed the highest densities on 2 July and was essentially absent by 26 August. By contrast, filamentous algae coverage on the SAV increased through the entire survey period. The surveys on 26 August 2002 identified 10 macrophyte species, with coontail (*Ceratophyllum demersum*), eelgrass (*Vallisneria americana*), common waterweed (*Elodea canadensis*), and Eurasian watermilfoil (*Myriophyllum spicatum*) as the dominant species. This survey also identified three genera of filamentous algae: *Cladophora* spp., *Spyrogyra* spp., and *Ulothrix* spp.

Chaumont Bay

The API of Guffin Bay, Lake Ontario (with high water clarity), revealed that SAV was represented by a continuous distribution along the shoreline during 2002, with an area of 2,168 ha^2, and the SAV extended to a maximum of ~6-m depth (Figure 5). Comparison of the SAV distribution estimated from API with the hydroacoustic survey and rake samples revealed the bay edge of the SAV distribution on the south shoreline was accurately represented as ~6-m depth. By contrast, the bay edge of SAV on the north shoreline estimated with the hydroacoustic survey and rake samples extended to ~7 m while the API estimated the maximum depth as ~6 m, a difference of ~1 m. The SAV on the north and south shorelines ended with the presence of rock substrate and/or steep slope.

Sodus Bay

Inter-annual changes in SAV distribution

The API of Sodus Bay, Lake Ontario (with moderate water clarity), revealed that SAV was represented by a continuous distribution around the islands of interest, and estimated to extend beyond the 6-m depth contour during 2000–2002 (Figure 6). These analyses identified inter-annual variability in SAV distribution, with a mean of 1,867 ha (SD=234 ha, Table 5); total area was lowest during 2002 and highest in 2000, with a coefficient of variation of 12.6%. The areas of greatest inter-annual variability in SAV distribution were east of Newark Island and north of Eagle Island. Comparison of the SAV estimated with API for 2001 and 2002 with the hydroacoustic survey and rake samples during 2002 revealed the SAV distribution was underestimated, and this distance ranged from 20–130 m from the bay edges identified with API; this difference corresponds to ~5 to 15% of the total shoreline width of SAV at these crossing points with the hydroacoustic survey (Figures 6 and 7). In addition, an evaluation of a plot of the distance between the surface of the water and apical tip of the SAV estimated from the hydroacoustic survey southeast of Eagle Island suggested that the API resolved all SAV <3 m below the surface of the water (Figure 8). This hydroacoustic survey showed also the distance between the shoreline to bay edge of SAV was ~294 m, and the distance from the shoreline to the bay edge of SAV estimated from the API was ~269 m, a difference of ~25 m or 8.5%. The bay edge of the SAV beds was also confirmed with rake samples.

Table 5. Estimates of SAV in Sodus Bay area of interest for years before and after the abatement of cultural eutrophication.

Year	Area of SAV (ha)
1972	992
1980	1,378
2000	2,027
2001	1,976
2002	1,598

Inter-decade change in SAV

Large increases in the areal distribution of SAV were identified from the API in Sodus Bay, Lake Ontario, as water clarity and habitats changed between 1972 and 2000 (Table 5). The API for 1972 revealed that the SAV was represented by a patchy distribution along the shoreline, and estimated to extend to a maximum of ~3-m depth (Figure 7). The API for 1980 revealed that the SAV had expanded to a continuous distribution along the shoreline, and estimated to extend to a maximum of ~4-m depth (Figure 7). A comparison of areal estimates for SAV between 1972 and 1980 identified a 32.6% increase (Table 5). This comparison between 1972 and 1980 identified that the SAV increased primarily east and south of Newark Island and east and north of Eagle Island. The API for 2000 revealed the SAV was represented by a continuous distribution along the shoreline, and estimated to extend to a maximum beyond 6-m depth (Figure 7). A comparison of SAV distribution between 1980 and 2000 identified an increase of 38.1%, and this indicates the presence of a strong dreissenid mussel effect; comparison of SAV between 1972 and 2000 identified an increase of 68.6% (Table 5). The comparison between 1980 and 2002 identified that SAV increased primarily along the north shore of the bay, east and south of Newark Island, north and west of Eagle Island, and west of LeRoy Island (Figure 7).

