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Landsat imagery reveals declining clarity of Maine's lakes during 1995–2010

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Abstract. Water clarity is a strong indicator of regional water quality. Unlike other common water-quality metrics, such as chlorophyll *a*, total P, or trophic status, clarity can be accurately and efficiently estimated remotely on a regional scale. Satellite-based remote sensing is useful in regions with many lakes where traditional field-sampling techniques may be prohibitively expensive. Repeated sampling of easily accessed lakes can lead to spatially irregular, nonrandom samples of a region. Remote sensing remedies this problem. We applied a remote monitoring protocol we had previously developed for Maine lakes >8 ha based on Landsat satellite data recorded during 1995–2010 to identify spatial and temporal patterns in Maine lake clarity. We focused on the overlapping region of Landsat paths 11 and 12 to increase availability of cloud-free images in August and early September, a period of relative lake stability and seasonal poor-clarity conditions well suited for annual monitoring. We divided Maine into 3 regions (northeastern, south-central, western) based on morphometric and chemical lake features. We found a general decrease in average statewide lake clarity from 4.94 to 4.38 m during 1995–2010. Water clarity ranged from 4 to 6 m during 1995–2010, but it decreased consistently during 2005–2010. Clarity in both the northeastern and western lake regions has decreased from 5.22 m in 1995 to 4.36 and 4.21 m, respectively, in 2010, whereas lake clarity in the south-central lake region (4.50 m) has not changed since 1995. Climate change, timber harvesting, or watershed morphometry may be responsible for regional water-clarity decline. Remote sensing of regional water clarity provides a more complete spatial perspective of lake water quality than existing, interest-based sampling. However, field sampling done under existing monitoring programs can be used to calibrate accurate models designed to estimate water clarity remotely.

Key words: Secchi disk, transparency, change detection, New England, remote sensing, satellite imagery, Landsat.

Water clarity, often quantified in terms of Secchi disk depth (SDD), is a strong indicator of chlorophyll *a*, total P, and trophic status (Carlson 1977). Clarity data are relatively cheap and easy to gather compared to these and other variables, so SDD is an ideal metric of regional water quality. Secchi data collected by existing state or citizen-based lake-monitoring programs can be used in satellite-based approaches to monitor lake water quality at regional scales (Kloiber et al. 2002, Chipman et al. 2004, Olmanson et al. 2008, 2011, Knight and Voth 2012, McCullough et al. 2012). Similar approaches can be used to monitor intralake water clarity of large lakes in targeted geographic areas (e.g., Duan et al. 2009, Zhao et al. 2011) and other water-quality metrics, such as colored dissolved organic matter (CDOM) (e.g., Brezonik et al. 2005, Kutser 2012) or chlorophyll *a* (e.g., Allan et al. 2011,

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Potes et al. 2011). However, application at regional scales is more limited by costs and availability of field data than in the case of water clarity. SDD measurements are widely conducted and less costly than other water-quality assessments requiring chemical analyses. However, large-scale field-sampling programs often gather a spatially irregular, nonrandom representation of regional water quality because of limited lake accessibility. Remote sensing can eliminate spatial biases associated with nonrandom sampling, particularly in regions with numerous lakes that cannot be monitored efficiently with traditional field methods. Much of existing field data is amassed by volunteer lakeshore residents who collectively make regional assessments more feasible by collecting necessary data for remote model calibration, and are important stakeholders in lake water quality. Increased lake clarity positively affects lakefront property value in Maine (Michael et al. 1996, Boyle et al. 1999) and New Hampshire (Gibbs et al. 2002) and enhances human-perception of lake water quality in Minnesota (Heiskary and Walker 1988).

Remote sensing often is used to detect landscape change and can be applied to monitor change in regional lake water quality. Peckham and Lillesand (2006) and Olmanson et al. (2008) used Landsat satellite imagery to evaluate long-term patterns in water quality of Wisconsin and Minnesota lakes, respectively. Identification of areas undergoing downward trends in water quality enables management agencies to direct limited resources more effectively and efficiently to remediate causes for water-quality decline. Accuracy of detection of long-term change is maximized with assessments focused on late summer, a period of relative stability in lake algal communities and lake stratification ideal for remote estimation of water clarity. Assessments during this period typically capture the seasonally poorest conditions in lake water clarity (Stadelmann et al. 2001, Kloiber et al. 2002, Chipman et al. 2004, Olmanson et al. 2008, 2011).

Our objectives were to: 1) examine spatial and temporal patterns in Maine lake clarity during 1995–2010 with a previously developed Landsat-based procedure (McCullough et al. 2012), 2) evaluate the effectiveness of Maine's existing field-sampling programs in characterizing regional water quality, and 3) attempt to explain regional differences in Maine lake clarity according to dominant land use (forest harvest) or watershed topography. Our analyses are an exemplary case study of the effectiveness and shortcomings of current satellite and field-based lake-monitoring programs from an applied perspective. We expect our findings to provide useful information to lake-management agencies inside and outside of Maine that face the challenge of cost-effective monitoring of numerous lakes over large areas.

