esa

ECOSPHERE

Quantifying land use effects on forested riparian buffer vegetation structure using LiDAR data

Leah Wasser,[1,]† Laura Chasmer,[2] Rick Day,[3] and Alan Taylor[4]

[1]*National Ecological Observatory Network, Boulder, Colorado 80301*
[2]*Department of Geography, University of Lethbridge, Lethbridge, Alberta T1K3M4 Canada*
[3]*Department of Ecosystem Science and Management, Pennsylvania State University, University Park, Pennsylvania 16802 USA*
[4]*Department of Geography, Earth and Environmental Systems Institute, Pennsylvania State University, University Park, Pennsylvania 16802 USA*

Citation: Wasser, L., L. Chasmer, R. Day, and A. Taylor. 2014. Quantifying land use effects on forested riparian buffer vegetation structure using LiDAR data. Ecosphere 6(1):10. http://dx.doi.org/10.1890/ES14-00204.1

Abstract. Quantifying variability of forested riparian buffer (FRB) vegetation structure with variation in adjacent land use supports an understanding of how anthropogenic disturbance influences the ability of riparian systems to perform ecosystem services. However, quantifying FRB structure over large regions is a challenge and requires efficient data collection and processing methods that integrate conventional in situ vegetation sampling with remote sensing data. This study uses automated algorithms to process airborne light detection and ranging (LiDAR) data for mapping of riparian vegetation height, canopy cover and corridor width along 5,900 transects using methods validated in 80 mensuration plots in central Pennsylvania, USA. The key objective of this study was to use airborne LiDAR data to quantify differences in edge vs interior vegetation structure as influenced by buffer width and adjacent land use type, continuously throughout a watershed. Riparian vegetation height, canopy cover and buffer width were estimated along FRB transects adjacent to developed (residential/commercial and agricultural) and undeveloped (grassland) land use types and compared to reference transects within larger forested areas and thus without an edge. On average, buffers adjacent to developed land use types were narrower than those adjacent to natural, undeveloped land use types. Approximately 50% of streams in the watershed had FRB corridors $\leq$30 m wide. Only 23% of streams had a corridor width $\geq$200 m, the width recommended to support key ecosystem services. Undeveloped land use types contained taller riparian vegetation and wider corridors, whereas developed land use types contained shorter riparian vegetation and narrow FRB corridors. Edge effects also affected vegetation structure. Vegetation height was 5–8 m shorter at the interface between the FRB and the adjacent land use (the matrix) than in the naturally occurring stream edge or in the corridor interior. Canopy cover was not influenced by adjacent land use type or width. This study demonstrates that airborne LiDAR data can be used to accurately map riparian buffer vegetation width, height and canopy cover to support ecological based management of riparian corridors over wide areas.

Key words: airborne laser scanning; canopy conditions; disturbance; edge effects; forest inventory; LiDAR; remote sensing; riparian forests; vegetation structure; wide area mapping.

Received 24 June 2014; revised 17 September 2014; accepted 18 September 2014; final version received 3 November 2014; **published** 21 January 2015. Corresponding Editor: D. P. C. Peters.

† **E-mail:** leahwasser@gmail.com

Introduction

Forested riparian buffers (FRBs) adjacent to aquatic ecosystems, perform critical ecosystem services that both maintain the integrity of adjacent water bodies (e.g., Naiman et al. 1993, Harper and Macdonald 2001, Anderson et al. 2007) and provide important wildlife habitat (Hennings and Edge 2003, Shirley and Smith 2005, Pennington et al. 2008). In recent decades, clearing of riparian forests for agriculture and urban development (e.g., Swift 1984, Naiman and Decamps 1997, Claggett et al. 2010), particularly in headwater streams, which represent the majority of all streams (Leopold et al. 1964), has contributed to significant, broad-scale ecological degradation of downstream ecosystems. Specifically in North America, degradation has been observed in the form of eutrophication in the Chesapeake Bay (Lowrance et al. 1997, Speiran 2010) and hypoxia in the Gulf of Mexico.

Increased agricultural, residential and commercial development has created narrow FRB corridors, predisposing riparian vegetation to 'edge effects' (Saunders et al. 1991, Ferreira et al. 2005), which modify hydrologic, atmospheric and physical processes at the interface between the riparian buffer and the adjacent, disturbed matrix (Harper et al. 2007). Edge effects can significantly influence riparian vegetation structure (Gascon et al. 2000, Apan et al. 2002, Laurance et al. 2002), which in turn can impede the effectiveness of riparian buffers to provide ecosystem services (Lussier et al. 2006, DeWalle 2010, Weller et al. 2010). Commonly observed changes in vegetation structure associated with edge effects include increased wind throw (Liechty and Guldin 2009, Bahuguna et al. 2010), decreased canopy cover and vegetation density (Lopez et al. 2006, Harper et al. 2007) and increased frequency and cover of invasive species (Rodewald 2009, Ives et al. 2011). Overall, FRB and stream integrity is often lowest where developed (agricultural and residential) land use types are dominant (Lussier et al. 2008, Rheinhardt et al. 2012) because of direct land use effects on FRB vegetation structure and spatial configuration.

Degradation of FRB corridors and associated downstream impacts (i.e., large-scale eutrophication, species decline) has driven efforts across the United States to monitor, assess and restore FRB ecosystems (Lowrance et al. 1997, Hall et al. 2009, Claggett et al. 2010). The most common approach has been to characterize edge effects using in situ methods (Harper and Macdonald 2001, Lopez et al. 2006, Harper et al. 2007). However, in situ methods are resource intensive, and cannot be easily applied over wide areas to support the broad-scale mapping that is needed for targeted ecological-based management.

Remote sensing data can be used to extend in situ characterization of FRB condition across broad geographic extents. The most common remote sensing methods applied in riparian studies use spectral sensors, such as Landsat or color-infrared imagery, which record reflectance and absorption of electromagnetic radiation (e.g., Goetz et al. 2003, Johansen et al. 2007, Johansen et al. 2010*b*). However, spectral methods do not capture all of the structural information needed to accurately assess riparian condition (Congalton et al. 2002, Johansen et al. 2010*c*). For example Congalton et al. (2002) observed that forest data derived from classified Landsat data was only in 20–30% agreement with actual forest conditions. Therefore, these and similar spectral datasets, which are often used by decision makers to manage riparian resources, may yield significant or varying error in riparian structure analysis. Congalton et al. (2002) further attributed analysis errors to data resolution; pixels were too large to capture narrow riparian corridors (also observed by Lattin et al. 2004, Johansen et al. 2010*c*). Moreover, spectral data do not provide a direct measure of vertical vegetation structure.

In recent years, high-resolution light detection and ranging (LiDAR) data have been used to map riparian forest structure (Seavy et al. 2009, Arroyo et al. 2010, Johansen et al. 2010*a*, Johansen et al. 2010*c*). A LiDAR system measures the three-dimensional characteristics of vegetation structure by sampling the canopy, understory and ground surface using light reflected from rapidly emitted laser pulses (Baltsavias 1999, Lefsky et al. 1999) at high resolution. LiDAR data support broad-scale mapping of key forest structure metrics including canopy height (Hopkinson et al. 2006), canopy cover (Morsdorf et al. 2006, Hopkinson and Chasmer 2009) and understory vegetation cover (Hill and Broughton 2009). LiDAR-derived estimates of forest structure have

