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Research article

Effects of land use change, wetland fragmentation, and best management practices on total suspended solids concentrations in an urbanizing Oregon watershed, USA

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ABSTRACT

While many different watershed management strategies have been implemented to improve water quality, relatively few studies empirically tested the combined effects of different strategies on water quality in relation to land cover changes using long-term empirical data at the sub-basin scale. Using 10 years of total suspended solids (TSS) data, we examined how the conversion of wetland, wetland fragmentation, beaver dams, and Best Management Practices (BMPs) affect wet season TSS concentrations for the 25 monitoring stations in the Tualatin River basin, USA. Geographic information systems, FRAGSTATS, and correlation analysis were used to identify the direction of land cover change, degree of wetland fragmentation, and the strength of the relationship between TSS change and explanatory variables. Improvement in TSS concentrations was tightly coupled with the aggregation of wetlands, presence of beaver dams, particularly during the mid-wet season when flows were highest. Other BMPs effectively reduced TSS concentrations for the early and late-wet seasons when flows were not as high as in the middle wet-season. Aggregated wetlands were more effective for improving water quality than smaller disaggregated wetlands of similar total area when combined with the presence of beaver dams and BMPs. These findings offer important scientific and practical implications for management of urbanizing watersheds that seek to achieve the dual goals of improving environmental quality and land development.

1. Introduction

Wetlands offer a suite of ecosystem services, benefits humans can obtain from nature. Wetlands, by retaining sediments and nutrients, could help improve downstream water quality (Widney et al., 2018) while protecting drinking water supply sources (Wang et al., 2016). Additionally, wetlands retain stormwater, releasing water gradually, and thus mitigating downstream flooding (Antolini et al., 2020). By creating a space and food web for aquatic and amphibian species and bird species that rely on wetland ecosystems, wetlands provide habitat for such species (Yamanaka et al., 2020). Because of these multiple benefits, the importance of wetlands in providing these ecosystem services has been studied globally (Albert et al., 2020). One study estimated the value of wetlands to be approximately $47.4 trillion per year, which is slightly less than half (43.5%) of the value of all-natural biomass (Davidson et al., 2019).

However, wetland ecosystem services have been dramatically reduced as humans have encroached on wetlands and converted them into agricultural or urban areas. At a global scale, less than 20% of preindustrial freshwater wetlands remain and, assuming the current rate of land development, half of the existing wetlands are projected to decline by the mid-21st century (Albert et al., 2020). At a regional or a local scale, wetland loss is a primary environmental concern for growing urban areas. Wetland losses are typically associated with degradation of water quality (Huang et al., 2019), lowered flow during dry periods (Blanchette et al., 2019), increases in flooding during wet seasons (Siverd et al., 2019), and the loss of habitat (Shepack et al., 2017). Continuous urban development in the Guanting reservoir basin in China is projected to decrease the water purification service of wetlands by nearly 10% due to wetland loss and fragmentation (Huang et al., 2019). A 15% decrease in wetland area is associated with up to a 20% reduction in low flows and a 20% increase in high flows in the St. Charles River, Quebec, Canada (Blanchette et al., 2019), which have important negative implications for water quality.

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The effects of wetlands on water quality have received much attention in the literature with some nuanced findings. Wetlands can intercept pollutants from nonpoint sources and reduce total suspended solids (TSS), nutrients, and metals reaching streams and rivers (Johnston 1991; Jia et al., 2016; Wan et al., 2014). While the construction or restoration of wetlands is expected to improve downstream water quality, the effects vary subject to local conditions. Created wetlands in the middle Rio Grande somewhat reduced nitrate concentrations in the river, likely due to denitrification enhanced by algae (Rodriguez and Lougheed 2010). In contrast, newly constructed wetlands released a higher amount of phosphorus than in Upper Klamath Lake in Oregon after flooding, suggesting that the initial phase of wetland restoration could increase nutrient concentrations (Wong et al., 2011). A study of four urban wetland restoration areas in Qinghai Plateau in China showed the differential performances of these wetlands in terms of ecosystem function and ecosystem services (Mao et al., 2019). An accumulation of small wetlands near streams retained phosphorus and improved water quality in the Lake Champlain Basin, USA (Singh et al., 2019).

To mitigate the negative consequences of urban development and wetland losses, some municipalities have introduced new laws and regulations that are designed to protect existing wetlands, construct artificial wetlands, restore old wetlands or install green stormwater infrastructure (Baker et al., 2019) as part of best management practices (BMPs). If wetland destruction is inevitable as part of development, developers need to find space for creating new wetlands, so-called wetland banking (Robertson 2006; Hough and Robertson 2009). Moreover, some pioneering communities purchased reclaimed or destroyed wetlands and converted them back to natural wetlands for restoring ecosystem services (Jackson Bottom Wetlands Preserve 2020). However, it is currently unknown how such wetland restoration and other BMPs, which affect the size and number of wetlands, positively affects downstream water quality at the watershed scale (Walsh et al., 2015). Two decades of wetland restoration, while slightly improving nutrient cycling, was unsuccessful in improving sedimentary balance (reduced slit loads) in the Danube river delta, resulting in increased coastal erosion (Gómez-Baggethun et al., 2019).

The reintroduction of beavers in urban streams has been used to restore wetlands and natural stream functions that provide ecosystem services, including the improvement of water quality (Maret et al., 1987; Westbrook et al. 2006, 2011; Puttock et al., 2017). Some studies showed that beaver dams have been effective in retaining sediment and nutrients, thus improving downstream water quality (Smith et al., 2020a), while others did not find significant improvement in water quality after beaver dams had been constructed (Smith et al., 2020b). Since many landscape factors can potentially affect water quality (Linton et al., 2018; Shi et al., 2019), effective watershed management relies on examining these human and natural systems holistically and the combined effects and importance of wetland fragmentation and aggregation, land use and urban development, BMP implementation, and reintroduction of beaver and their impact on water quality. This study seeks to fill gaps using long-term data sets to better understand the importance of these factors for watershed management to meet water quality requirements and improved environmental services (Fig. 1).

We seek to answer the following research questions.

(1) What is the relationship between wetland change (gain or loss and fragmentation) and TSS change? We hypothesize wetland loss and fragmentation of wetlands have negative impacts on TSS (i.e., increase in TSS concentration);
(2) What is the effect of sub-seasons within the wet season with varying flow variability on TSS change at the subwatershed scale? We hypothesize changes in TSS differ by sub-season. In other words, we expect higher changes in TSS during the peak flow season, but the degree of change could vary by subwatershed;
(3) How are beaver dams and other BMPs associated with wet sub-season changes in TSS? We hypothesize the effects of beaver dams are more pronounced during the peak flow season than the beginning or late wet seasons.