Methods

Description of study area

Maine is in the northeastern USA and ranks first among states east of the Great Lakes in total area of inland surface waters (Davis et al. 1978). Maine contains over 5500 lakes and ponds >1 ha in surface area across an area of ~90,000 km^2, and wetlands cover 26% of the state (Tiner 1998). The climate is cold–temperate and moist with long, cold winters and short, warm summers. Maine is dominated by the Northeastern Highlands (No. 58) and the Acadian Plains and Hills (No. 82) Level III Ecoregions (Omernik 1987). The Northeastern Highlands are remote, mostly forested, mountainous, and contain numerous high-elevation, glacial lakes. The Acadian Plains and Hills are relatively more populated and less rugged, but the area also is heavily forested and contains dense concentrations of glacial lakes (USEPA 2010). Statewide lake water-clarity monitoring began in 1970. The average annual SDD consistently has remained 4 to 6 m, with a historical average of 5.28 m during 1970–2011, and was 5.46 m in 2011 (n = 367; MDEP and Bacon 2012, VLMP 2012). The number of lakes sampled in the field by state biologists and volunteers changes annually and generally has increased from 18 lakes in 1970 to consistently >350 lakes since 1999.

We focused our study on the overlapping region of Landsat paths 11 (rows 27–29) and 12 (rows 27–30), which captures a strong north–south gradient over an area of 3,000,000 ha, and includes 570 lakes >8 ha (Fig. 1). Lakes <8 ha cannot be estimated reliably with 30-m Landsat data (Olmanson et al. 2008). We narrowed our study to the overlap area because it allowed us to examine a consistent set of lakes based on an image from either path 11 or 12. We partitioned Maine's lakes (>8 ha) into 3 geographic regions (northeastern: 227 lakes, south-central: 256 lakes, western: 162 lakes) based on cluster analysis of morphometric and chemical lake variables including surface area, flushing rate, average and maximum depth, elevation, color, alkalinity, and specific conductance (Bacon and Bouchard 1997) (Fig. 1).

Satellite background

The Landsat satellite program was launched in 1972. Three satellites currently are in operation

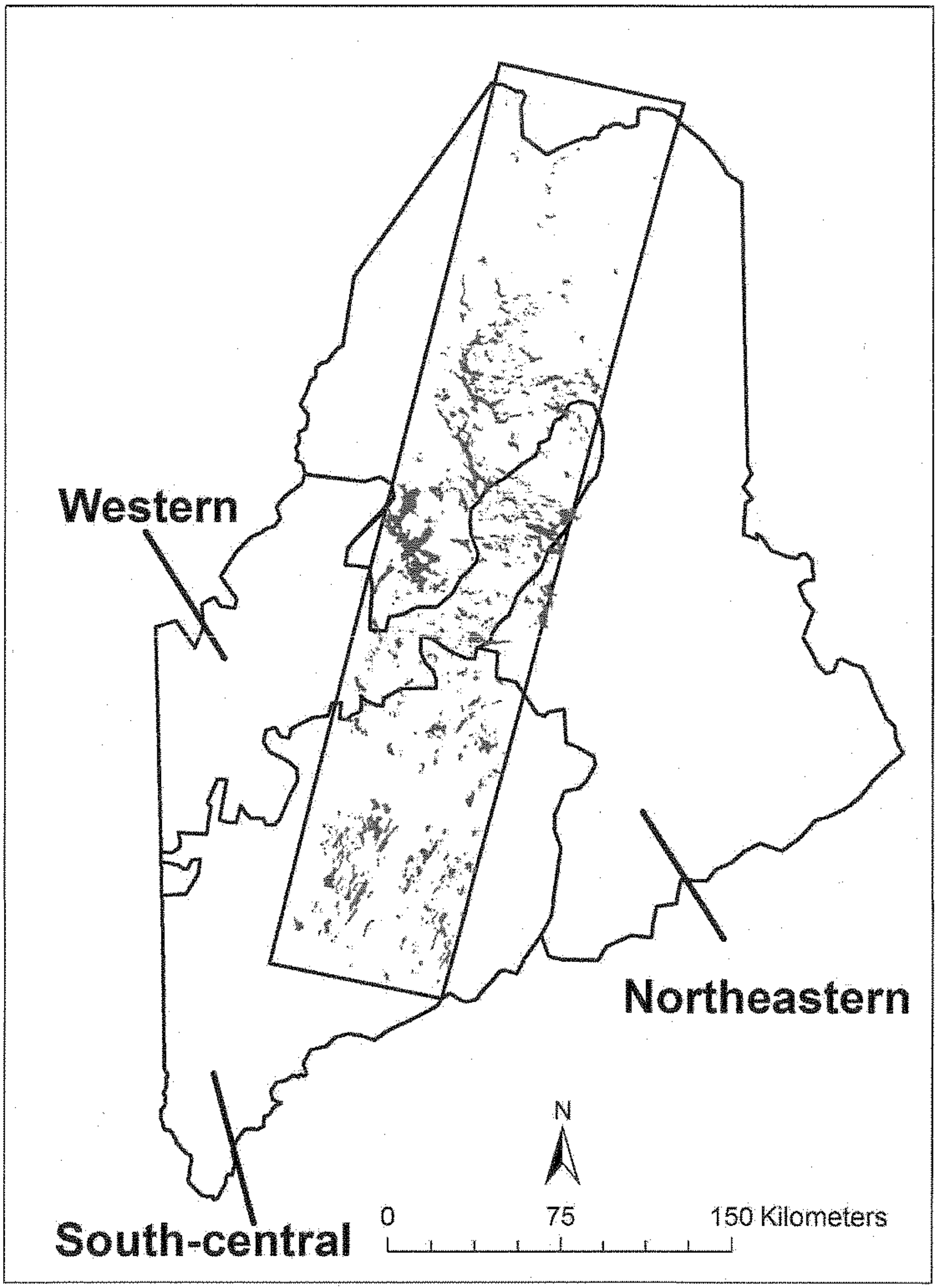

FIG. 1. Lake regions of Maine and the overlap area between Landsat paths 11 and 12, containing 570 lakes >8 ha.