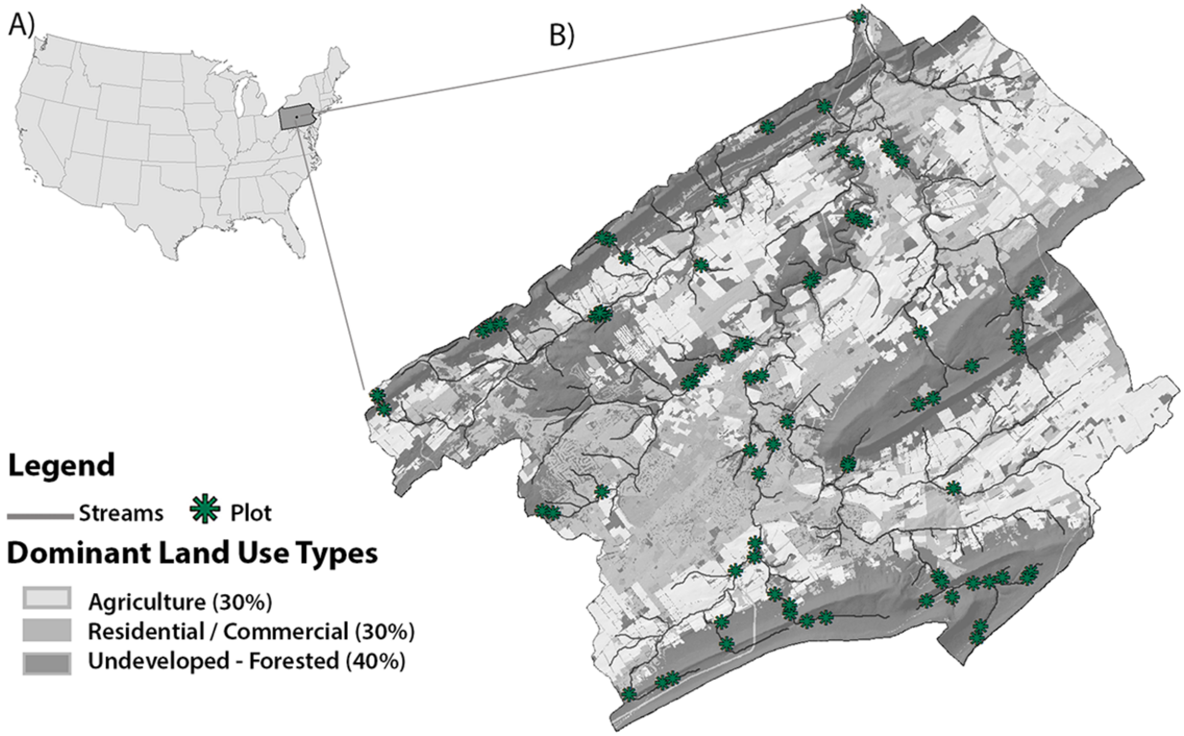

Fig. 1. (A) Context map illustrating the location of Spring Creek watershed within the United States. (B) Land use/land cover map for the Spring Creek watershed.

also been found to be comparable to in situ measurements in riparian areas (Arroyo et al. 2010, Wasser et al. 2013). Furthermore, LiDAR methods are of sufficient resolution to detect narrow riparian corridors and they may provide more accurate estimates of corridor width than spectral methods (Johansen et al. 2010*c*). Therefore the use of LiDAR data for mapping and monitoring riparian buffers can significantly advance our understanding of environmental and anthropogenic impacts to both riparian and other ecological systems over space and time.

The recent availability of wide area coverage LiDAR data in the USA, Australia, Canada and the United Kingdom (Johansen et al. 2010*a*, Hopkinson et al. 2013) has provided an unprecedented opportunity to analyze variability in riparian vegetation structure, continuously, over large areas with heterogeneous land use. However, we are unaware of any studies that have tested and used LiDAR data to quantify the effect of different land use types on both FRB vegetation structure and corridor width throughout a watershed. The specific objectives of this study were thus to use airborne LiDAR data to quantify (1) differences between LiDAR-derived estimates of canopy height, canopy cover and FRB corridor width compared to conventional in situ plot and manual interpretation of spectral imagery; (2) differences in edge vs. interior vegetation height and canopy cover, and FRB width within riparian areas adjacent to residential/commercial, agricultural and undeveloped land use types; and (3) differences in vegetation height and canopy cover within FRBs of varying widths, continuously across an entire watershed.

Methods

Study area

Our study area was the Spring Creek watershed (40.818° N, –77.841° W; Fig. 1), which is located in central Pennsylvania and drains into the Chesapeake Bay. The watershed covers 45,300 ha (175 sq. mi.) and contains 6.4 linear km of first to fourth order streams. Land use in Spring Creek watershed is dominated by undeveloped forested woodlots (40%), agriculture (30%), and urban/suburban development (30%) (Centre County Planning Commission 2007–2012). Riparian vegetation is diverse and includes >60 tree species. Abundant species include black walnut (*Juglans nigra*), red maple (*Acer rubrum*), black cherry (*Prunus serotina*), oak (*Quercus* spp.) and eastern hemlock (*Tsuga canadensis*). Elevation in the watershed ranges from 200 m to 730 m. The climate is temperate and characterized by hot, humid summers and cold winters with average monthly temperatures ranging from −3°C to 22°C and an average annual precipitation of 1,060 mm (Fulton et al. 2005).

Vegetation measurements

We sampled FRB vegetation in July and August, 2010 to validate LiDAR-derived estimates of vegetation structure. A total of 80, 11.3 m radius plots were selected according to vegetation type (deciduous, conifer and mixed) and height (8–30 m) within a 100-m buffer on either side of watershed streams using a random stratified sampling design (Fig. 1). Plots were located using a Trimble GeoXH differential GPS unit (Trimble, GeoExplorer, Idaho, USA). Within each plot, we measured canopy height and we also identified individual tree species. We measured fractional canopy cover (herein "canopy cover", defined between 0 and 1, where 1 is full (100%) canopy cover and 0 is no canopy, or gap) using digital hemispherical photography. For further discussion of sampling design and methods, see Wasser et al. (2013).

LiDAR data collection, processing and analysis

Airborne LiDAR data and 0.8-m resolution color-infrared aerial photography were collected between June 15 and June 18, 2007 using a Leica ALS50 at a flying height of 900 m and a pulse repetition frequency of 100 kHz. Scan angles of up to ±20° resulted in laser return spacing of 0.8 m. We classified laser returns as ground and non-ground using TerraScan (TerraSolid, Finland). We removed returns from buildings using a locally generated building footprint layer (Centre County Planning Office, State College, Pennsylvania, USA). Outlying returns (air and below ground) were also removed. Aboveground pulse return heights were calculated by subtracting the ground elevation derived from a 1-m resolution digital elevation model (DEM) from the elevation of each vegetation return.

We developed an automated algorithm in Matlab (Mathworks, Natick, Massachusetts, USA), to identify 10 m wide by 200 m long sampling transects. Transects were used to estimate vegetation height, canopy cover and corridor width from LiDAR data (Fig. 2) within 5 m wide by 10 m long moving windows. Transects were located every 100 m along either side of the stream and ran perpendicular to stream centerlines. Stream centerlines were identified using USGS National Hydrography Streams Data (NHD, 1:24,000), geometrically corrected to match a 1-m resolution, LiDAR-derived digital elevation model. Transects were classified according to adjacent land use types (agricultural, residential/commercial and undeveloped grassland) using a local land use map (Centre County Planning Agency, 2007, State College, Pennsylvania, USA). Developed land use types had a combination of infrastructure (i.e., buildings and roads) and agricultural fields. Undeveloped land use was grassland, managed by public agencies such as the United States Fish and Wildlife Service. Transects located within larger regions of undeveloped forest (FRB width >200 m) did not cross a disturbed edge and were classified as reference transects. To avoid mixed influences caused by land use conversion (e.g., agriculture to residential or undeveloped forest to agriculture) on vegetation structure, we only analyzed transects in locations where land use was stable for ≥10 years. A total of 5,900 transects were used in the analysis.

Four products were generated from LiDAR data: (1) average canopy height and percent canopy cover in the in situ sampling plots; (2) average canopy height and percent canopy cover for each 5 × 10 m along-transect moving window extending from the stream to the edge of the forested riparian buffer; (3) average canopy height and percent canopy cover for windows located at the stream and matrix edges and interior of each transect; and (4) forested riparian buffer corridor width for each transect.

Average canopy height was estimated from LiDAR data in the in situ plots and within each 5 × 10 m along-transect moving window using the 70th percentile of the vertical vegetation return distribution (e.g., Naesset 2005, Wasser et al.

Fig. 2. An example stream segment and associated transects. The algorithm processed each transect on either side of the stream separately. Each transect was then used to estimate buffer width and quantify vegetation structure throughout the Spring Creek watershed.

2013). Vertical height percentiles were used in lieu of gridded canopy height models because they are more accurate (Gaveau and Hill 2003, Hopkinson et al. 2005). Percentile methods, use the entire vertical profile of laser pulse returns, which better represents the entire vertical profile of vegetation structure from the top of the canopy to the ground (Hopkinson et al. 2005). Percent canopy cover was estimated as the ratio of canopy (return z height > 1.4 m, corresponding to digital hemispherical photography), to all returns in the in situ plots and moving windows (Chasmer et al. 2008).