2. Study area

The 1844 Km^2 Tualatin River basin, located in the northwestern part of Oregon is a tributary to the Willamette River, is our study area (Fig. 2). The study basin was chosen because it is a representative Level

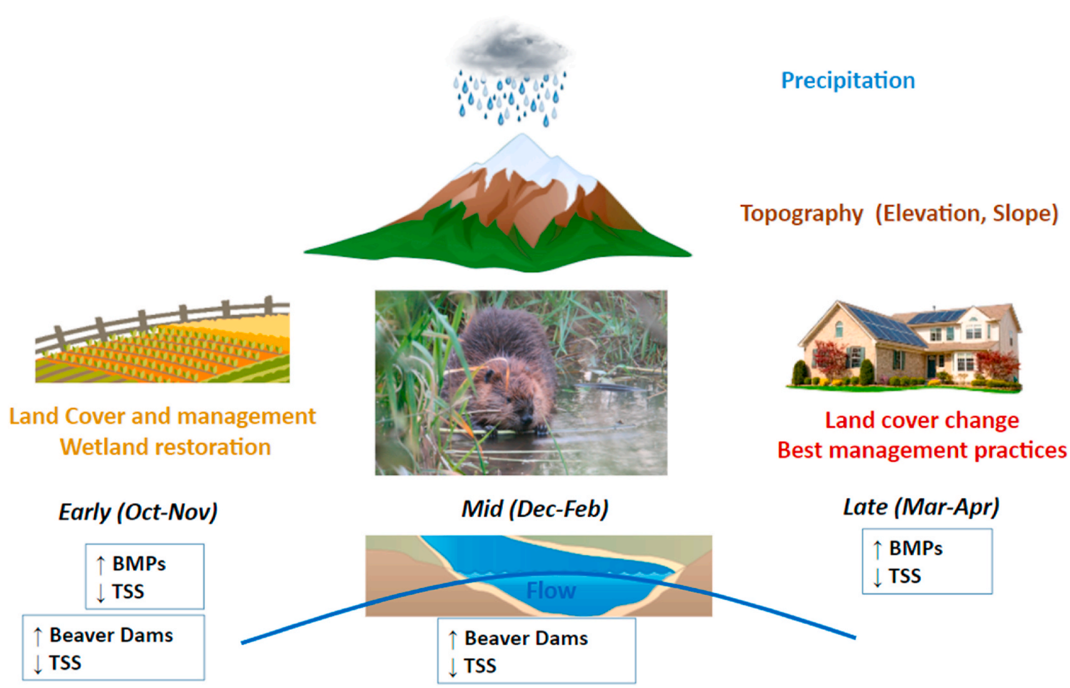

Fig. 1. Conceptual diagram for understanding change in TSS in relation to flow, landscape change and management practices.

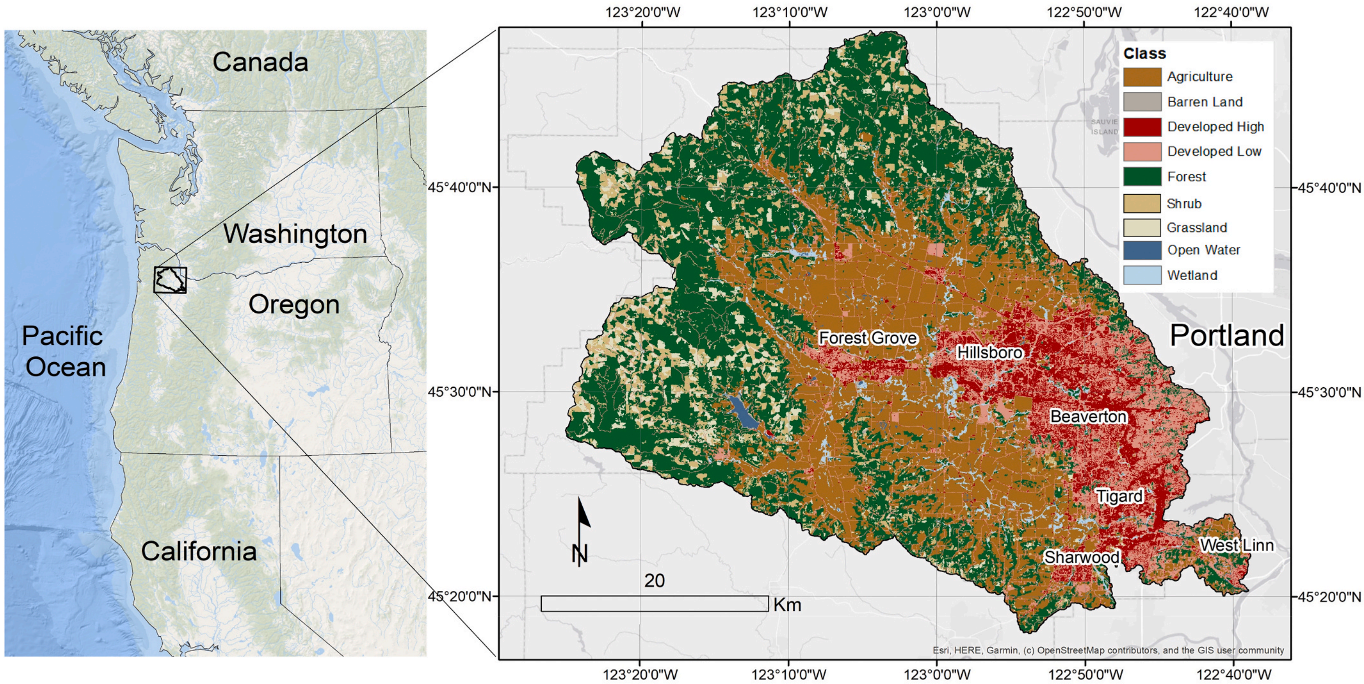

Fig. 2. Study area: Tualatin River basin, Oregon, USA.

III ecoregions (Wiken et al., 2011), is wetland-rich relative to the Willamette basin, is expected to have overall high ecosystem function and services; is projected to have a mix of human population densities (urban vs. rural) and human modifications (natural vs. human-made environments); and has many management actions to improve water quality. Located between the Coast Range and Cascade mountains, the basin has a modified marine west coast climate. The mean annual precipitation, measured at Hillsboro weather station, is approximately 970 mm (1981–2010), with 83% of the precipitation falling in the wet season from October to April (US Climate Data 2020). The Tualatin river begins in the Coast Range mountains in the west, flows east through low lying areas in the middle valley, and empties into the Willamette River near West Linn. Except for its headwaters where there exist a steep gradient including some waterfalls, the river, named after the Native American word for "lazy", exhibits a very low-gradient. In particular, the elevation change between river kilometers 53.6 and 5.5 has an estimated slope of 1.5 cm/km (Tualatin River Watershed Council 2020). The low gradient of the river contributed to the historic existence of numerous wetlands throughout the basin.

Located near and within the greater Portland metropolitan area, the basin has undergone significant land cover changes in the last century. Since European settlement in the mid-19th century, the basin's once-abundant wetlands declined substantially as natural wetlands were converted to either agricultural, commercial or industrial areas. While Oregon's progressive land-use laws (e.g., exclusive farming zone, natural protected area) have preserved some natural areas from new development, a gradual expansion and development infilling natural areas within the urbanized area, have made some wetlands more fragile. Population growth and the disappearance of natural areas, including wetlands, led to severely degraded water quality in the mid-to late- 20th century. Water quality improved for some parameters through Clean Water Act regulations to address point sources, but other water quality parameters affected by runoff from nonpoint sources have not improved (ODEQ 2012). However, there have been watershed management actions implemented through regulatory and voluntary mechanisms that have resulted in stream, riparian, and wetland restoration coordinated between Clean Water Services, Tualatin Soil Water Conservation District, private landowners, and multiple state and federal agencies that could improve water quality affected by nonpoint sources (CWS 2017a; SWCD, 2020). Such efforts include the construction of storm detention and retention ponds, particularly in newly developed neighborhoods, and reintroducing beavers to urban streams, which can lead to reduced sediment loads.