(Landsat 5, 7, and 8), but the image quality of the former 2 is compromised by mechanical failures. The successful February 2013 launch of Landsat 8 ensures future availability of Landsat data for remote lake monitoring. Landsat 5, launched in 1984, experienced failure of its main sensor (Thematic Mapper [TM]) in November 2011 and is no longer a source of future lake-monitoring data. Landsat 7 was launched in 1999, but the 2003 failure of the scan-line corrector (SLC), an instrument that corrects for the forward motion of the satellite, has resulted in considerable data loss. Post-2002 (SLC-off) images are usable for remote lake monitoring with some additional processing (Olmanson et al. 2008, 2011), but fewer lakes can be monitored than before 2003. Landsat 5 and 7 contain 3 visible bands and 4 infrared bands at 30-m resolution, and Landsat 7 contains a 15-m panchromatic band. Landsat 8 contains the same bands as its predecessors, and 2 new 30-m bands. Images (scenes) of the same location are captured every 16 d and cover ~185 km^2. Scenes are indexed by path and row and are freely downloadable from the US Geological Survey Global Visualization Viewer (http://glovis.usgs.gov/).

Catalog of lake-clarity estimates during 1995–2010

Our methods used to create the catalog of lake clarity estimates are detailed in McCullough et al. (2012) and summarized here. We estimated regional lake clarity with field-collected SDD data ±1–7 d of satellite image capture, Landsat brightness values from bands 1 (blue visible; 0.45–0.52 μm) and 3 (red visible; 0.63–0.69 μm), average lake depth (MDEP and Bacon 2012), and the proportion of a lake watershed in wetlands (National Wetlands Inventory [NWI]) with linear regression. Landsat bands 1 and 3 are strongly correlated with SDD (Kloiber et al. 2002, Chipman et al. 2004, Olmanson et al. 2008, McCullough et al. 2012), and lake depth and landscape characteristics that affect water clarity (such as watershed wetland area) improve model accuracy (McCullough et al. 2012). We extracted spectral data from areas delineated by a 75-m buffered geographic information system (GIS) points layer in ArcGIS® (version 10.0; Environmental Systems Research Institute, Redlands, California) of digitized sampling stations where SDD data are collected in the field, usually in the deepest areas of lakes. We used lake centers in the absence of established sampling locations. Targeting deep portions of lakes away from the shoreline avoids spectral interference from aquatic plants, lake bottoms, and shoreline features (Kloiber et al. 2002, Olmanson et al. 2008). We analyzed radiometrically normalized, mostly cloud-free (<10% cloud cover) Landsat 5 and 7 images captured in 1995, 1999, 2002, 2003, 2005 (2 dates), 2008, 2009, and 2010. We restricted our image dates to late summer (1 August–5 September) to capture the seasonally poor clarity conditions that occur in late summer before autumn turnover. Dimictic lakes can undergo turnover as early as late August in northern Maine (Davis et al. 1978), but we found that SDD estimates generated from 5 September 2009 were consistent with late summer, preturnover clarity conditions (McCullough et al. 2012).

SLC-off images have been used to calibrate remote SDD estimation models for Minnesota lakes with strong fitness (R^2 = 0.72–0.86) (Olmanson et al. 2008, 2011). However, we were forced to use only Landsat 5 and 7 SLC-on images (Table 1) because of inconsistencies in our calibrations of models generated with SLC-off images (17 August 2003, 8 August 2005, and 1 September 2008) resulting from a combination of SLC-related data loss and cloudy conditions. We calibrated primary regression models (R^2 = 0.73–0.90) for the 6 remaining years (1995, 1999, 2002, 2005, 2009, and 2010) during 1995–2010 (Table 1). We also fit 6 similar, alternate models with slightly reduced fitness (R^2 = 0.70–0.86) corresponding to each primary model when ancillary lake data were unavailable (102 lakes). These 102 lakes, mostly in remote areas, have not yet been bathymetrically surveyed. Calibration data sets included 31 to 119 field-collected SDD data points based on the number of lakes sampled within the ±1–7 d calibration window.

Statistical analyses

Our assessment of Maine's recent water-clarity history included nearly the entire population of lakes >8 ha in the Landsat overlap region. We used SDD data from a minimum of 455 lake estimates in 2005 to a maximum of 644 lake estimates in 1999 (some lakes have >1 sampling station). We tested for differences in SDD according to lake region and year with a 3 × 5 factorial analysis of variance (ANOVA) (with 3 and 5 levels of 2 factors) based on type-III sum of squares and unequal sample sizes to avoid eliminating data points. We considered using a repeated measures design, but shifting positions of clouds (which prevent remote monitoring) would have resulted in unnecessary elimination of lakes. Furthermore, part of the intention of remote monitoring of water quality is to reduce the need for extrapolations based on incomplete data, and our ability to include nearly all of the lakes in the study area reduced the need for complex statistics. Restricting our data set to lakes

TABLE 1. Primary regression models for remote clarity estimation in Maine's lakes. TM1 = Landsat band 1, TM3 = Landsat band 3, AvgDepth = average lake depth, Wetland = proportion of watershed covered by wetland.