For each transect, the Matlab algorithm calculated riparian buffer width and along-transect stream edge to matrix edge canopy cover and vegetation height using LiDAR data percentile and ratio methods. To calculate width, the algorithm first identified the forested riparian buffer matrix edge (the interface between forest and an adjacent, non-forest land use type) using a continuous, moving window approach. The moving window began at the stream edge, sampling continuously until it identified a forest gap larger than 10×10 m. This gap was determined by a LiDAR-derived percent canopy cover value of $\leq$30% for vegetation taller than 1.4 m. Thirty percent is the cover threshold for forest as defined by the United Nations Framework Convention on Climate Change and the US National Vegetation Classification standards for forest classification. Finally, transects were classified by width, defined using minimal thresholds for key riparian buffer ecosystem services (0 m, 1–10 m, 10–30 m, 30–50 m, 50–100 m, 100–199 m, 200 m; e.g., Fischer and Fischenich 2000).

Along-transect, LiDAR-derived FRB width estimates, were verified using manual interpretation of high-resolution color-infrared imagery (Congalton et al. 2002, Claggett et al. 2010). For manual interpretation, we visually identified 10 $\times$ 10 m gaps (using the algorithm criteria) in the forest canopy to determine each transect matrix edge. We then used the manually identified edge to calculate transect width. Step differences between riparian forests and grasslands, agricultural areas and urban development, used to identify matrix edges, were easily discernible.

Statistical methods and validation

We first used regression analysis to quantify relationships between (1) LiDAR-derived average canopy height and canopy cover and in situ measurements and (2) LiDAR estimates of FRB width and width estimated using color-infrared imagery. We next quantified along-transect (stream edge to matrix edge) variability in average canopy height and percent canopy cover both within transects of varying widths and adjacent to developed and undeveloped land use types and within 587 reference transects. Reference transects extended 200 m from the stream edge and occurred within a larger (width $>$ 200 m) undisturbed forested area. These transects did not cross an exposed, non-forested edge. We compared vegetation structure within reference transects that did not cross an edge, to non-reference transects (with edges), grouped by width and adjacent land use type to quantify the impacts of adjacent land use and width on vegetation structure.

To quantify vegetation structure along each transect, both reference (containing no edge) and non-reference (containing an edge) transects were divided into 5 $\times$ 10 m wide sections, or 'windows' from stream edge to matrix edge. Within each 5 $\times$ 10 m window, we calculated average canopy height and canopy cover. Direct comparisons of similarity/difference between reference and non-reference transect vegetation structural characteristics were completed using the "magnitude of edge influence" (MEI; Harper et al. 2005, Harper and Macdonald 2011) index:

$$\mathrm{MEI} = \frac{\bar{\imath} - \bar{e}}{\bar{\imath} + \bar{e}} \tag{1}$$

where $\bar{e}$ is the average vegetation structure (height, canopy cover) within each window and $\bar{\imath}$ is average vegetation height/cover determined at the center of the transect (mid-way between stream edge and matrix or adjacent land use type edge). MEI values range between -1 and 1 where values close to 0 indicate similarity between edge and interior conditions and values approaching ± 1 indicate greater differences (or possibly greater disturbance). We calculated mean MEI value (and associated 95% confidence interval) within 5 $\times$ 10 m windows for both non-reference transects grouped by adjacent land use type and buffer width, and reference transects. Mean MEI values were then compared as they occurred along the stream edge and matrix edge, to quantify differences in vegetation structure both within non reference transects as influenced by adjacent land use type and buffer width and compared to reference transects. We hypothesized that MEI values for reference transects would be close to zero throughout the transects as they did not cross an edge. In contrast, we expected that non-reference transects would deviate further from 0 due to edge influences associated with both adjacent land cover types and environmental feedbacks associated with these edges.

Results

LiDAR estimates of riparian buffer vegetation structure and width

LiDAR data methods accurately estimated the most commonly measured riparian forest characteristics: average canopy height, canopy cover and buffer width. LiDAR-derived estimates of average canopy height and canopy cover corresponded well with in situ average canopy height ($R^2 = 0.90$, root mean squared error (RMSE) = 1.68 m; Fig. 3A) and percent canopy cover ($R^2 = 0.59$, RMSE = 2%, Fig. 3B). LiDAR slightly overestimated canopy cover measured using digital hemispherical photography.

There was also a strong correlation between LiDAR-derived estimates and manual photo interpretation estimates of riparian buffer width ($R^2 = 0.89$, RMSE = 26.2 m, Fig. 3C). However, LiDAR compared to manually derived width estimates were less similar in areas that traversed riparian vegetation in varying stages of succession (shrub to mature forest), where manual interpre-

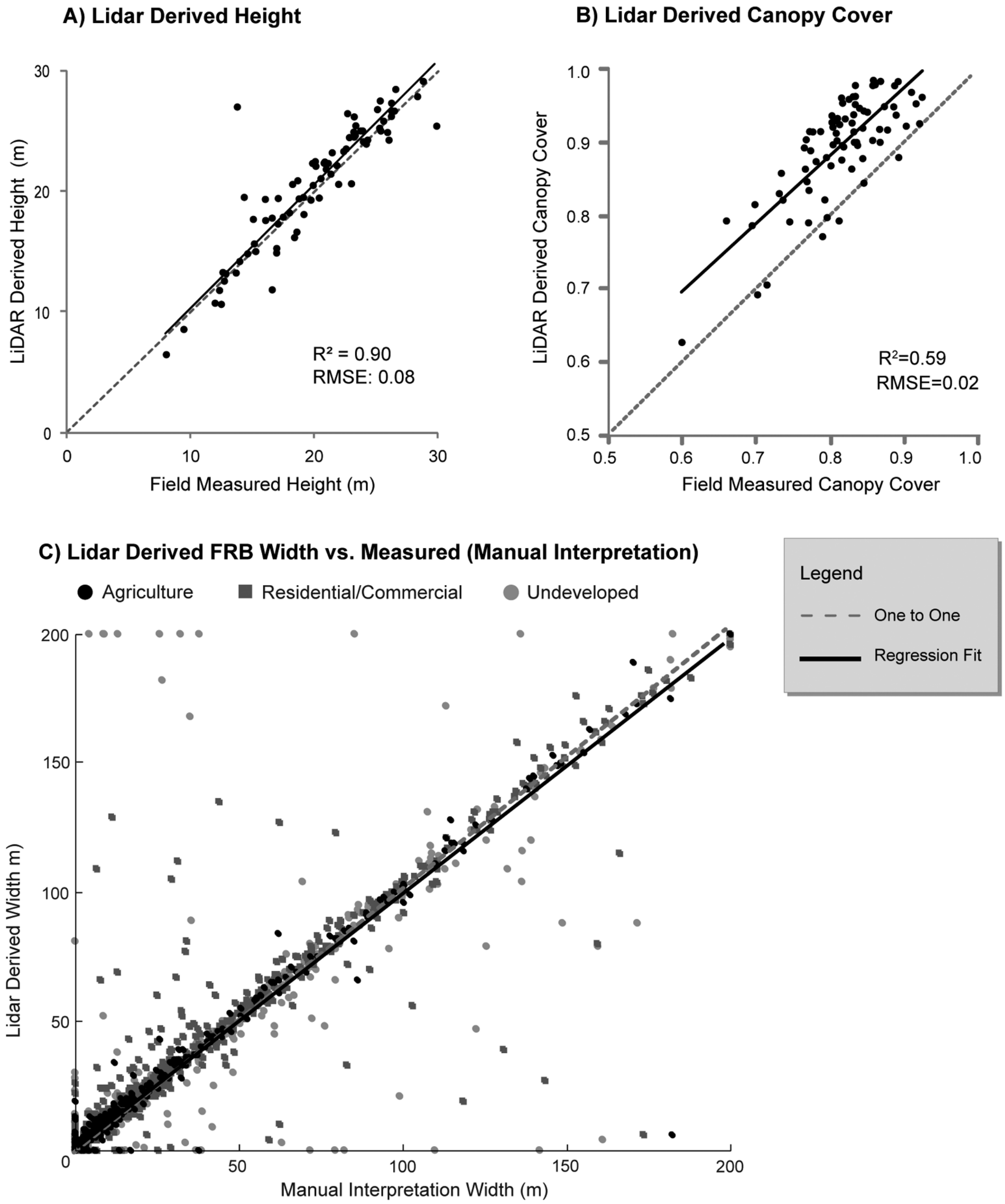

Fig. 3. One to one (dashed line) and linear regression-derived (solid black line) relationships between (A) LiDAR-derived vs. measured average canopy height, (B) LiDAR-derived vs. measured canopy cover (digital hemispherical photography) and (C) LiDAR-derived vs. manual interpretation of CIR imagery-derived corridor width.

tation edge delineation errors were most common. This error, which occurred in 10% of the transects, increased the RMSE (26.2 m) despite a strong linear relationship between LiDAR and manual interpretation-derived FRB width values. Differences between LiDAR and manually derived widths were also caused by buildings not included in the original removal algorithm.