3. Data and methods

3.1. Data

3.1.1. Land cover and topographic data

Land cover data for 2001 and 2016 were obtained from the Multi-Resolution Land Characteristics Consortium National Land Cover Database (NLCD) (U.S. Geological Survey USGS, 2020). The NLCD provides nationwide land cover data at a 30 m resolution with a 20-class legend based on a modified Anderson Level II classification system (Anderson 1976; Jin et al., 2016). The original 20 categories were reclassified into nine classes that have similar levels of erosion potential. Topographic data such as mean elevation and slope were derived from 5 m Lidar derived digital elevation model. Within the study area, the south and eastern parts are classified as developed areas, surrounded by agricultural fields. Forest and shrubland mainly cover the north and west sides of the study area (Fig. 2).

3.1.2. Water quality and other management data

We selected TSS because it has been used in water quality regulation and management in the U.S. (USEPA 2010), serving as an indicator for other water quality parameters, such as DDT (ODEQ 2008), mercury (ODEQ 2019) and metals (Nasrabadi et al., 2016). TSS data were obtained from 25 monitoring stations (8 mainstem stations and 17 tributary stations) maintained by Clean Water Services (See Fig. 3, Table S1). Grab samples were taken at least monthly for all these sites, following the methods described in CWS (2015). TSS samples were analyzed using methods based on Standard Methods 2540 D and 2540 E with a method reporting limit of 0.2 mg/L (Rice et al., 2018) and standard quality assurance and quality control for sample collection and analysis (CWS

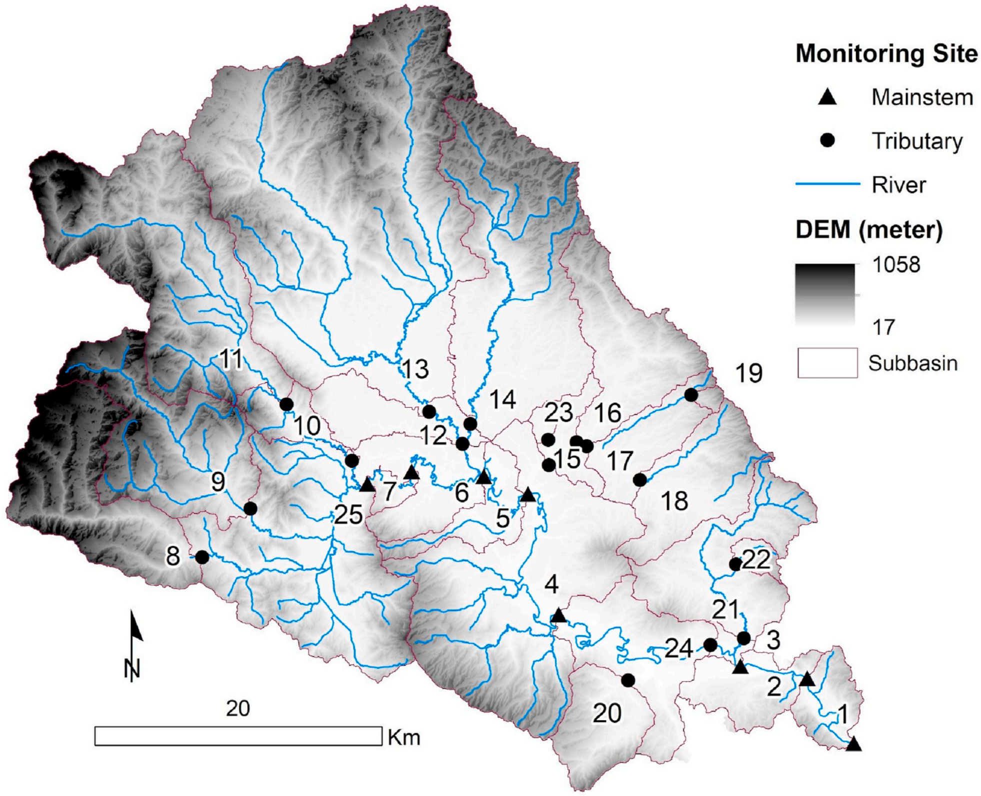

Fig. 3. Locations of 25 monitoring sites, subbasin boundaries, and elevation ranges derived from a 5-meter digital elevation model (DEM). Numbers correspond to the station numbers in Supplementary Table 1.

2017b). These sites were chosen because they have long-term TSS data since the late 1990s with fewer missing values. We focused our analysis on the wet season because TSS is highly flow-dependent and hypothesized that wetlands and other BMPs are likely to retain TSS during the wet period. The TSS data were summarized for two time periods with the first time period bracketing 2001 (Oct 1998–Apr 2003) and the second time period bracketing 2016 (Oct 2013–Apr 2018) to match the corresponding land cover data from 2001 and 2016. These two time periods have similar flows with wet, dry, and normal years in each period, thus representing hydroclimate variability well. The wet season was further divided into three periods - beginning, mid and end of the wet seasons - because flow regimes and the delivery mechanisms of TSS are likely to vary as sources might be depleted over time (Chen and Chang 2019). Other BMP data, associated with improving water quality, such as stormwater structures, stream enhancement projects, and beaver activity, were obtained from the Clean Water Services. Most of the stormwater infrastructure (both grey and green) are located within the urban growth boundary, while beaver dams and stream enhancement projects are scattered throughout the basin.

3.2. Methods

3.2.1. GIS change detection

ArcGIS version 10.6 was used to conduct various geospatial analysis (ESRI 2020). First, by using ArcGIS's Hydrology toolset, we delineated the contributing areas upstream of the monitoring sites using a 5 m resolution digital elevation model (Fig. 3). Second, we reclassified the 2001 and 2016 NLCD data into binary classes (wetland and others) to identify the total wetland area within each subbasin in each year. We then calculated the difference in wetland areas (both absolute and relative terms) between the two years. Third, we identified the total area of wetland conversion using a metric to detect whether wetlands were converted to non-wetlands or vice versa. Fourth, once each subbasin boundary was delineated, we calculated the percentage of each land cover, slope, elevation, and other stormwater management variables (e.g., beaver dam density, stream enhancement project density, storm infrastructure density) in each subbasin.