Date	Satellite	Path	Model	R^2
14 August 1995	Landsat 5	11	(9.35×10^{-3})TM1 − (5.87×10^{-2})TM3 + (9.83×10^{-3}) AvgDepth − (3.06×10^{-4})Wetland + 3.91	0.7919
1 September 1999	Landsat 5	12	(−0.427)TM3 + (4.48×10^{-3})AvgDepth + 6.22	0.8939
9 August 2002	Landsat 7	11	(-3.22×10^{-2})TM3 + (1.29×10^{-2})AvgDepth − (7.51×10^{-4}) Wetland + 4.25	0.9010
9 August 2005	Landsat 5	11	(0.113)TM1 − (0.315)TM3 + (7.89×10^{-3})AvgDepth − (3.70×10^{-4})Wetland − 0.868	0.8244
5 September 2009	Landsat 5	11	(3.715×10^{-2})TM1 − (0.320)TM3 + (7.77×10^{-3})AvgDepth − (3.61×10^{-4})Wetland + 5.51	0.8631
30 August 2010	Landsat 5	12	(−0.244)TM3 + (8.39×10^{-3})AvgDepth + 5.22	0.7305

sampled in each year of the study would have reduced our data set to 347 lake estimates, whereas maintaining a larger sample size during the 15-y interval reduced the risk of committing type I and II errors. We compared average SDD between pairs of years and lake regions with pairwise t-tests ($\alpha = 0.05$). We did not pool standard deviation, and we assumed equal variance within group pairs. We also used pairwise t-tests to assess Maine's existing field-based lake clarity monitoring program by comparing average SDD data collected remotely on our 6 image dates to all field data collected in the overlap region during theoretical model calibration windows (±7 d of image capture constrained within 1 August–5 September; McCullough et al. 2012). Basing our comparison on field data gathered within 7 d of satellite overpass during the late summer period of lake stability reduced introduction of error associated with changing lake conditions. These field data would be considered sufficiently reflective of lake conditions captured by the satellite to be used in a remote lake-clarity estimation model. We considered comparing remotely sensed data to all field data collected in Maine during the ±7-d window, but including lakes outside the overlap region could introduce unnecessary error attributable to geographic variability. These analyses allowed us to evaluate the effectiveness of current field monitoring for assessing regional water quality in Maine. We were unable to analyze lake regions separately because we had insufficient field data in the northeastern and western regions.

We also investigated potential explanations of water-clarity change in Maine during 1995–2010. Such analyses were somewhat limited by availability of widespread data. We first examined the effect of the proportion of lake watersheds harvested for timber (the dominant land use in northern Maine) on SDD during 1995–2010 using Landsat-derived forest-harvest data from 1991–2007 (Noone et al. 2012). Acknowledging that total harvest area is insensitive to harvest intensity, we examined the effects of recent and cumulative light and heavy partial harvest/clearcuts on SDD during 1995–2005 using annual forest-harvest maps from 1988–2004 (K. R. Legaard, University of Maine, unpublished data). Locations under light partial harvest were characterized by <70% basal area removal. Long-term forest-harvest data in spatial form restricted our analyses to lakes in Landsat path 12, scene 28. We also analyzed effects of watershed topography (average and maximum slope) on SDD for the same set of lakes. Watershed topography can influence SDD strongly (D'Arcy and Carignan 1997) and, therefore, may determine the extent of effects of landuse practices (e.g., forest harvest) on specific lakes.

Results

Temporal analysis

Water clarity estimated by SDD during 1995–2010 was related to year (ANOVA, $F_{5,10} = 16.472$, $p < 0.001$). Average SDD decreased from 4.94 to 4.38 m during 1995–2010 (Table 2, Fig. 2). SDD varied during this 15-y period, with a statewide peak of 5.64 m in 1999, followed by a consistently shallower SDD (<5.00 m) since 2002. The 0.56-m estimated decrease during 1995–2010 was a significant reduction ($t_{1130} = 4.605$, $p < 0.001$) representing an 11% overall reduction in lake clarity.

The proportion of eutrophic lakes in Maine increased from 35.3 to 42.6% during 1995–2010 (Fig. 3), based on all lakes remotely assessed. The proportion of mesotrophic lakes was unchanged since 1995. However, the proportion of oligotrophic lakes decreased from 14.8% in 1995 to 6.8% in 2010 (Fig. 3), suggesting that Maine lakes are becoming generally more eutrophic. Of the 547 lakes from which SDD data were retrieved during 1995–2010, 79 (14.4%) previously mesotrophic lakes became eutrophic and 66 (12.1%) previously oligotrophic lakes became

TABLE 2. Remotely estimated annual Secchi disk depth (m) in Maine (1995–2010). n varied among years because of cloud cover.

Statistic	1995	1999	2002	2005	2009	2010
Mean	4.94	5.64	4.64	4.81	4.65	4.38
Median	4.75	6.09	4.36	4.67	4.52	4.27
Min	0.43	0.02	0.30	0.86	0.34	0.02
Max	14.25	11.83	15.02	11.65	10.90	11.41
n	587	644	630	455	517	633

mesotrophic, whereas 327 (59.8%) lakes were unchanged in trophic status, 72 (13.2%) lakes improved, and 3 (0.55%) previously oligotrophic lakes became eutrophic (Fig. 3).