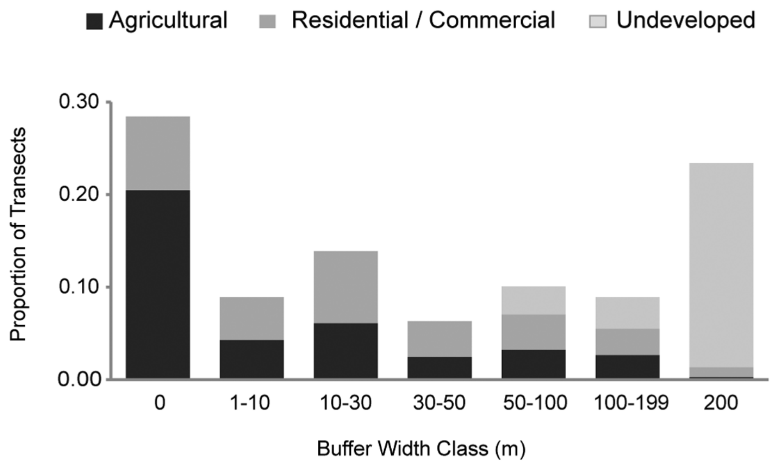

Fig. 4. Proportion of buffers stratified by adjacent land use type (agricultural, residential/commercial and undeveloped) grouped by riparian buffer width class. Width classes were determined using minimal suggested thresholds for key stream ecosystem services.

Land use effects on riparian corridor width and forest structure

Riparian buffer corridor width was strongly associated with adjacent land use type. Agricultural areas had the narrowest buffer corridors. FRBs were widest adjacent to undeveloped land use types (Fig. 4). Approximately 23% of streams had riparian buffers $\geq$200 m wide and 50% of streams had no riparian vegetation or had a riparian buffer $\leq$30 m wide.

There were also relationships between riparian buffer width, adjacent land use type and vegetation height. For all transects, vegetation was on average 2–5 m shorter in narrow vs. wide riparian buffers (Fig. 5). Vegetation was also shortest at the matrix edge and gradually increased in height to the stream edge by an average of 5–8 m, regardless of adjacent land use type. Vegetation height near the stream edge did vary by adjacent land use type. Vegetation towards the stream edge was taller within buffers adjacent to undeveloped land use types (average canopy height range = 18.9–21.2 m) compared with stream edge vegetation next to agricultural (average canopy height range = 14.2–16.8 m) and residential/commercial (average canopy height range = 15.9–16.7 m) land use types.

For all transects, percent canopy cover was nominally higher at the stream edge, relative to the matrix edge and within undeveloped compared to developed (agricultural, residential/commercial) land use types (Fig. 6). However, nominal edge/interior differences in percent canopy cover were within the 10–20% range of error expected for percent canopy cover estimates (Chen et al. 2006, Wasser et al. 2013).

Relative variability, calculated using the magnitude of edge influence (MEI) further demonstrated relationships between edge and interior vegetation structure as it varied with adjacent land use type. For all non-reference transects, vegetation height towards the stream edge of each transect was, on average, similar to vegetation height at each transect interior (stream edge MEI range = 0.0 to -0.1) and within the range of variability observed for reference transects (MEI average range = 0.0 to -0.1; Fig. 7). In contrast, vegetation towards the matrix edge was consistently shorter than vegetation at the interior (MEI average range = 0.1–0.2) with values outside the range of those observed in reference transects. MEI values occurring within 5–30 m from the matrix edge along both developed and undeveloped land use types were also similar, particularly within corridors $<$100 m wide, suggesting that width and associated environmental effects of edge exposure may play a significant role in the MEI on canopy height along the matrix edge (rather than land use type).

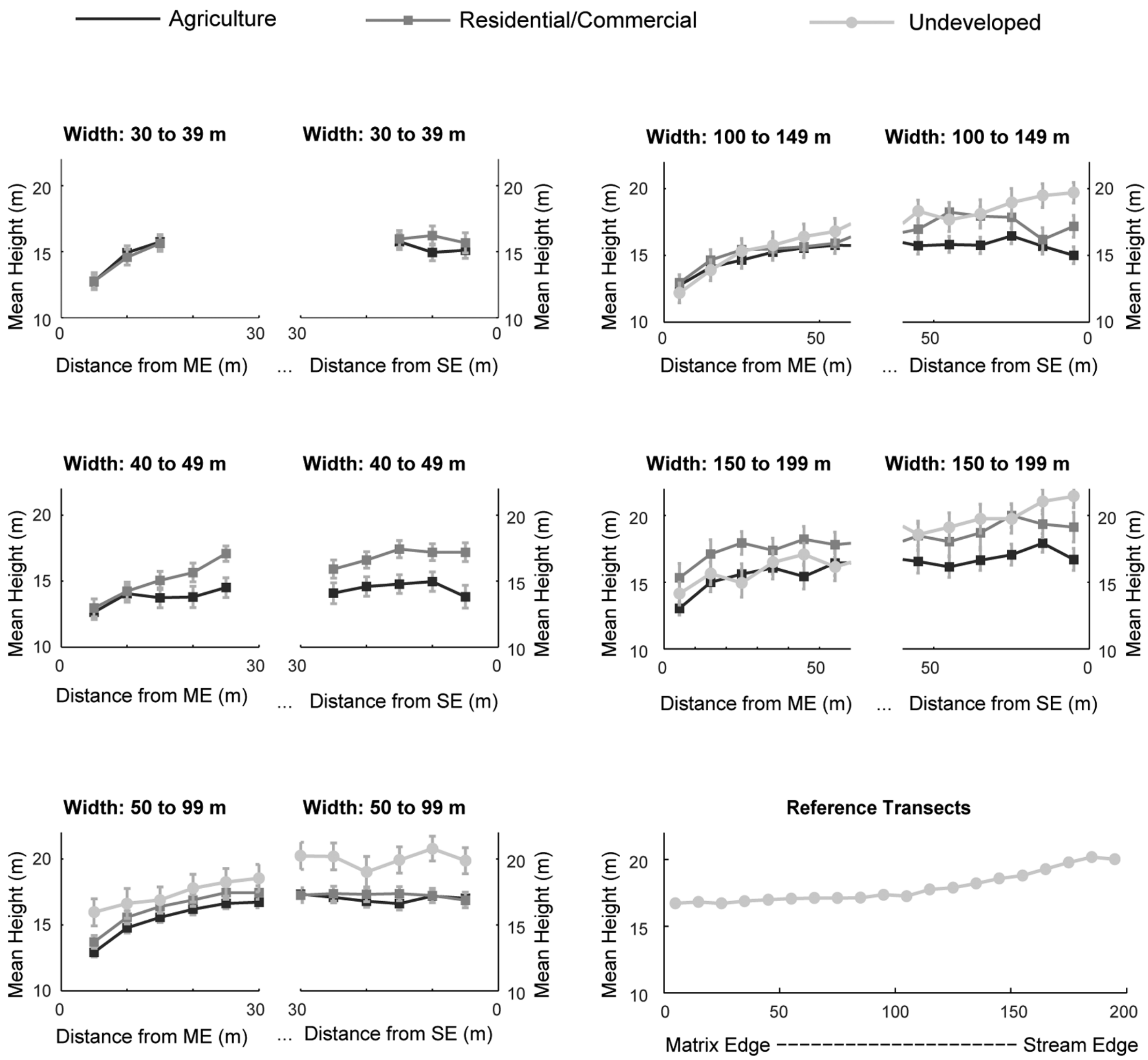

Fig. 5. Mean canopy height (H) and 95% confidence interval, for each 5×10 m window within transects. Mean values are plotted starting at the matrix edge (ME) to the transect interior and starting at the stream edge (SE) to the transect interior. Results are grouped by buffer width class and adjacent land use type (agriculture, residential/commercial, undeveloped). Reference transects are located within a larger region of undeveloped forest.

Discussion

In this study, a transect-based approach using airborne LiDAR data was used to identify relationships between adjacent land use type and three commonly measured indicators of stream ecological integrity: FRB width, vegetation height and canopy cover over a wide area. We demonstrated that LiDAR data methods are comparable to and can extend conventional in situ and manual imagery interpretation methods that are used to estimate these indicators. We further presented an automated, transect based approach using LiDAR methods to efficiently characterize vegetation structure and corridor width along thousands of transects to identify how adjacent land use affects variability in canopy height, canopy cover and FRB width.