3.2.2. Landscape fragmentation

We conducted a landscape fragmentation analysis using spatial landscape metrics (Gustafson 1998), which quantifies the composition and configuration of the study landscapes (Turner 2001; O'Neill et al., 1999). Using the FRAGSTATS software program (Version 4.2) (McGarigal et al., 2012), we computed six metrics for wetland class: Number of Patches (NP), Patch Density (PD), Landscape Shape Index (LSI), Mean Patch Area (AREA_MN), Aggregation Index (AI), and Patch Cohesion Index (Cohesion) within each subbasin (Table 1). These five class metrics were chosen based on previous studies since they represent size (AREA_MN), shape (LSI), number (NP), density (PD), aggregation (AI), and connectivity (Cohesion) of wetlands and thus are hypothesized to be associated with water quality (Couvillion 2016; Junhong 2008; Xiao 2016; Xu 2010).

3.2.3. Statistical analysis

We first summarized winter TSS concentrations in early (October–November), middle (December–February), late (March–April) wet seasons, following a previous study in the region that indicated different turbidity dynamics at different times of the wet season (Chen and Chang 2019). We then calculated change in median concentrations in TSS for each subseason for the 19 stations (12 tributary stations and 7 mainstem stations) that have long-term TSS data. Given the relatively small sample size and non-normal distribution of the TSS and other data, we used a non-parametric Spearman rank correlation coefficient to identify the relationship between TSS concentration change and landscape and management variables at each subbasin derived in section 3.2.1 and 3.2.2.

Table 1
Landscape metrics used in the current study.

Landscape metrics	Explanation	Range
Number of Patches (NP)	NP equals the number of patches of the corresponding patch type	NP $\geq$ 1, without limit
Patch Density (PD)	PD equals the number of patches of the corresponding patch type divided by total landscape area (m^2), multiplied by 10,000 and 100 (to convert to 100 ha)	PD > 0, constrained by cell size
Landscape Shape Index (LSI)	LSI equals the total length of edge (or perimeter) involving the corresponding class, given in number of cell surfaces, divided by the minimum length of class edge	LSI $\geq$ 1, without limit
Mean Patch Area (AREA_MN)	AREA_MN equals the sum, across all patches of the corresponding patch type, divided by the number of patches of the same type	AREA_MN > 0
Aggregation Index (AI)	AI equals the number of like adjacencies involving the corresponding class, divided by the maximum possible number of like adjacencies involving the corresponding class	0 $\leq$ AI $\leq$ 100
Patch Cohesion Index (COHESION)	COHESION equals 1 minus the sum of patch perimeter divided by the sum of patch perimeter times the square root of patch area for patches of the corresponding patch type, divided by 1 minus 1 over the square root of the total number of cells in the landscape, multiplied by 100 to convert to a percentage.	0 $\leq$ COHESION < 100

4. Results

4.1. Wetland changes (loss and gains)

The total wetland area slightly decreased by approximately 0.056 km^2 between the two time periods. However, the area of increase/decrease varied by subbasin. Within 25 subbasins (there is no wetland area in subbasin 19), wetland area increased in 8 subbasins, while it decreased in 14 subbasins (Fig. 4). There was no wetland area change in two subbasins. Subbasin 12, near the rapidly growing cities of Hillsboro and Forest Grove, had the greatest decrease in wetland area. Subbasin 6, located in the middle section of the mainstem Tualatin River, had the greatest increase in wetland area. As shown in Fig. 4, most of the decrease in wetland areas occurred within the urban growth boundaries in the study basin.

4.2. Wetland metrics

Fig. 5 displays how wetland patches have changed spatially between 2001 and 2016. The number of patches generally increased in the middle section of the basin (subbasin # 4, 6, 7, 12, 13, 15), while it decreased in the most developed eastern part of the basin (subbasin # 2, 3, 18, 20, 21, 22, 24). Mean patch area exhibits a somewhat opposite spatial pattern. Mean patch area increased in subbasins within the urban growth boundary (2, 3, 15, 18, 21, 22, 24), while it decreased in the middle section of the basin. The spatial patterns of changes in landscape shape index and aggregation index are somewhat similar, exhibiting an east-west gradient with increases toward the eastern part of the basin. More specifically, in subbasin 18 and 24, the number of patches decreased, the mean patch areas increased, and the shape became more aggregated (compact). While in subbasins 6 and 9, where the number of wetland patches increased, the size of the patches decreased, and the

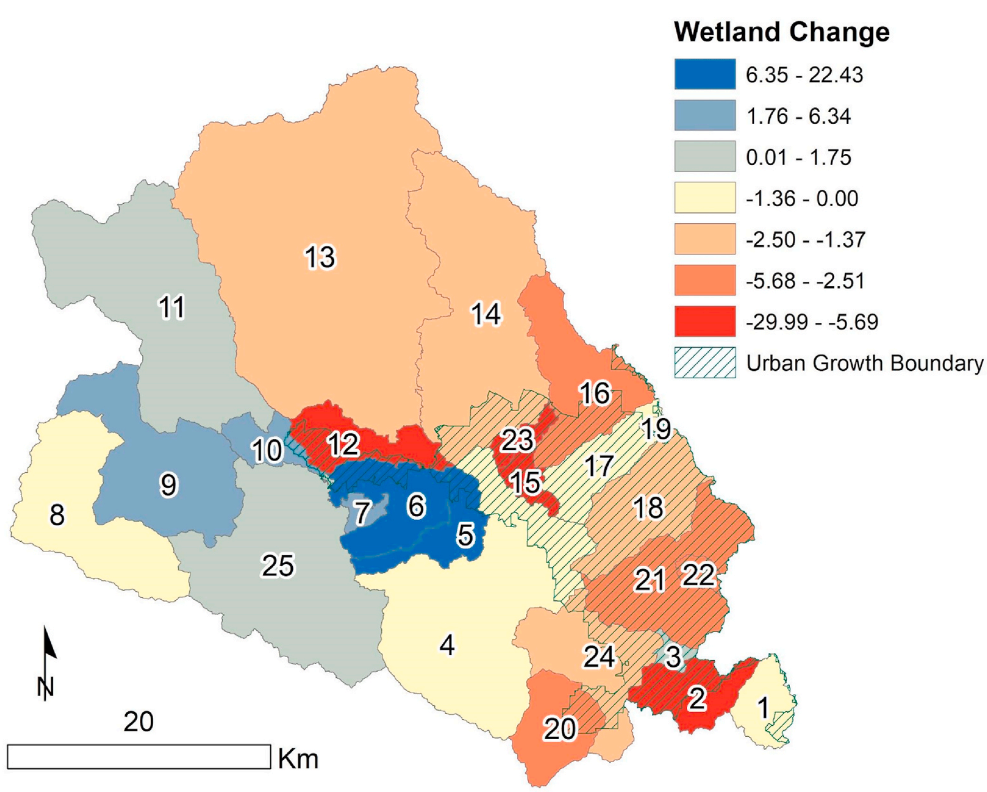

Fig. 4. Change in wetland area between 2001 and 2016 divided by subbasin area (multiply by 10^4 (% X 100)).