Regional analysis

Water clarity estimated by SDD during 1995–2010 was related to lake region (ANOVA, $F_{2,5} = 8.015$, $p < 0.001$). Average SDD was slightly >5 m in the northeastern and western lake regions and ~0.5 m less than this depth in the south-central lake region, except in 2005, when SDD was fairly uniform throughout Maine, and in 2010, when SDD in the south-central region exceeded SDD in the other 2 regions (Table 3, Fig. 4). Pairwise t-tests revealed significant differences ($\alpha = 0.05$, $p < 0.001$ except where specified) between average SDD in the northeastern and south-central lake regions in 1995 ($t_{436} = 3.320$), 1999 ($t_{480} = 3.808$), and 2009 ($t_{358} = 3.902$) and in the western and south-central lake regions in 1995 ($t_{376} = 3.496$), 1999 ($t_{415} = 2.026$, $p = 0.043$), 2002 ($t_{406} = 4.121$), and 2009 ($t_{401} = 5.488$). In 1995, average SDD in both the northeastern and western regions was estimated at 5.22 m, though it decreased to 4.36 and 4.23 m, respectively, in 2010. Conversely, average SDD in the south-central lake region fluctuated within a 1 m range and was nearly the same in 1995 as in 2010 (4.50 m) (Table 3, Fig. 4).

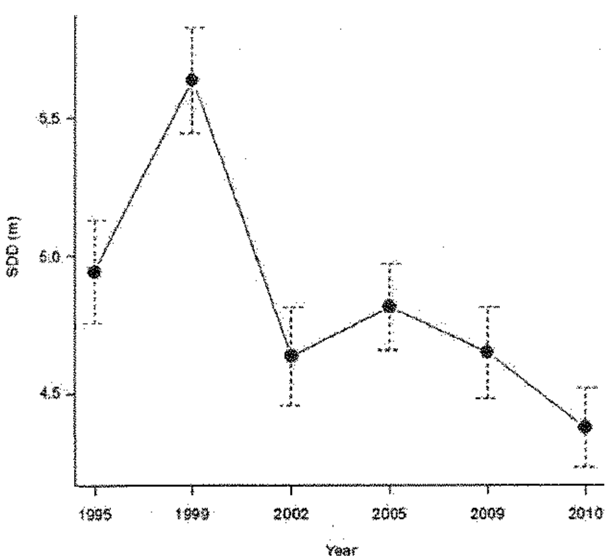

FIG. 2. Remotely estimated mean (±95% confidence interval) annual late summer Secchi disk depth (SDD) of Maine lakes during 1995–2010 based on the overlap area between Landsat paths 11 and 12. n = 455–645 lake samples (see Table 2).

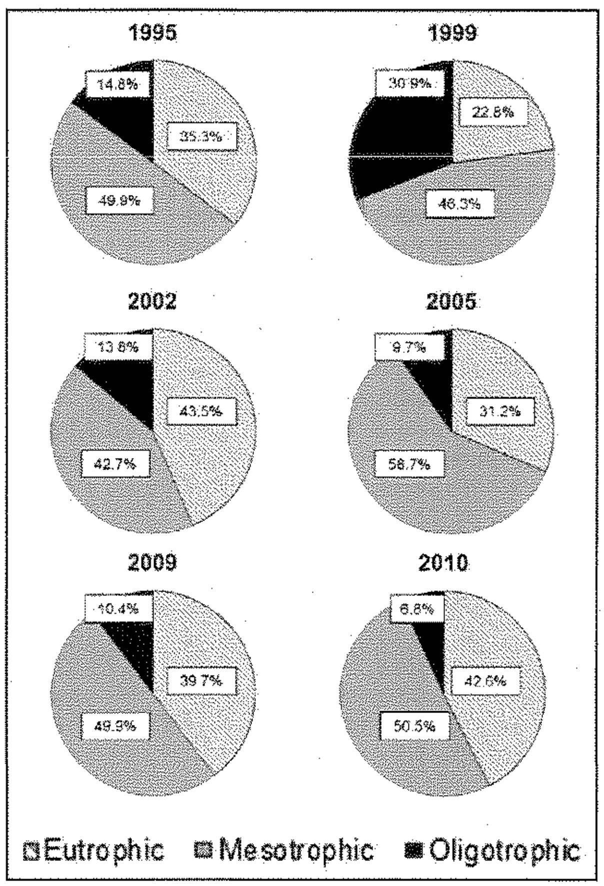

FIG. 3. Proportions of Maine lakes in various trophic states during 1995–2010 based on remotely sensed data in the Landsat paths 11 and 12 overlap area. Eutrophic: Secchi disk depth (SDD) < 4 m, mesotrophic: SDD = 4 to 7 m, oligotrophic: SDD > 7 m.

TABLE 3. Mean (±1 SE) annual late summer Secchi disk depth (m) by lake region (remote assessment) and assessment type in Maine (1995–2010). n varied in remote assessments because of cloud cover and in field assessments because of data availability.

Variable	1995	1999	2002	2005	2009	2010
Lake region						
Northeastern	5.22 ± 0.19 n = 209	6.07 ± 0.18 n = 227	4.82 ± 0.17 n = 222	4.89 ± 0.17 n = 152	5.02 ± 0.22 n = 114	4.36 ± 0.14 n = 227
South-central	4.51 ± 0.10 n = 229	5.21 ± 0.14 n = 255	4.20 ± 0.12 n = 248	4.79 ± 0.10 n = 168	4.18 ± 0.10 n = 246	4.50 ± 0.12 n = 256
Western	5.22 ± 0.20 n = 149	5.69 ± 0.18 n = 162	5.06 ± 0.19 n = 160	4.76 ± 0.13 n = 135	5.11 ± 0.14 n = 157	4.23 ± 0.12 n = 159
Assessment						
Field	5.46 ± 0.57 n = 91	5.51 ± 0.69 n = 63	5.22 ± 0.58 n = 81	4.96 ± 0.54 n = 84	4.43 ± 0.68 n = 43	5.31 ± 0.63 n = 71
Remote	4.94 ± 0.20 n = 587	5.64 ± 0.22 n = 644	4.64 ± 0.18 n = 630	4.81 ± 0.23 n = 455	4.65 ± 0.20 n = 517	4.38 ± 0.17 n = 633