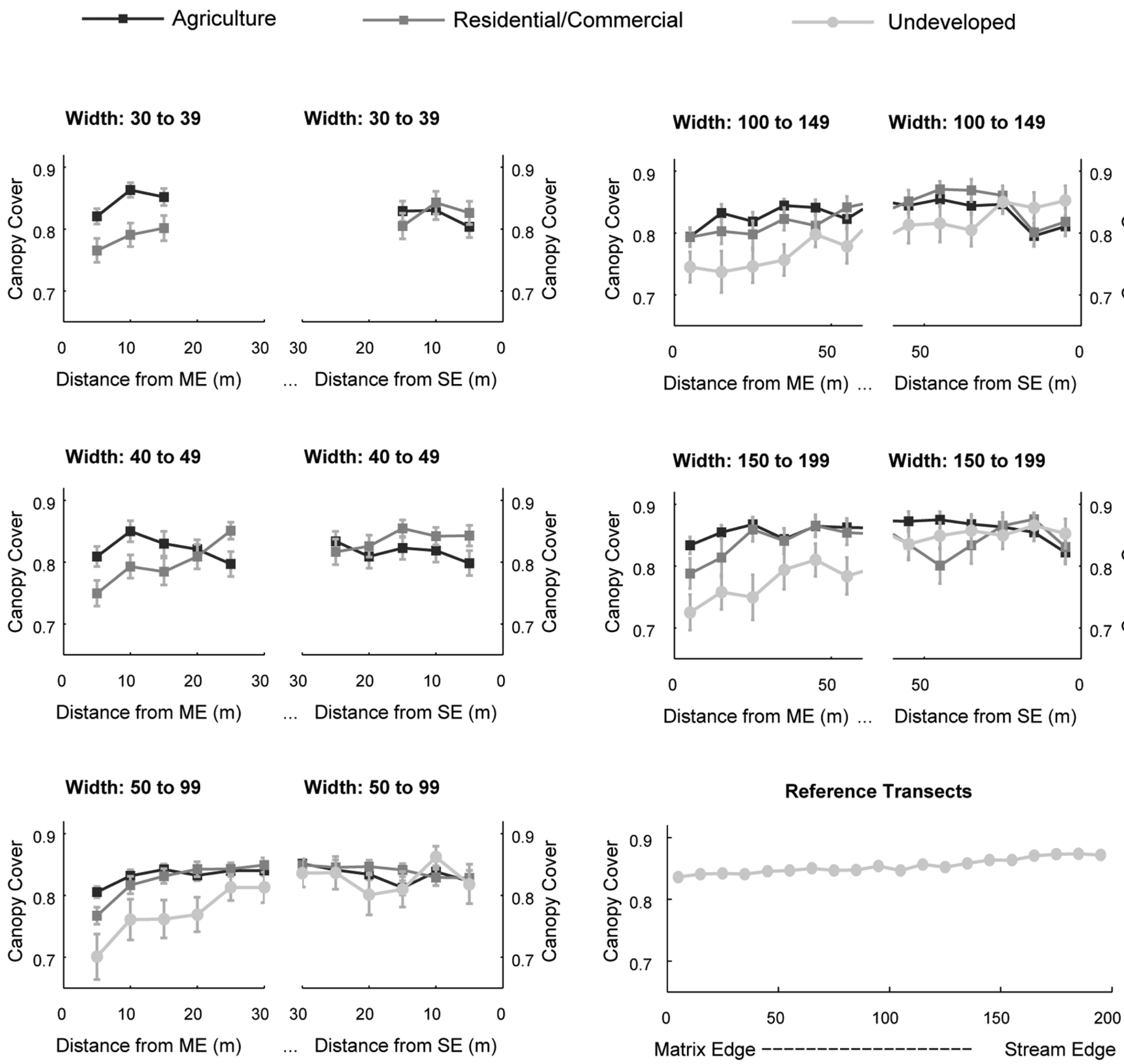

Fig. 6. Mean canopy cover and 95% confidence interval, for each 5 × 10 m window within transects. Mean values are plotted starting at the matrix edge (ME) to the transect interior and starting at the stream edge (SE) to the transect interior. Results are grouped by buffer width class and adjacent land use type (agriculture, residential/commercial, undeveloped). Reference transects are located within a larger region of undeveloped forest.

This LiDAR approach could be highly valuable in identifying stream reaches that are most vulnerable to impacts of adjacent land use in support of ecological analysis across large regions such as the Chesapeake Bay Watershed.

LiDAR methods effectively estimate riparian vegetation height, canopy cover and corridor width

Our results demonstrate that LiDAR technology can be effectively used for wide-area mapping of riparian vegetation structure when compared with in situ plot measurements. Few studies have used LiDAR data to map riparian vegetation structure. However, there are some

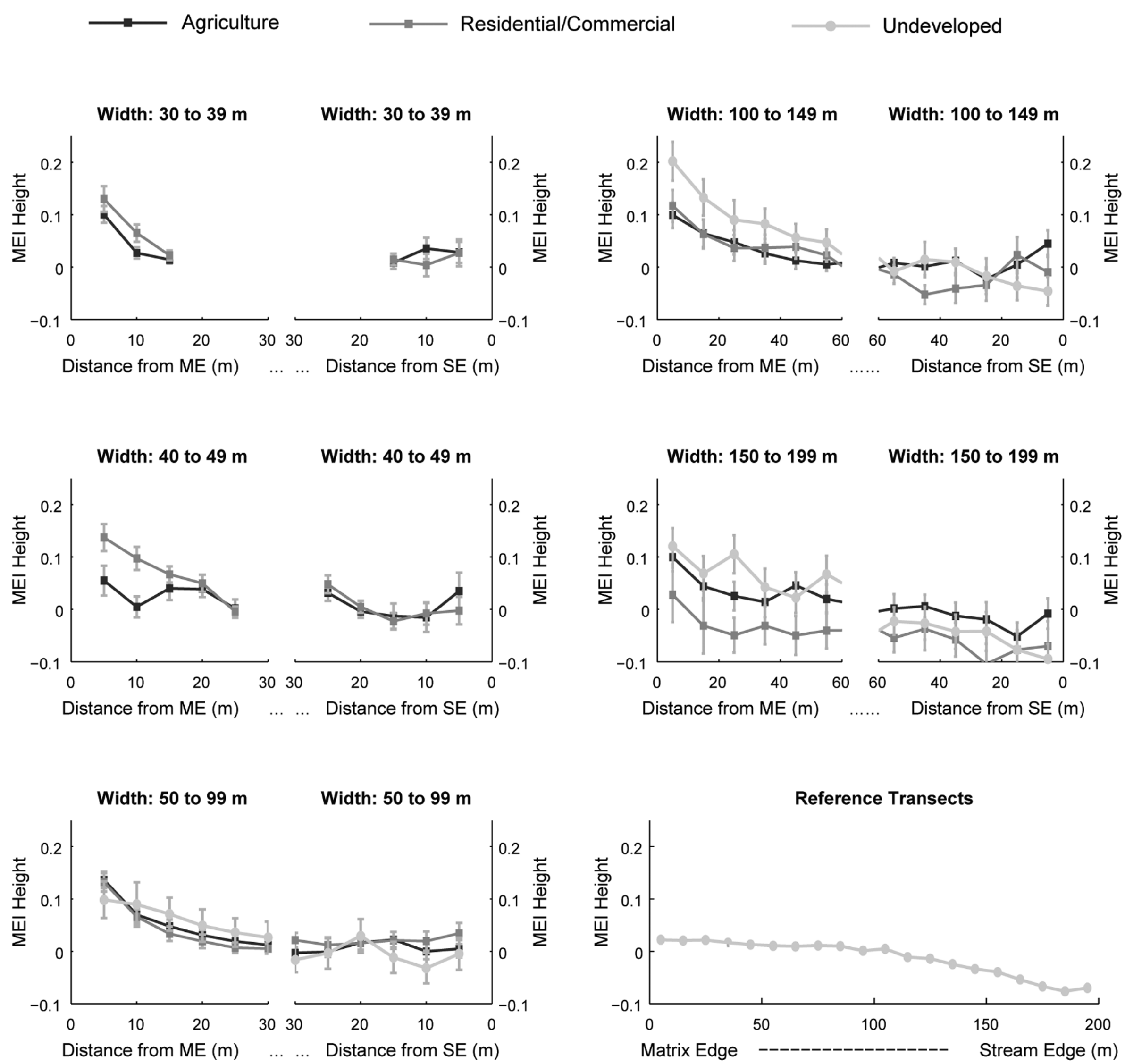

Fig. 7. Mean magnitude of edge influence (MEI) for canopy height (H) and 95% confidence interval, for each 5 $\times$ 10 m window within transects. Mean values are plotted starting at the matrix edge (ME) to the transect interior and starting at the stream edge (SE) to the transect interior. Results are grouped by buffer width class and adjacent land use type (agriculture, residential/commercial, undeveloped). Reference transects are located within a larger region of undeveloped forest.