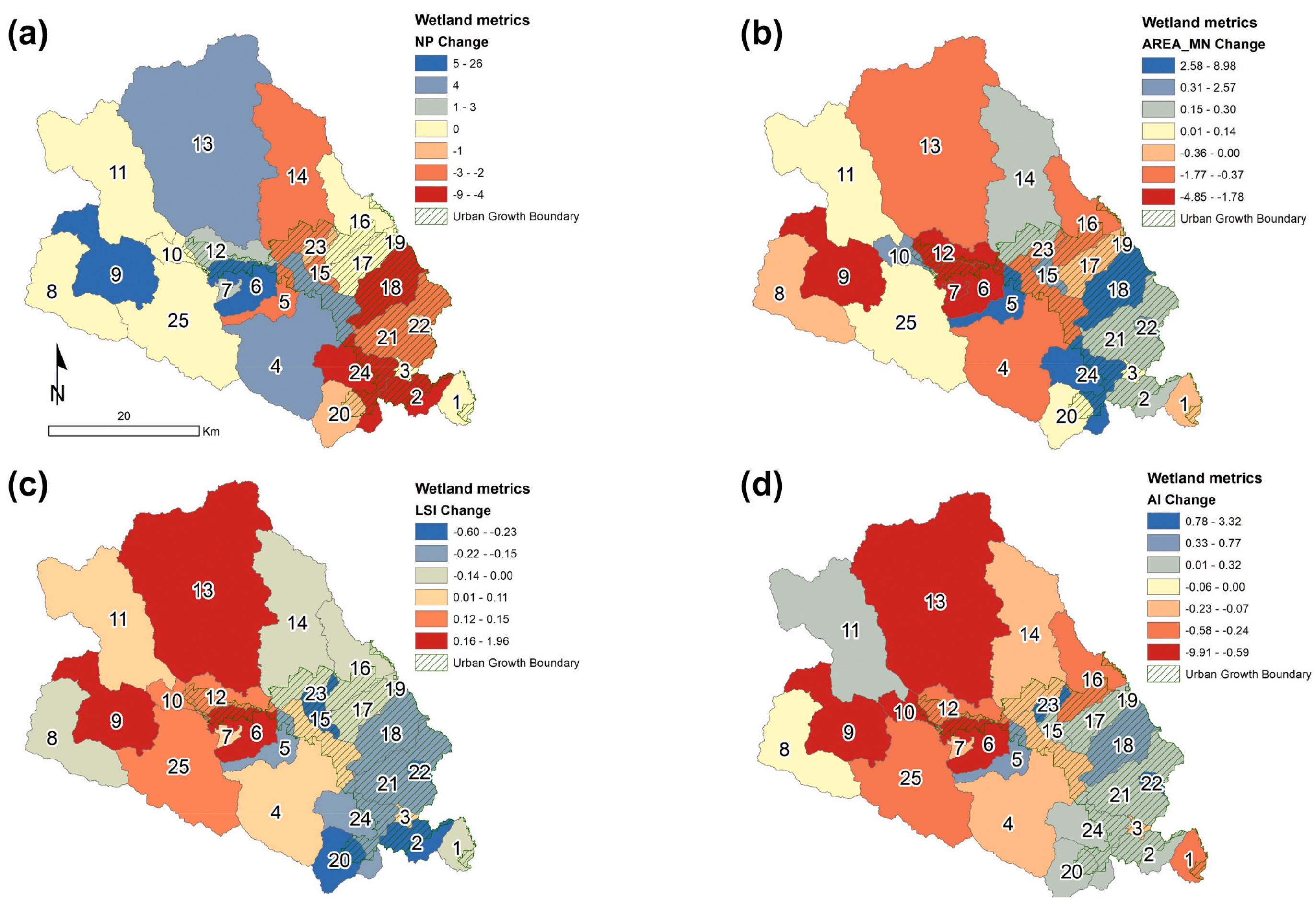

Fig. 5. Change in (a) number of patches (NP), (b) mean patch area (AREA_MN), (c) landscape shape index (LSI), and (d) aggregation index (AI) between 2001 and 2016. (a): Positive number indicates that the number of wetland patches increased, and negative number indicates that the number of wetland patches decreased; (b): Positive number indicates that Mean Patch Area increased, and negative number indicates that Mean Patch Area decreased, (c): Positive value indicates that wetlands are more disaggregated (fragmented), and negative value indicates that wetlands are more aggregated (compact), (d): Positive value indicates that wetlands are more aggregated (compact), and negative value indicates that wetlands are more disaggregated (fragmented)).

shape became more fragmented (less compact).

4.3. Direction of land cover change (wetland to non-wetland or non-wetland to wetland)

For the areas that were wetlands in 2001 but were converted to non-wetland areas in 2016, there is an interesting spatial pattern in relation to urban growth boundary (UGB) (Fig. 6a). Out of 25 subbasins, 13 subbasins were mainly converted from the wetland class to the low-density development class and they are all located within the UGB. Five subbasins, mostly located in the periphery of the UGB, were mainly converted from wetland class to water class. For the conversion of other land covers in 2001 to wetlands in 2016, the dominant land conversion is from agricultural land to wetlands (11 subbasins), likely due to stream enhancement projects implemented in those subbasins. These subbasins are all located outside of the UGB. Six other subbasins, mostly within the UBG, were mainly converted from water to wetland, likely due to partial infilling from new development or new beaver dams (Fig. 6b).

4.4. TSS concentration in WY 13–18

Fig. 7 shows the spatial variation of TSS concentrations for each of the monitoring stations in the Tualatin River basin for the water years between 2013 and 2018 (representing 2016). As shown in this figure, subbasins in the middle of the basin exhibit higher TSS concentrations than other sub-watersheds in western (forested subbasins) or eastern side (within UGB) of the basin. These subbasins with higher TSS concentrations are predominantly used for agricultural activities such as subbasin 13, Dairy Creek subbasin. This subbasin shows consistently high TSS concentrations in all wet seasons. It is also notable that late wet season TSS concentrations are highest in the middle section of the basin. Additionally, some subbasins (subbasin 2, 3, 16, 22, 24) within the UGB exhibit relatively higher TSS concentrations in the early and middle wet seasons than in the late wet season.

4.5. Change in TSS between WY 98-03 and WY 13–18 median TSS concentrations

Between WY 98-03 and WY 13–18, TSS concentrations decreased in most mainstem and tributary stations in the Tualatin River basin (Fig. 8). More substantial declines in TSS were found in the tributary stations (nearly 25 mg/L reduction in station 21) than the mainstem stations (less than 5 mg/L reduction). In general, TSS concentrations decreased more at downstream stations than upstream stations (Fig. 8b). Even with a similar degree of urban development, subbasin 23 showed a more substantial improvement (higher reduction in TSS concentration) than subbasin 22, which showed an increase in TSS for the early and middle wet season. Overall, the highest decreases were found in the middle wet season when flows were the highest. Increases in TSS concentrations were reported in two mainstem (# 5, 7) and four tributary stations (# 8, 10, 12, 22) in the early wet season.