DISCUSSION

Aerial photograph interpretation of photographs that were collected for other purposes consistently confirmed the spatial and temporal dynamics of SAV in open and closed bays of the Lake Ontario watershed. These analyses also quantified patterns in the distribution of SAV when no historical vegetation maps were available for this purpose. For example, the expansion of SAV in Sodus Bay was concomitant initially with the increases in water clarity that followed the GLWQA of 1972, and then with the dreissenid mussel invasion of 1991. In this bay, the SAV increased from isolated patches in 1972 to a continuous distribution by 1980, an increase of >1.0-m in depth and >30% in area. Between 1980 and 2000, the expansion in SAV was >2-m in depth and >35% by area while mean water clarity measured by summer Secchi tripled during this period, from ~1.0 to ~3.0 m (Gilman and Smith 1988; Makarewicz 2000). Comparison with field methods like hydroacoustic surveys identified API as a conservative method that underestimated the actual SAV distribution from 5–15% in habitats with contrasting water clarity. Assessment of the intra-annual density of SAV confirmed that the largest areal coverage in these lakes likely occurred during the late summer. Consideration of consecutive years in Sodus Bay identified the presence of inter-annual variation in the distribution of SAV. In combination, this suggests that the time (season) and year of photograph collection will have a larger effect on estimates of area or SAV depth distribution compared with limits of SAV estimates from the API method alone.

The waxing and waning in the SAV abundance and distribution in our study, within and across years and decades, is concordant with studies in other inland lakes (Crum and Bachmann 1973; Bumby 1977) and bays of the Great Lakes (Wentz and Stuckey 1971; Skubinna et al. 1995; Chu et al. 2004). In this study, API revealed the expansion of SAV distribution in Sodus Bay occurred over decades, and was in step with known improvements in water clarity. Even if these analyses of spatial change are conservative, they identify significant transition in SAV distribution patterns, and indicate modified trophic interactions when habitat inventories were lacking (e.g., Makarewicz 2000). For Oneida Lake, where habitat inventories are available from diver and hydroacoustic surveys, Zhu et al. (in press) reported the SAV (i.e., angiosperm macrophytes) species richness and distribution increased with improvements in water clarity during 1975–2002. Zhu et al. (in press) recognized also the density of macrophyte species that can tolerate low light levels declined (e.g., common waterweed), with corresponding increases in species that can tolerate a wide range of light levels (e.g., Richardson's pondweed *Potamogeton richardsonii*). Mills et al. (1978) reported that species like Richardson's pondweed declined in this lake between 1910 and 1967 due to cultural eutrophication, so these observations indicate the recovery of native species.

All resource inventories completed with remote sensing methods are associated with inaccuracies proportional to the total area surveyed (e.g., Lillesand and Keifer 2000). The presence of differences between the distribution data of SAV estimated with API compared with estimates from hydroacoustic surveys does not invalidate API-based resource inventories if these differences are small relative to the total distribution under study. The opportunity to complete repeated interpretations by more than one analyst can also help gauge the degree of inaccuracies stemming from API. In this study, the field surveys indicated the SAV distribution was consistently underestimated, and the percent difference was relatively small (i.e., <15%); these differences were similar across the study systems with contrasting water clarity. Because API of aquatic habitats is strongly dependent on water clarity, it would be expected that low water clarity will likely lead to a larger underestimates of SAV than at high water clarity. But because the depth distribution of aquatic plants is strongly correlated with water clarity (e.g., Chambers and Kalff 1985), the error in SAV distribution at lower water clarity may not increase as rapidly as might be expected. It is likely that API primarily detects plants that reach the surface, and the depth that these plants can achieve with some minimum water clarity. Thus, we believe the increase in SAV observed during 1972–2002 in Sodus Bay to be real and not an artifact of lower water clarity in the early years. Other confounding factors like attributes of the photograph and the visibility of the species under study will also contribute to the accuracy of interpretations. We expect the degree of underestimate will likely be lowest for water with high water clarity, although this represents an area that requires additional study. In the case of SAV in Sodus Bay, the underestimates would be expected to be lowest during the low water clarity period, as the SAV was light-limited and the distribution known to be restricted to shallow water relatively close to shore (e.g., Makarewicz 2000). Even when water clarity increased to ~2-m mean Secchi during the late 1980s, the SAV distribution only extended to a maximum of ~4-m depth (Gilman and Smith 1988). Habitat factors like degree of openness to wave action and slope also act to determine SAV distribution in lakes (e.g., Håkanson and Boulion 2002), and so it is not possible to suggest a scalar to modify potential underestimates of SAV derived from inventory methods like API. The most reliable analyses from API will be those that can use some form of field validation (e.g., Finkbeiner et al. 2001). Determination of the validity of findings from API will come from individuals familiar with both the species and habitat under study. The maps of SAV generated for past periods when no complete field surveys exist cannot be verified, and therefore should be considered to represent the minimum distribution of SAV at each sampling time.