challenges to consider. For example, Hopkinson et al. (2005) compared riparian and forest vegetation height derived using gridding vs. frequency distribution LiDAR methods with in situ measurements. While strong correspondence between LiDAR and in situ measurements was observed in this study, they found that LiDAR methods typically underestimated short shrub and herbaceous vegetation, especially within saturated riparian/aquatic areas. Heights of shrub and tree species within drier environments could be accurately characterized using LiDAR data. Hutton and Brazier (2012) attributed potential underestimation of vegetation height using LiDAR to errors in digital elevation models. They observed that digital elevation model height values, which are used to derive vegetation height from LiDAR data, were often overestimated in areas with dense vegetation yielding low LiDAR-derived height values. Johansen et al. (2010*c*) also found that LiDAR data accurately characterized streambed width, riparian buffer width and canopy cover. In this study, we found strong, linear correspondence between vegetation height from LiDAR data and in situ measurements across an entire watershed. We also found strong linear relationships between LiDAR- and in situ-derived canopy cover, although LiDAR methods overestimated canopy cover (see also Morsdorf et al. 2006, Hopkinson and Chasmer 2009). Overestimates of canopy cover using LiDAR data have been attributed to reduced ground penetration and saturation of vegetation returns in areas of dense vegetation.

LiDAR-derived buffer width also corresponded strongly with width measured using conventional aerial photography indicating that automated methods are an efficient alternative to resource intensive manual interpretation methods (e.g., Schuft et al. 1999, Snyder et al. 2003, Snyder et al. 2005, Claggett et al. 2010). Manual interpretation is also susceptible to bias. Automated methods remove bias associated with individual interpreter error and standardize detection of small canopy gaps that may be missed in manual interpretation of corridor boundaries. Automation also permits vegetation characterization along thousands of transects, continuously, supporting improved identification of stream reaches that might be most vulnerable to adjacent land use impacts. Other studies have also found that LiDAR data can be used to derive buffer width. For example, in a comparison of spectral methods with LiDAR methods developed for Australian riparian zones, Johansen et al. (2010*c*) found that LiDAR estimates of riparian buffer width strongly approximated in situ conditions ($R^2 = 0.91$) and were more accurate than spectral methods. They concluded that LiDAR data may be ideal for mapping riparian areas over large extents. Similarly, Arroyo et al. (2010) observed improved correspondence between measured width and width derived using a combination of LiDAR data and spectral (QuickBird-2) data within Australian tropical savannas/riparian areas ($R^2 = 0.82$).

Relationships between land use, riparian buffer width and vegetation structure

Our study also demonstrates that LiDAR methods can efficiently and effectively be used to quantify relationships between riparian buffer width, edge and interior vegetation structure and adjacent land use type across thousands of transects. A similar analysis using in situ methods would be cost prohibitive.

It is well documented that within a human impacted landscape, wider buffers are needed to provide many key ecosystem services that support aquatic ecosystem health (Castelle et al. 1994, Fischer and Fischenich 2000, Shirley 2004, DeWalle 2010). In our study, riparian corridors were narrowest where land use was the most developed (agriculture, commercial/residential development). Specifically, half of all corridors next to developed land use types were narrow (width < 30 m) or had no riparian vegetation (width $= 0$). The narrowest corridors were adjacent to agricultural areas which yield higher levels of disturbance inputs to streams. Such pollutant inputs, water runoff and sedimentation could affect the health and structure of both the adjacent riparian area and the stream channel that it shields from adjacent development (e.g., Jones and Holmes 1985, Jones et al. 2001, He et al. 2011) in these areas. Since these locations can be clearly identified, transect by transect, using methods presented in this study, they can be target areas for monitoring and restoration. In contrast, few streams had buffers $\geq$200 m wide, a width that supports most key ecosystem services. These areas could be targeted as conservation zones to preserve ecological integrity.

We also found that riparian vegetation was taller within: (1) wider compared to narrow corridors; (2) buffers adjacent to undeveloped vs. developed land use types, and (3) along the stream edge and interior vs. the matrix edge within corridors wider than 50 m. The relationship between tall vegetation, wider corridors and undeveloped land use types was also observed in situ. Corridors adjacent to undeveloped land use types often contained taller species including oaks (*Quercus* spp.), maples (*Acer* spp.) and hemlocks (*T. canadensis*) while corridors adjacent to developed land use types often contained shorter species including black walnut (*J. nigra*) and red maple (*A. rubrum*). Taller vegetation within larger patches has also been observed in other non-riparian forests (e.g., Mohandass and Davidar 2010, Brearley et al. 2011) and may be attributed in part to increased structural diversity (and increased species diversity) associated with species-area relationships that are expected within larger patches (MacArthur and Wilson 1967).

Several studies have used in situ methods to quantify the impacts of adjacent land use on vegetation structure in both riparian and non-riparian forests. These relationships are important for quantifying pre- and post-disturbance influences on aquatic ecosystem health (Langer et al. 2008). For example, similar to our findings, Mohandass and Davidar (2010) observed that larger patches were often less disturbed and thus contained older, more mature and thus taller vegetation using in situ methods. In contrast to our findings of nominal variability in canopy cover associated with land use, Langer et al. (2008) found decreased canopy cover within the upper tiers of a riparian buffer adjacent to a managed forest, compared to an unmanaged forest. Similarly, Rheinhardt et al. (2012) observed that riparian forest cover (and associated biomass) is often lower within forested riparian buffer corridors adjacent to developed compared to undeveloped land use types.

We also observed differences between matrix edge, stream edge and interior vegetation height. Throughout the watershed, riparian vegetation at the matrix edge was consistently (16–18%) shorter than vegetation at the transect interior whereas vegetation along the stream edge was similar in height (within 2%) to vegetation at the interior. We interpret this pattern of edge vs. interior differences in vegetation height, which occurred across all buffer width classes as being related to local physical and environmental processes (e.g., wind throw, soil moisture, tree mortality; Lopez et al. 2006, Liechty and Guldin 2009, Bahuguna et al. 2010), and to anthropogenic modifications not observed in remote sensing data (e.g., nutrient runoff, land use impacts, mowing, plowing; Ferreira et al. 2005), which may manifest similarly along forest edges, regardless of corridor width or adjacent land use type. Lopez et al. (2006) attributed increased tree mortality and wind throw at the matrix compared to the stream edge to adaptations that streamside riparian vegetation may have to a natural stream edge environment (Harper et al. 2007). Within tropical forests, Ribeiro et al. (2009) found that vegetation structure (average tree diameter, height, density and number of snags) was similar within both the matrix and naturally occurring edges. In slight contrast to the findings in our study where variability in canopy cover was nominal, Lopez et al. (2006) observed small differences in edge vs. interior vegetation canopy cover within riparian forests, with larger edge effect influence observed in non-riparian forests. They attributed smaller edge influence effects within riparian forests in part to riparian vegetation having pre-existing, stable internal (streamside) edges.

Conclusions and Implications

Characterizing riparian buffer conditions is a key first step in ecologically sound resource management and planning for maintaining aquatic ecosystem integrity. It is especially important in headwater watersheds such as Spring Creek where increased disturbance compounds downstream, significantly impacting water quality, quantity and wildlife habitat (Alexander et al. 2007, Preston et al. 2011).

This study demonstrates that automated LiDAR processing methods can be used to accurately quantify riparian buffer width, vegetation height and canopy cover at spatial extent and resolutions not possible using conventional in situ and manual methods (Arroyo et al. 2010, Johansen et al. 2010*c*, Fernandes et al. 2011). To conduct a similar study using plot and transect measurements could take months to years, and would be prohibitively expensive. Moreover, the methods are reproducible over time and can support high-resolution change detection. Increasingly, studies find that high-resolution remote sensing data, such as LiDAR, can be used as a cost effective alternative to in situ measurements of vegetation structure (Johansen et al. 2010*c*, Fernandes et al. 2011). However, large LiDAR datasets demand significant processing resources (hardware, software and time) and necessitate developing processing methods and testing against standard measurement techniques (such as the methods presented in this study) before they can be widely applied.