Analysis of existing sampling record

The existing water-clarity field-sampling program in Maine does not consistently provide a representative sample of regional water quality. We compared the average SDD of all remote estimates of lakes >8 ha in the overlap region on each of our 6 dates (Table 2) to the average field-collected SDD during theoretical model-calibration windows (±7 d of image capture, constrained within 1 August–5 September). Pairwise t-tests indicated that remotely sensed average SDD estimates differed significantly from field data in 3 of 6 y: 1995 (t_{676} = 1.985, p = 0.048), 2002 (t_{709} = 2.165, p = 0.031), and 2010 (t_{703} = 3.796, p = 0.001) (Table 3). The absolute differences between annual average SDD measured in the field and remotely ranged 0.13–0.93 m and remote estimates underpredicted overall field conditions in 4 of 6 y (Table 3).

Landscape drivers of lake clarity

We found no correlation between the 15-y decline in SDD and the proportion of lake watersheds harvested for timber during 1991–2007 based on Landsat-derived forest-harvest data (Noone et al. 2012). Using the harvest intensity data set from 1988–2004 (K. R. Legaard, University of Maine, unpublished data), we found a significant (α = 0.05) negative correlation between the proportion of lake watersheds under light partial harvest during 1988–

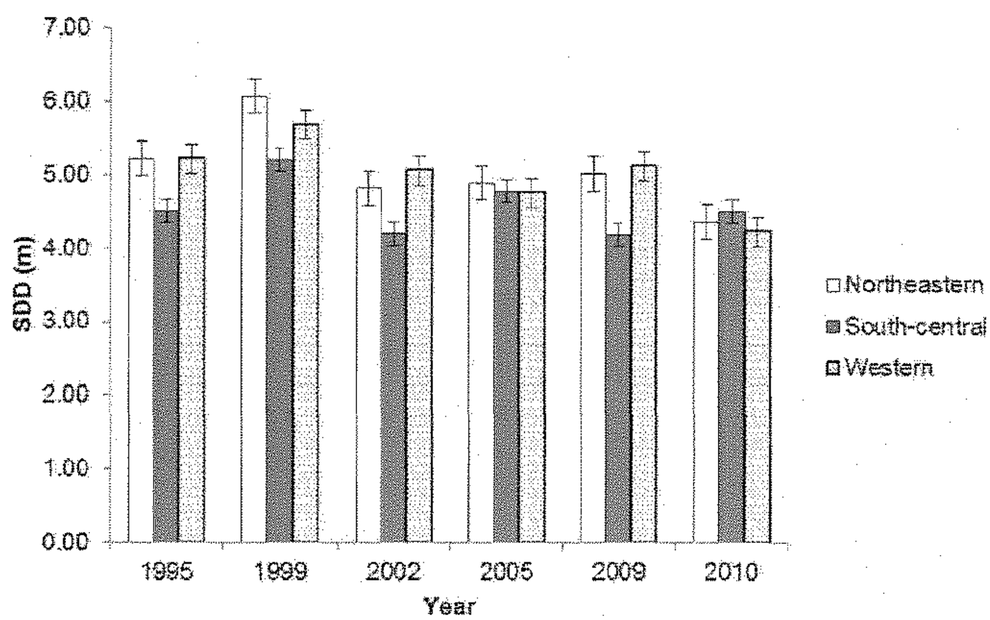

FIG. 4. Mean (±1 SE) annual late summer Secchi disk depth (SDD) of Maine lakes by lake region during 1995–2010 based on remotely sensed data from the Landsat paths 11 and 12 overlap area.

Table 4. Effects of light partial and heavy partial forest harvest/clear-cuts on remotely estimated Secchi disk depth (m) in 1995, 1999, 2002 and 2005. Italicized p-values are significant based on $\alpha = 0.05$.

Harvest	Year	Harvest Period	r	df	p
Light	1995	1988–1995	−0.1824	265	*0.0028*
	1999	1995–1999	0.0075	288	0.7417
	1999	1988–1999	−0.1432	288	*0.0087*
	2002	1999–2001	−0.1819	282	*0.0071*
	2002	1988–2001	−0.2059	282	*0.0005*
	2005	2001–2004	−0.0484	204	0.5575
	2005	1988–2004	−0.1449	204	*0.0242*
Heavy	1995	1988–1995	−0.0418	265	0.8560
	1999	1995–1999	−0.0974	288	*0.0100*
	1999	1988–1999	−0.1243	288	*0.0143*
	2002	1999–2001	−0.1549	282	*0.0229*
	2002	1988–2001	−0.1059	282	*0.0268*
	2005	2001–2004	−0.0447	204	0.5893
	2005	1988–2004	−0.0748	204	0.2605

2004 and SDD in 2005 ($r = -0.1449$, df $= 204$, $p = 0.0242$) (Table 4). Correlations consistently were negative and significant when cumulative light harvest (total area harvested before a certain date) was compared to SDD in each year, whereas effects of recent harvest (years since previous harvest period) were less consistent. We found no significant effect of heavy harvest/clear-cuts during 1988–2004 on remotely estimated SDD in 2005 ($r = -0.0748$, df $=$ 204, $p = 0.2605$). However, effects of recent and cumulative heavy harvest/clear-cuts on SDD were negative in all 4 y and significant in 1999 and 2002 (Table 4). Exclusion of lakes in protected areas (e.g., Baxter State Park) had minimal effects on correlations.