The C-CAP protocol used in this analysis of SAV from different habitats of the Lake Ontario watershed was developed to provide guidance for the systematic assessment of benthic habitats in both lentic and lotic ecosystems (e.g., Dobson et al. 1995). For example, Nieder et al. (2004) used a modified version of this protocol and API to assess the distribution of aquatic plants like

SAV and non-native water chestnut (*Trapa natans*), in the fluvial freshwater and brackish tidal Hudson River across habitats differing in water clarity, flow, and substrates. In contrast to the assessment of habitats in the Lake Ontario watershed, the study of the Hudson River used photographs specifically collected for this purpose (Nieder et al. 2004). In that study, the remotely-sensed images of the Hudson River were collected with aerial film and converted to contact prints (e.g., Figure 9). This interpretation was completed on photograph overlays and recompiled to U.S. Geologic Survey 7.5' topographic maps. The rectified map data was then digitized. This processing generated digital map data in ESRI shapefiles that were readily available for GIS analysis (Nieder et al. 2004). The evaluation of SAV in the Hudson River was motivated by ecological and economic priorities. Specifically, SAV in this river, unlike many lakes, rarely achieves nuisance levels, except for patches of water chestnut, and the SAV that is present is highly valued and managed by the state of New York to conserve existing beds (Caraco and Cole 2002; Nieder et al. 2004). An understanding of the dynamics of SAV in the Hudson River will facilitate a more complete understanding of the food web in these habitats, and the scale of SAV affect on the physical and biological conditions in the river (Caraco and Cole 2002; Strayer et al. 2003). Because API can be used across a range of aquatic ecosystems, assessments of aquatic resources can be completed in diverse habitats. The accuracy and precision of the API can then be assessed with field surveys (e.g., Finkbeiner et al. 2001). Such efforts can provide useful information for habitat inventories when historical ground surveys or vegetation maps are not available.

Image formats beyond conventional photographs are used to inventory SAV, and digital methods can be used to expedite and/or facilitate image interpretation, processing, and storage (ASPRS 1997; Lehmann and Lachavanne 1997; Finkbeiner et al. 2001; Vis et al. 2003). Such alternative methods were not used for this project, as digitally-collected images do not exist for recent or historical time periods in the Lake Ontario watershed. Budget constraints did not allow for contracting dedicated flight(s) of the habitats to collect digital images; such data collection is also more complicated than conventional aerial photography and highly sensitive to unfavorable meteorological conditions. Also, image processing of either digitally-collected or scanned films, although available for decades, is not yet widely used for resource inventories in freshwater habitats (Remilard and Welch 1993; Lehmann and Lachavanne 1997; Lillesand and Keifer 2000; Vis et al. 2003). These methods typically involve the classification of vegetation visible in the image to estimate presence/absence or areal coverage. Another element of this process involves an assessment of the pixel classifications (i.e., precision of interpretation), which is done through re-sampling of the original image or actual field sampling of randomly selected locations visible in the image (Remilard and Welch 1993; Lehmann and Lachavanne 1997). Such image processing requires training in pixel classification. Additional data collection is both costly and time restrictive, resulting in relatively limited use of these methods. However, digital data collection and analysis has been used successfully to assess SAV in small and large lakes (e.g., Vis et al. 2003; Valley et al. 2005) and in marine habitats (e.g., Sotheran et al. 1997). These latter studies employed the skills of a trained analyst.

The API method can be used for benthic habitat evaluation to assess the consequences of natural and anthropogenic factors shaping features of aquatic resources of importance to fishes and other species (Table 4). These analyses can use photographs or 35-mm slides collected with dedicated flights or similar images available from other low-altitude flights, as long as the prints allow an analyst to readily interpret habitat features like vegetation beds (e.g., Finkbeiner et al. 2001). In all cases, environmental conditions like turbid water or hazy atmospheric conditions affects the attributes of the photograph and reduces the capability to identify the bay edge of the feature(s) under study and this necessitates use of field validation. Image analysis of existing photographs or prints made from scanned photographs allows for repeated measures to evaluate the accuracy and precision of the interpretation, and also affords the opportunity to create inexpensive digital data archives. Availability of images in an archive for an aquatic habitat could facilitate repeated analyses over long time periods (Pavlidis 1988; Finkbeiner et al. 2001). Existence of aerial photography for each county in the United States available at a low cost through government agencies like the Farm Service Agency of the USDA or other sources warrant further consideration for use with API to quantify fish habitat. Selection of suitable spatial coverage can yield reliable estimates of aquatic resources with API that will allow for the characterization of habitat features of importance to fishes and other species over different time frames, from inter-season to inter-annual to inter-decade.

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