Development of these methods is also integral to efforts to inventory, assess and track change over time to support conservation, restoration, and broad-scale ecological planning. For example, our study quantitatively demonstrates that streams most vulnerable to degradation are located near developed land use types. These areas could be prioritized and targeted for restoration in the context of regional environmental planning efforts to improve aquatic ecosystem health in the highly impacted Chesapeake Bay watershed.

Acknowledgments

The authors would like to thank Dr. Chris Hopkinson (University of Lethbridge) for use of LiDAR processing tools and Dr. Robert Brooks (Penn State Riparia) for providing the LiDAR dataset and CIR imagery used in this study. Leah Wasser has been supported in part by a grant from the National Consortium for Rural Geospatial Innovations in America, the Penn State Geospatial Technology Program and the Penn State Vegetation Dynamics Laboratory. She is currently employed at the National Ecological Observatory Network in Boulder, Colorado. The authors also wish to thank the anonymous reviewers for suggestions and comments that have contributed greatly to this manuscript.

Literature Cited

Alexander, R., E. Boyer, R. Smith, G. Schwartz, and R. Moore. 2007. The role of headwater streams in downstream water quality. Journal of the American Water Resources Association 43:41–59.

Anderson, P. D., D. J. Larson, and S. S. Chan. 2007. Riparian buffer and density management influences on microclimate of young headwater forests of western Oregon. Forest Science 53:254–269.

Apan, A. A., S. R. Raine, and M. S. Paterson. 2002. Mapping and analysis of changes in the riparian landscape structure of the Lockyer Valley catchment, Queensland, Australia. Landscape and Urban Planning 59:43–57.

Arroyo, L. A., K. Johansen, J. Armston, and S. Phinn. 2010. Integration of LiDAR and QuickBird imagery for mapping riparian biophysical parameters and land cover types in Australian tropical savannas. Forest Ecology and Management 259:598–606.

Bahuguna, D., S. J. Mitchell, and Y. Miquelajauregui. 2010. Windthrow and recruitment of large woody debris in riparian stands. Forest Ecology and Management 259:2048–2055.

Baltsavias, E. P. 1999. Airborne laser scanning: basic relations and formulas. Journal of Photogrammetry and Remote Sensing 54:199–214.

Brearley, G., A. Bradley, S. Bell, and C. McAlpine. 2011. Change in habitat resources and structure near urban edges and its influence on the squirrel glider (*Petaurus norfolcensis*) in southeast Queensland, Australia. Austral Ecology 36:425–432.

Castelle, A. J., A. W. Johnson, and C. Connolly. 1994. Wetland and stream buffer size requirements: a review. Journal of Environmental Quality 23:878–882.

Centre County Planning Commission. 2007–2012. Centre County landuse data. Centre County Planning Commission, Bellefonte, Pennsylvania, USA.

Chasmer, L., C. Hopkinson, P. Treitz, H. McCaughey, A. Barr, and A. Black. 2008. A LiDAR-based hierarchical approach for assessing MODIS fPAR. Remote Sensing of Environment 112:4344–4357.

Chen, J. M., A. Govind, O. Sonnentag, Y. Q. Zhang, A. Barr, and B. Amiro. 2006. Leaf area index measurements at Fluxnet-Canada forest sites. Agricultural and Forest Meteorology 140:257–268.

Claggett, P. R., J. A. Okay, and S. V. Stehman. 2010. Monitoring regional riparian forest cover change using stratified sampling and multiresolution imagery. Journal of the American Water Resources Association 46:334–343.

Congalton, R. G., K. Birch, R. Jones, and J. Schriever. 2002. Evaluating remotely sensed techniques for mapping riparian vegetation. Computers and Electronics in Agriculture 37:113–126.

DeWalle, D. R. 2010. Modeling stream shade: riparian buffer height and density as important as buffer width. Journal of the American Water Resources Association 46:323–333.

Fernandes, M. R., F. C. Aguiar, and M. T. Ferreira. 2011. Assessing riparian vegetation structure and the influence of land use using landscape metrics and geostatistical tools. Landscape and Urban Planning 99:166–177.

Ferreira, M. T., F. C. Aguiar, and C. Nogueira. 2005. Changes in riparian woods over space and time: Influence of environment and land use. Forest Ecology and Management 212:145–159.

Fischer, R. A. and J. C. Fischenich. 2000. Design recommendations for riparian corridors and vegetated buffer strips. Technical Note. Army Engineer Waterways Experiment Station, Vicksburgh, Mississippi, USA.

Fulton, J., E. Koerkle, S. McAuthor, S. Hoffman, and L. Zarr. 2005. Hydrogeologic setting and conceptual hydrological model of Spring Creek Basin, Centre County, Pennsylvania. Scientific Investigations Report 2005-5091. USGS Information Services, Denver, Colorado, USA. http://pubs.usgs.gov/sir/2005/5091/sir2005-5091.pdf

Gascon, C., G. B. Williamson, and G. A. B. da Fonseca.

2000. Ecology: Receding forest edges and vanishing reserves. Science 288:1356–1358.

Gaveau, D. L. A. and R. A. Hill. 2003. Quantifying canopy height underestimation by laser pulse penetration in small-footprint airborne laser scanning data. Canadian Journal of Remote Sensing 29:650–657.

Goetz, S. J., R. K. Wright, A. J. Smith, E. Zinecker, and E. Schaub. 2003. IKONOS imagery for resource management: Tree cover, impervious surfaces, and riparian buffer analyses in the mid-Atlantic region. Remote Sensing of Environment 88:195–208.

Hall, R., R. Watkins, D. Heggem, K. Jones, P. Kaufmann, S. Moore, and S. Gregory. 2009. Quantifying structural physical habitat attributes using LIDAR and hyperspectral imagery. Environmental Monitoring and Assessment 159:63–83.

Harper, K. A. and S. E. Macdonald. 2001. Structure and composition of riparian boreal forest: New methods for analyzing edge influence. Ecology 82:649–659.

Harper, K. A. and S. E. Macdonald. 2011. Quantifying distance of edge influence: a comparison of methods and a new randomization method. Ecosphere 2:94.

Harper, K. A., S. E. Macdonald, P. J. Burton, J. Q. Chen, K. D. Brosofske, S. C. Saunders, E. S. Euskirchen, D. Roberts, M. S. Jaiteh, and P. A. Esseen. 2005. Edge influence on forest structure and composition in fragmented landscapes. Conservation Biology 19:768–782.

Harper, K. A., L. Mascarua-Lopez, S. E. Macdonald, and P. Drapeau. 2007. Interaction of edge influence from multiple edges: examples from narrow corridors. Plant Ecology 192:71–84.

He, B., S. Kanae, T. Oki, Y. Hirabayashi, Y. Yamashiki, and K. Takara. 2011. Assessment of global nitrogen pollution in rivers using an integrated biogeochemical modeling framework. Water Research 45:2573–2586.

Hennings, L. A. and W. D. Edge. 2003. Riparian bird community structure in Portland, Oregon: Habitat, urbanization, and spatial scale patterns. Condor 105:288–302.

Hill, R. A. and R. K. Broughton. 2009. Mapping the understorey of deciduous woodland from leaf-on and leaf-off airborne LiDAR data: A case study in lowland Britain. Journal of Photogrammetry and Remote Sensing 64:223–233.

Hopkinson, C. and L. Chasmer. 2009. Testing LiDAR models of fractional cover across multiple forest ecozones. Remote Sensing of Environment 113:275–288.

Hopkinson, C., L. Chasmer, D. Colville, R. Fournier, R. Hall, J. Luther, T. Milne, R. Petrone, and B. St-Onge. 2013. Moving towards consistent ALS monitoring of forest attributes across Canada; the 'C-CLEAR' approach. Photogrammetric Engineering and Remote Sensing National LiDAR Special Issue 79:159–173.

Hopkinson, C., L. Chasmer, K. Lim, P. Treitz, and I. Creed. 2006. Towards a universal lidar canopy height indicator. Canadian Journal of Remote Sensing 32:139–152.

Hopkinson, C., L. E. Chasmer, G. Sass, I. F. Creed, M. Sitar, W. Kalbfleisch, and P. Treitz. 2005. Vegetation class dependent errors in lidar ground elevation and canopy height estimates in a boreal wetland environment. Canadian Journal of Remote Sensing 31:191–206.

Hutton, C. and R. Brazier. 2012. Quantifying riparian zone structure from airborne LiDAR: Vegetation filtering, anisotropic interpolation, and uncertainty propagation. Journal of Hydrology 442–443:36–45.

Ives, C. D., G. C. Hose, D. A. Nipperess, and M. P. Taylor. 2011. The influence of riparian corridor width on ant and plant assemblages in northern Sydney, Australia. Urban Ecosystems 14:1–16.