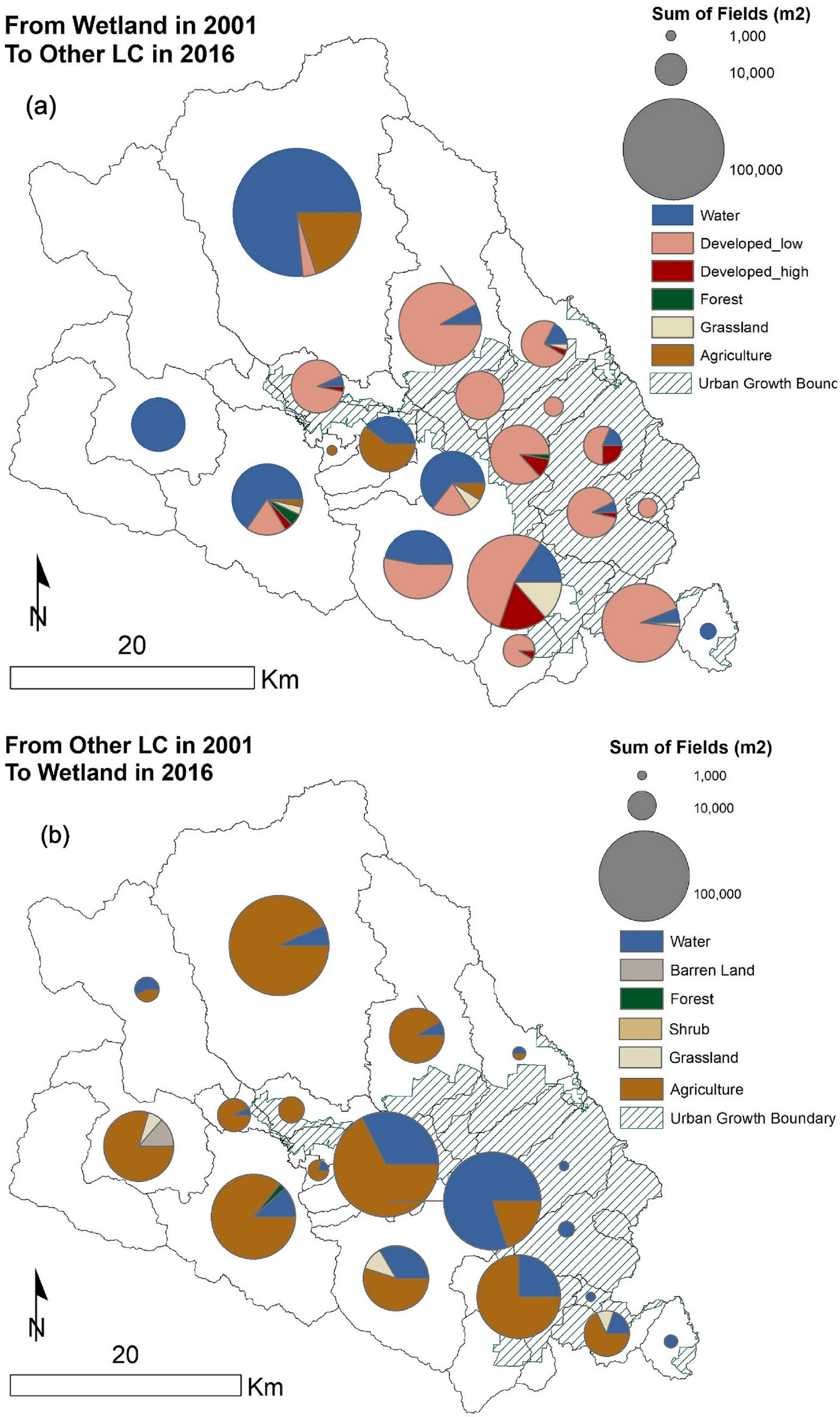

Fig. 6. Conversion of (a) wetland to other land covers and (b) other land covers to wetland between 2001 and 2016 (Unit m^2).

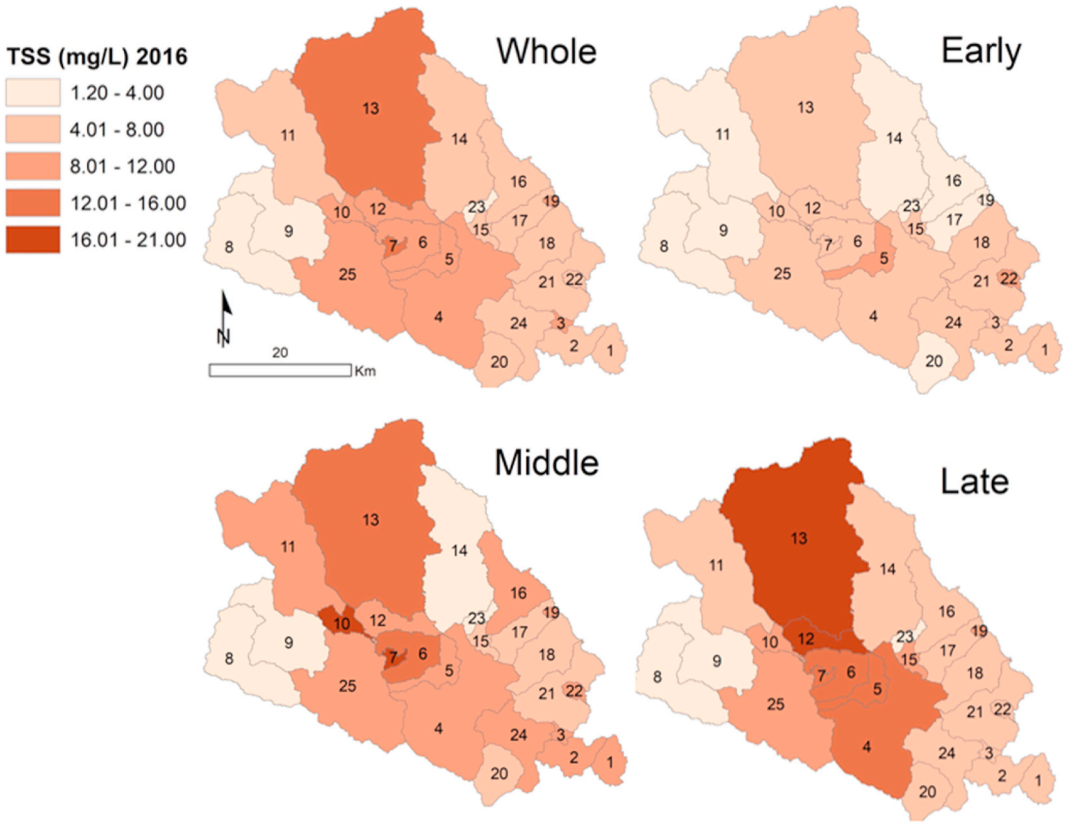

Fig. 7. TSS (mg/L) concentration in each subbasin of the Tualatin river in WY13-18.

4.6. TSS change and wetland and landscape management variables

Table 2 shows the relationship between TSS change and static landscape variables. As shown in this table, topographic variables are positively related to TSS changes (degradation) in the whole season and the late wet season. The 2016 land cover variables are either negatively (low development and high development, patch size) or positively (forest and scrubland) associated with TSS change. Forest land cover (r = 0.643) is more strongly positively correlated to TSS change than scrubland (r = 0.58) for the whole season, but not for the late season. Stormwater management variables are always negative related to TSS change (improvement), suggesting that either beaver dams or stream enhancement projects or storm infrastructure improve TSS. Beaver dam is negatively related to changes in TSS only in the early and mid wet seasons. In other words, the more beaver activities in a subbasin, the higher the decrease in TSS (improvement in TSS), particularly significant during the mid wet season ($p < 0.01$) and marginally significant in the early wet season ($p < 0.1$). Storm infrastructure appears to be more effective during the beginning and the end of the wet season, but not in the mid-wet season when flow is highest. Stream enhancement projects are always negatively related to TSS change in all seasons.