Our analyses of topographic effects on SDD yielded more straightforward and consistent results. Maximum watershed slope was significantly positively associated with SDD in all 6 y ($r = 0.30$–0.46, df $=$ 202–288, $p < 0.0001$). Average watershed slope was significantly positively associated with SDD in all 6 y except 1999 ($r = 0.15$–0.49, df $=$ 202–288, $p \leq 0.01$).

Discussion

Spatial and temporal patterns in Maine lake clarity

Water clarity of Maine lakes appears to be declining statewide. Although average SDD in both the northeastern and western regions was >5 m in 2009, depths similar to 1995 levels (Table 3), we may be witnessing a downward shift in the baseline and general trend toward eutrophication in Maine lakes. The summer of 1999 was unusually dry (NOAA 2013), which probably explains the relatively deep SDD observations in that year because of reduced amounts of DOC-containing runoff. The proportion of Maine lakes in mesotrophic status appears stable, but 79 formerly mesotrophic lakes have become eutrophic and 64 previously oligotrophic lakes have become mesotrophic, further evidence of a general trend toward eutrophication (SDD < 4 m). Based on our regional analysis, the disproportionate shifts in the northeastern and western regions and stability in the south-central region are corroborated when SDD change during 1995–2010 is mapped (Fig. 5). Despite overall stability in the south-central region, lakes with increased SDD during 1995–2010 occurred most often in this region (52 of 72 lakes) and were comparatively small in size (average = 49 ha), whereas lakes with reduced clarity occurred disproportionately in the remote northeastern and western lake regions (52 of 63 lakes) and were relatively larger (average = 403 ha). Lake size is an inconsistent predictor of Maine lake clarity (McCullough et al. 2012), but smaller lakes may be more immediately responsive to management strategies aimed at improving lake water quality.

Changes in climate that affect algal growth and changes in forest cover in lake watersheds may explain the disproportionate decline in lake clarity in the northeastern and western lake regions. Warmer temperatures and extended growing seasons associated with climate change may be creating conditions that favor increased lake productivity. Alternatively, the dominant land use (forest harvest) in northern Maine also may affect the region's lake water clarity. The significant negative correlations between the proportion of lake watersheds under light partial harvest and SDD (Table 4), particularly over longer time periods (1988–1995, 1988–1999, 1988–2001, 1988–2004), suggest potential long-term, cumulative impacts of light harvest on SDD. Correlations between proportions of lake watersheds under heavy harvest/clear-cuts on SDD were less suggestive of cumulative