Johansen, K., L. A. Arroyo, J. Armston, S. Phinn, and C. Witte. 2010*a*. Mapping riparian condition indicators in a sub-tropical savanna environment from discrete return LiDAR data using object-based image analysis. Ecological Indicators 10:796–807.

Johansen, K., L. A. Arroyo, S. Phinn, and C. Witte. 2010*b*. Comparison of geo-object based and pixel-based change detection of riparian environments using high spatial resolution multi-spectral imagery. Photogrammetric Engineering and Remote Sensing 76:123–136.

Johansen, K., N. C. Coops, S. E. Gergel, and Y. Stange. 2007. Application of high spatial resolution satellite imagery for riparian and forest ecosystem classification. Remote Sensing of Environment 110:29–44.

Johansen, K., S. Phinn, and C. Witte. 2010*c*. Mapping of riparian zone attributes using discrete return LiDAR, QuickBird and SPOT-5 imagery: Assessing accuracy and costs. Remote Sensing of Environment 114:2679–2691.

Jones, K. B., A. C. Neale, M. S. Nash, R. D. Van Remortel, J. D. Wickham, K. H. Riitters, and R. V. O'Neill. 2001. Predicting nutrient and sediment loadings to streams from landscape metrics: A multiple watershed study from the United States Mid-Atlantic Region. Landscape Ecology 16:301–312.

Jones, R. C. and B. H. Holmes. 1985. Effects of land use practices on water resources in Virginia USA. Virginia Polytechnic Institute and State University Water Resources Research Center Bulletin 1–114.

Langer, E. R., G. A. Steward, and M. O. Kimberley. 2008. Vegetation structure, composition and effect of pine plantation harvesting on riparian buffers in New Zealand. Forest Ecology and Management

256:949–957.

Lattin, P. D., P. J. Wigington, T. J. Moser, B. E. Peniston, D. R. Lindeman, and D. R. Oetter. 2004. Influence of remote sensing imagery source on quantification of Riparian land cover and use. Journal of the American Water Resources Association 40:215–227.

Laurance, W. F., T. E. Lovejoy, H. L. Vasconcelos, E. M. Bruna, R. K. Didham, P. C. Stouffer, C. Gascon, R. O. Bierregaard, S. G. Laurance, and E. Sampaio. 2002. Ecosystem decay of Amazonian forest fragments: A 22-year investigation. Conservation Biology 16:605–618.

Lefsky, M. A., W. B. Cohen, S. A. Acker, G. G. Parker, T. A. Spies, and D. Harding. 1999. Lidar remote sensing of the canopy structure and biophysical properties of Douglas-fir western hemlock forests. Remote Sensing of Environment 70:339–361.

Leopold, L. B., M. G. Wolman, and J. P. Miller. 1964. Fluvial processes in geomorphology. W. H. Freeman, San Fransisco, California, USA.

Liechty, H. O. and J. M. Guldin. 2009. Structure and composition of streamside management zones following reproduction cutting in shortleaf pine stands. Forest Ecology and Management 258:1407–1413.

Lopez, L. E. M., K. A. Harper, and P. Drapeau. 2006. Edge influence on forest structure in large forest remnants, cutblock separators, and riparian buffers in managed black spruce forests. Ecoscience 13:226–233.

Lowrance, R., L. S. Altier, J. D. Newbold, R. R. Schnabel, P. M. Groffman, J. M. Denver, D. L. Correll, J. W. Gilliam, J. L. Robinson, R. B. Brinsfield, K. W. Staver, W. Lucas, and A. H. Todd. 1997. Water quality functions of riparian forest buffers in Chesapeake Bay watersheds. Environmental Management 21:687–712.

Lussier, S. M., S. N. da Silva, M. Charpentier, J. F. Heltshe, S. M. Cormier, D. J. Klemm, M. Chintala, and S. Jayaraman. 2008. The influence of suburban land use on habitat and biotic integrity of coastal Rhode Island streams. Environmental Monitoring and Assessment 139:119–136.

Lussier, S. M., R. W. Enser, S. N. Dasilva, and M. Charpentier. 2006. Effects of habitat disturbance from residential development on breeding bird communities in riparian corridors. Environmental Management 38:504–521.

MacArthur, R. H. and E. O. Wilson. 1967. The theory of island biogeography. Princeton University Press, Princeton, New Jersey, USA.

Mohandass, D. and P. Davidar. 2010. The relationship between area, and vegetation structure and diversity in montane forest (shola) patches in southern India. Plant Ecology & Diversity 3:67–76.

Morsdorf, F., B. Kötz, E. Meier, K. I. Itten, and B. Allgöwer. 2006. Estimation of LAI and fractional cover from small footprint airborne laser scanning data based on gap fraction. Remote Sensing of Environment 104:50–61.

Naesset, E. 2005. Assessing sensor effects and effects of leaf-off and leaf-on canopy conditions on biophysical stand properties derived from small-footprint airborne laser data. Remote Sensing of Environment 98:356–370.

Naiman, R. J. and H. Decamps. 1997. The ecology of interfaces: riparian zones. Annual Review of Ecology and Systematics 28:621–658.

Naiman, R. J., H. Decamps, and M. Pollock. 1993. The role of riparian corridors in maintaining regional biodiversity. Ecological Applications 3:209–212.

Pennington, D. N., J. Hansel, and R. B. Blair. 2008. The conservation value of urban riparian areas for landbirds during spring migration: Land cover, scale, and vegetation effects. Biological Conservation 141:1235–1248.

Preston, S. D., R. B. Alexander, G. E. Schwarz, and C. G. Crawford. 2011. Factors affecting stream nutrient loads: a synthesis of regional sparrow model results for the continental United States. Journal of the American Water Resources Association 47:891–915.

Rheinhardt, R., M. Brinson, G. Meyer, and K. Miller. 2012. Integrating forest biomass and distance from channel to develop an indicator of riparian condition. Ecological Indicators 23:46–55.

Ribeiro, M. T., F. N. Ramos, and F. A. M. Dos Santos. 2009. Tree structure and richness in an Atlantic forest fragment: distance from anthropogenic and natural edges. Revista Arvore 33:1123–1132.

Rodewald, A. D. 2009. Urban-associated habitat alteration promotes brood parasitism of Acadian Flycatchers. Journal of Field Ornithology 80:234–241.

Saunders, D. A., R. J. Hobbs, and C. R. Margules. 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology 5:18–32.

Schuft, M. J., T. J. Moser, P. J. Wigington, D. L. Stevens, L. S. McAllister, S. S. Chapman, and T. L. Ernst. 1999. Development of landscape metrics for characterizing riparian-stream networks. Photogrammetric Engineering and Remote Sensing 65:1157–1167.

Seavy, N. E., J. H. Viers, and J. K. Wood. 2009. Riparian bird response to vegetation structure: a multiscale analysis using LiDAR measurements of canopy height. Ecological Applications 19:1848–1857.

Shirley, S. 2004. The influence of habitat diversity and structure on bird use of riparian buffer strips in coastal forests of British Columbia, Canada. Canadian Journal of Forest Research 34:1499–1510.

Shirley, S. M. and J. N. M. Smith. 2005. Bird community structure across riparian buffer strips of varying width in a coastal temperate forest.

Biological Conservation 125:475–489.

Snyder, C. D., J. A. Young, R. Villella, and D. P. Lemarie. 2003. Influences of upland and riparian land use patterns on stream biotic integrity. Landscape Ecology 18:647–664.

Snyder, M. N., S. J. Goetz, and R. K. Wright. 2005. Stream health rankings predicted by satellite derived land cover metrics. Journal of the American Water Resources Association 41:659–677.

Speiran, G. K. 2010. Effects of groundwater-flow paths on nitrate concentrations across two riparian forest corridors. Journal of the American Water Resources Association 46:246–260.

Swift, B. L. 1984. Status of riparian ecosystems in the United States. Water Resources Bulletin 20:223–228.

Wasser, L., R. Day, L. Chasmer, and A. Taylor. 2013. Influence of vegetation structure on LiDAR-derived canopy height and fractional cover in forested riparian buffers during leaf-off and leaf-on conditions. PLoS ONE.

Weller, D. E., M. E. Baker, and T. E. Jordan. 2010. Effects of riparian buffers on nitrate concentrations in watershed discharges: new models and management implications. Ecological Applications 21:1679–1695.

Supplemental Material

Supplement

Matlab files to calculate transects and measure vegetation height and cover along each stream reach (*Ecological Archives* http://dx.doi.org/10.1890/ES14-00204.1.sm).