4.7. TSS change and changes in landscape variables

Table 3 shows the correlation between TSS change and changes in land cover and wetland fragmentation indices. Conversion of wetlands to low-density development and water to wetlands is negatively related to TSS change in different wet seasons. Additionally, increases in forest areas and high density development are negatively related to TSS (improvement), while increases in shrublands are positively correlated to TSS (degradation). Forest land cover change (r = - 0.755) is more strongly correlated to TSS change than scrubland change (r = 0.657) for the whole season. These changes in land cover between the two years are most significant in the late wet season. Given that slope and elevation are highly positively related to forest land cover, reduction in forest land cover in steep slope areas are likely to lead to increases in TSS concentrations, as demonstrated by the negative correlation between forest land cover change and TSS changes. Changes in wetland fragmentation indices are also only statistically significant in the late season. While increases in the number of wetland patches (NP), patch density (PD), and shape index (LSI) are positively related to TSS increase, wetland aggregation indices (AI) are all negatively related to TSS change. Wetland connectivity metric (Cohesion) is not statistically significantly related to TSS change (not shown).

5. Discussion

5.1. Effects of restoration and long-term monitoring

Our study, which investigated TSS changes in relation to modifications of the wetlands and other management practices in the past two decades, indicates the importance of long-term monitoring and spatial scale when assessing the effectiveness of restoration and other best management practices in urbanizing watersheds. While a previous study (Singh and Chang 2014) in the same basin did not find much improvement in summer water quality in most stations, the current study shows more promising improvement in wet season water quality. This different finding is most likely related to the fact that it would take some time to detect the effects of restoration efforts and other management actions on stream water quality. Singh and Chang (2014) used data from 1991 to 2010, while this study used the data from 1998 to 2018. The two decades of restoration and best management efforts appear to be functioning in the study basin, particularly in newly developed sub-watersheds. This finding is in good agreement with a previous study that found different TSS mean concentrations in two similar urban catchments with different land management practices (Shi et al., 2019).

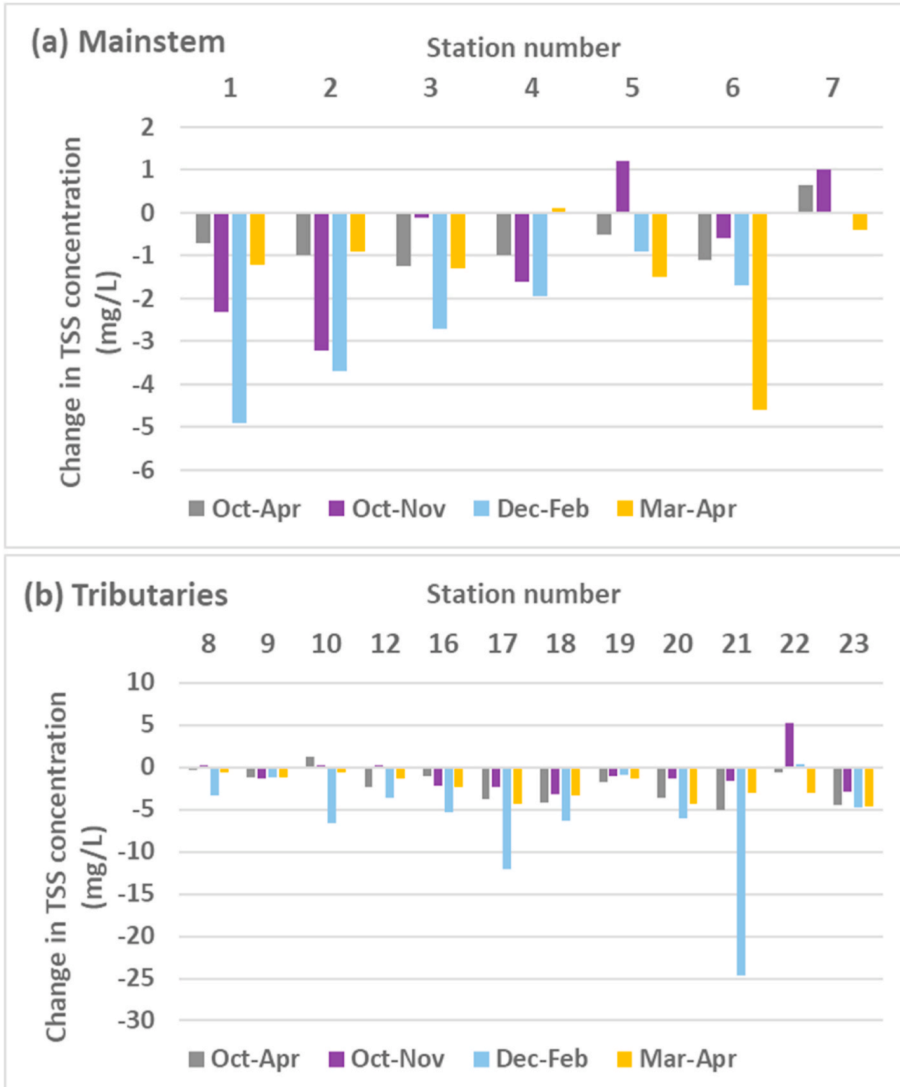

Fig. 8. Difference between WY98-03 and WY13-18 median TSS (mg/L) for the (a) mainstem Tualatin River stations and (b) tributary stations. See Table S1 for station number description.

Similarly, studies reported positive effects of riparian planting on water quality in the basin. Trees need to grow to retain sediments and provide shades to offer beneficial effects (Cochran and Logue 2011). Since nearly 70% of sediments are derived from banks in Fanno Creek (Keith et al., 2014), a tributary of the Tualatin River, restoration of riparian areas are highly likely to reduce sediment transport to the mainstem.

5.2. Potential effects of flow and sediment mass load

Changes in TSS concentrations are also likely to result from changes in streamflow and sediment mass load. While no comprehensive streamflow monitoring data are available, using a tributary site, Fanno Creek at Durham Road's (subbasin 21) daily flow data shows that wet season streamflow between the two periods (Oct 2001–April 2003 vs. Oct 2013–April 2018) are similar to each other. The relationship between streamflow and TSS, as indicated by the position of the regression line, decreased between the two periods, suggesting that less sediment is likely to be transported in a given amount of flow (see Supplementary material). We suspect that decreases in sediment loads are partially explained by a combination of stream restoration along the creek and

Table 2
Spearman's rank correlation coefficient between change in seasonal TSS concentration and landscape and management variables (n = 12).