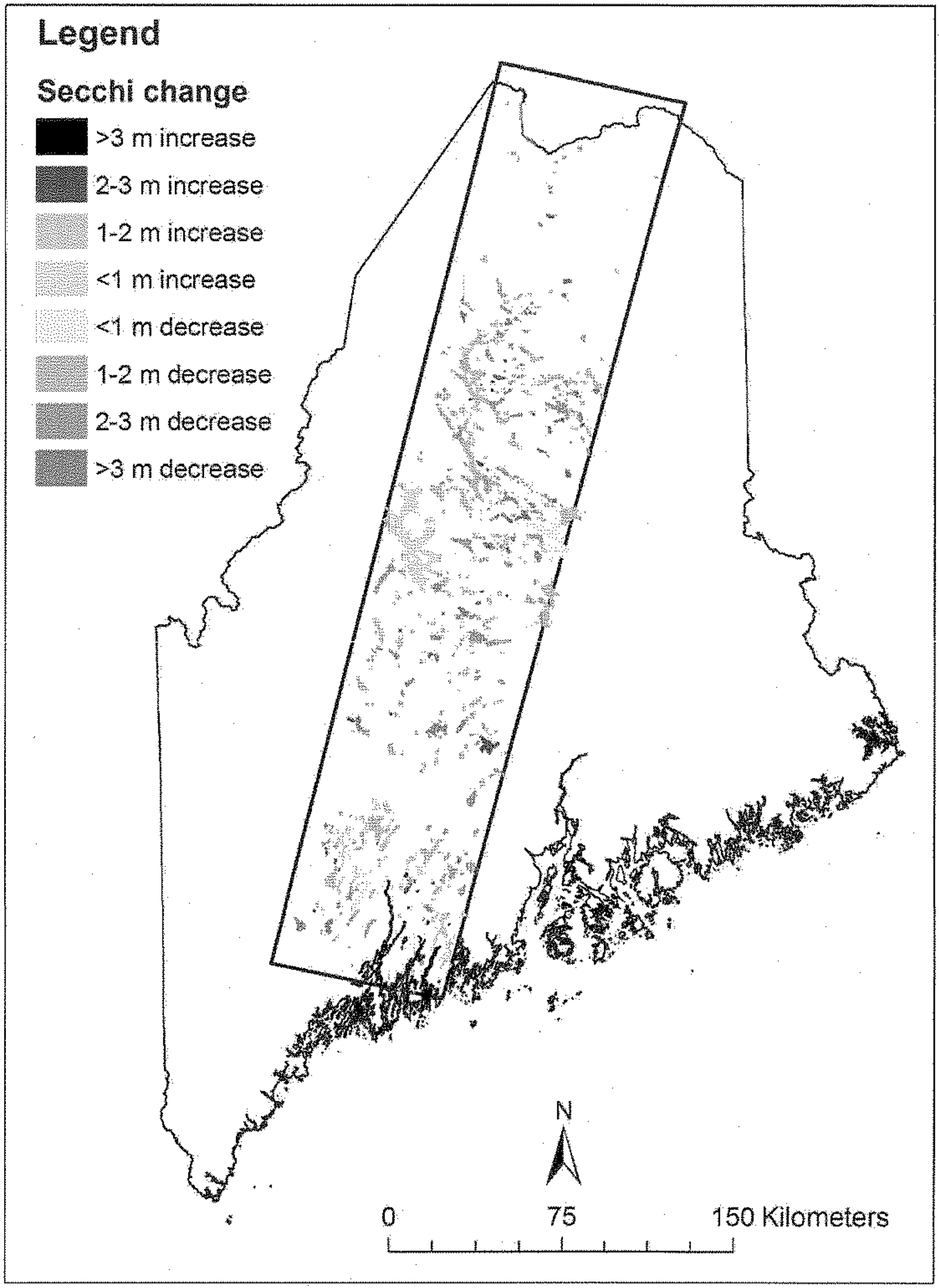

FIG. 5. Change in water clarity in Maine lakes based on remotely estimated Secchi disk depth (m) during 1995–2010 in the overlap region between Landsat paths 11 and 12.

effects, perhaps because heavy harvests occur less frequently. However, effects of recent and cumulative heavy harvest on SDD were negative in all 4 y and significant in 1999 and 2002 (Table 4), evidence that heavy harvest may affect SDD. Exclusion of lakes in protected areas (e.g., Baxter State Park) had minimal effects on correlations.

Other investigators have suggested that forest harvest affects lake clarity. Steedman and Kushneriuk (2000) found that experimental clear-cuts decreased SDD in 3 Ontario lakes by 0.4 to 1.0 m (6.0–14.1%) 3 y post-harvest. Factors that affect lake clarity including concentrations of total P, chlorophyll *a*, cyanobacteria, and cyanotoxins increased in a 2-y posttreatment study of 0 to 35% harvesting of lake watersheds on Alberta's Boreal Plain (Prepas et al. 2001). Shallow or weakly stratified lakes were most affected by forest harvest, and forested buffers of 20, 100, and 200 m around lakes had no effect on water quality, results suggesting that forest harvest in entire watersheds must be managed carefully to maintain water quality (Prepas et al. 2001). Carignan et al. (2000) found comparatively greater total P, dissolved organic C (DOC), and extinction of photosynthetically active radiation (PAR) in logged (14–97%) vs undisturbed watersheds of Quebec Shield lakes and suggested these lake effects were long lasting.

The notion that forest harvest can affect lake water clarity, combined with our findings that steeply sloping watersheds are associated with increased water clarity (findings that confirm those of D'Arcy and Carignan 1997), suggest that timber harvests in steep watersheds have relatively less impact on water clarity. Our study area is a transition zone between the Northeastern Highlands (No. 58) and the Acadian Plains and Hills (No. 82) Level III Ecoregions (Omernik 1987). This topographic heterogeneity creates widely variable watershed morphometries in west-central Maine, whereas eastern Maine is relatively flat and contains more wetlands. Steep slopes contain fewer water containment areas in which clarity-reducing sediments and DOC can accumulate.

Evaluation of existing sampling record

Maine's current water-clarity sampling approach does not necessarily acquire a representative sample of regional water quality because of spatially biased field sampling and omission of inaccessible lakes. This sort of selective sampling system and its implications are unlikely to be unique to Maine. Remote-lake monitoring schemes enable spatially balanced sampling because assessment is not limited by access. Landsat-based models produce accurate estimates of water clarity in Maine overall and can be calibrated with nonrandom field data, but prediction error is greater in regions with few field-sampled lakes. During the selected 6 study years, field data were available for 43 to 91 unique lakes, representing only 8 to 16% of the 570 lakes >8 ha in the imagery overlap region. Field-collected data ($\leq$5 sampled lakes within $\pm$7-d calibration windows) in the northeastern and western lake regions were insufficient to evaluate model predictions for lakes in those regions, underscoring the spatial biases in current field-sampling programs. Regional water-quality analyses may be similarly limited in areas outside of Maine that use spatially biased field sampling.

Application of Landsat imagery for change detection of regional lake water quality

Landsat data are an effective tool in regional water-quality monitoring because the spatial extent of Landsat imagery eliminates the biases of nonrandom sampling typically used in the field. Near-concurrent ($\pm$7 d of satellite overpass) field data must be collected for model calibration, but remote water-quality monitoring with Landsat data can make use of existing field-based lake monitoring programs to increase substantially the geographic extent of lake monitoring at the disproportionately small expense of conducting GIS analyses. Annual monitoring for purposes of detecting changes in water quality is unreliable because of irregular availability of clear images. However, the complete, spatially extensive data sets afforded by remote sensing methods every few years represent a potentially major asset for lake management agencies. Another notable limitation of Landsat-based monitoring is that restricting usable images to late summer when lakes are expected to be least clear reduces image availability. This self-imposed restriction can be further complicated by cloud cover or the 16-d Landsat revisit cycle. However, using scene overlap areas between Landsat paths is a practical approach to increasing image availability.

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