	Topography		Land cover in 2016					Storm management		
	Slope	Elev	Low_Dv	High_Dv	Forest	Shrub	AREA_MN	Beaver dam	Enhance	Storm_Inf
Whole	0.722***	0.664**	−0.545*	−0.734***	0.643**	0.580*			−0.573*	−0.739**
Early				−0.669**			−0.642**	−0.611***	−0.609**	−0.684**
Middle							−0.555*	−0.847***	−0.79***	
Late	0.754***	0.684**	−0.611**	−0.867***	0.723***	0.730**			−0.572*	−0.823***

* = statistically significant at 0.1 level; ** = statistically significant at the 0.05 level; *** = statistically significant at the 0.01 level; Elev = elevation; Low_Dv = low density development; High_Dv = High density development; AREA_MN = mean patch area; Storm_Inf = storm infrastructure.

Table 3
Spearman Rank Correlation Coefficient between change in seasonal TSS concentration and change in landscape variables (n = 12).

	Conversion		Change in Land cover, 2001–2016			Change in wetland fragmentation			
	Wet2LDv	Wat2wet	High_Dv	Forest	Shrub	NP	PD	LSI	AI
Whole	−0.635**		−0.613**	−0.755***	0.657**	0.526*		0.539*	−0.539*
Early			−0.540*						
Middle		−0.647**							
Late	−0.608**		−0.626**	−0.751***	0.608**	0.616**	0.611**	0.830***	−0.78***

* = statistically significant at 0.1 level; ** = statistically significant at the 0.05 level; *** = statistically significant at the 0.01 level; Wt2LDv = wetland to low developed land; Wat2wet = water to wetland; High_Dv = High density development; NP = number of patch; PD = patch density; LSI = landscape shape index; AI = Aggregation index.

the introduction of beavers to the stream. The park's trailway can be flooded during high flow events, spilling over sediment to surrounding areas instead of transporting to downstream areas where the monitoring station is located. Similar sediment reduction after floodplain restoration has been reported for Johnson Creek (Ahilan et al., 2018). The reduction in TSS in a later period could be due to changes in wet season peak flow, which may also be affected by beaver dams and other BMPs. By retaining stormwater longer, these nature-based storm infrastructures release water gradually, extending the timing of peak flow and reducing peak flow, resulting in TSS reduction downstream. Previous studies in the Portland metropolitan region identified that green infrastructure (e.g., bioswales, green roofs) effectively reduced total storm runoff amount (Chang et al., 2020; Fahy and Chang, 2019).

5.3. Implications for wetland restoration

The findings of the current study also offer some insights into the direction of wetland restoration. Wetland restoration appeared to be most effective when wetland sizes got bigger rather than an increasing number of small fragmented wetlands, as indicated by increases in wetland aggregation index with decreased TSS concentrations. The importance of wetland aggregation is supported in that two urban subbasins of similar size (subbasins 22, 23) showed different degrees of improvement in TSS even with a similar degree of urban development. Additionally, by reducing shrublands (potentially converting to wetlands), it might be possible to improve water quality in the study area. The study basin had many wetlands before European settlements, therefore restoring converted agricultural lands to original wetlands can be considered a priority restoration strategy. Our study suggests that wetland restoration efforts should be consistently implemented for a long period of time. It is crucial to create wetlands that are extensive enough to offer the desired functionalities, as reduced wetland cover was detrimental to wetland quality (Patenaude et al., 2015). Large wetlands effectively serve as sinks of sediments or other nutrients and improving downstream water quality. Small fragmented wetlands may not function as desired, particularly during high flow events, when water can overflow to surrounding areas and mobilize sediment or nutrients (Jia et al., 2019).

5.4. Introducing nature-based solutions

Our study also indicates that implementing nature-based green stormwater infrastructure and introducing beavers into urban watersheds positively influence water quality. The different significant correlations between TSS and nature-based solutions by different seasons suggest that a combination of stormwater facilities and beaver activities can effectively reduce sediment concentrations. While constructed stormwater detention and retention facilities are somewhat effective in removing sediments at the beginning and the end of wet seasons when flows are relatively low, beaver dams appear to be most effective during the high flow seasons. This finding is confirmed by a recent US Geological Survey study where upstream beaver dams helped reduce TSS concentrations in downstream reaches in Fanno Creek (Smith et al., 2020a). By attenuating peak discharge during storm events (Neumayer et al., 2020) and increasing connectivity to adjacent floodplains and wetlands (Westbrook et al., 2006; Macfalane et al., 2017), beaver dams appear to effectively reduce sediment transport downstream during the highest flow season. However, the full benefits of beaver dams in retaining and removing nutrients and sediments in urban watersheds could be further investigated (Vidon et al., 2019).

5.5. Limitations of the current study and future research directions

The current research could be improved in several ways. First, while we used widely available national land cover data for characterizing land cover change and fragmentation indices, we acknowledge that the results could be different or improved had we used higher resolution land cover data. Second, the effectiveness of BMPs can be more thoroughly examined by collecting finer spatial and temporal resolution TSS data. As suggested by previous studies, TSS concentrations could change abruptly during storm events demonstrating hysteresis effects (Chen and Chang 2019). A storm event scale analysis can further identify the potential sources of sediments in the study area. Third, the interaction between beaver dams and wetlands warrant more attention. Since beavers move around dams over time, the effects of changing beaver dam locations and dam heights and depths in relation to changes in the surrounding riparian conditions and their effects on retaining sediment could be examined further. Fourth, a processed-based model can be employed to identify potential costs and benefits of different choices among grey to green infrastructure to beavers. An agent-based model could help inform decision-makers about various outcomes of different choices. Finally, as successful restoration and implementing new regulations and policies relies on broad participation of stakeholders, informing them the results and gathering feedback would be essential.

6. Conclusions

This study examined how changes in wetland area and fragmentation, beaver dams, and other storm green infrastructure are related to TSS changes in an urbanizing watershed at the subbasin scale during the wet season. TSS concentrations decreased in most of the subbasins for all seasons between 2001 and 2016, suggesting that decades of restoration might eventually be working. While wetland areas were converted to low-density urban development in the periphery of the urban growth boundary, TSS concentrations decreased. The improvement in TSS is likely attributed to increases in mean patch area and wetland aggregation index. Our sub-season analysis shows that introducing beaver dams to urban streams effectively reduces TSS concentrations in the middle of the high flow season, while other stormwater infrastructure retain sediment in the early and late wet seasons.

The current study's findings have fresh scientific insights for understanding the relationship between landscape change and water quality. Additionally, it has broad practical implications for sustainable land and water quality management in a growing urban region. We identified the importance of considering the direction of land conversion, landscape fragmentation, nature-based solutions, and seasonality to assess water quality changes. With the right combination of best management practices, growing urban regions can minimize the negative consequences of urban development on water quality, and it is even possible to improve water quality. There is still room for future studies, including finer spatial and temporal scale analysis, how different choices of BMPs yield different water quality outcomes, and quantifying costs and benefits of different strategies.

Credit author statement

All authors contributed to the study conception and design, data collection and some form of analysis. HC wrote the first draft of the manuscript, and YM and EF commented on previous versions of the manuscript. All authors read and approved the final manuscript.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2021.111962.

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