Climate and local geomorphic interactions drive patterns of riparian forest decline along a Mediterranean Basin river

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Abstract

Dynamic fluvial processes strongly influence ecological communities and ecosystem health in riverine and riparian ecosystems, particularly in drought-prone regions. In these systems, there is a need to develop tools to measure impacts from local and regional hydrogeomorphic changes on the key biological and physical processes that sustain riparian ecosystem health and potential recovery. We used dendrochronology of Populus nigra, a riparian tree that is vulnerable to changes in local hydrology, to analyze ecosystem response following channel incision due to gravel mining along the Drôme River, a Mediterranean Basin stream in southern France. We cored 55 trees at seven floodplain sites, measured ring widths, and calculated basal area growth to compare the severity and timing of local growth decline along the river. Current basal area increment (BAI) growth per tree ranged almost 10-fold among sites (7.7±1.3 to 63.9±15.2 cm² year⁻¹, mean±SE) and these differences were significant. Mean BAI was correlated positively with the proportion of healthy trees at a site, and negatively with proportion of dead canopy area. Regime shift analysis of the tree-ring series indicates that tree growth declined significantly at four sites since 1978, coincident with documented channel incision. In addition, patterns of low growth and crown dieback are consistent with stress due to reduced water supply. The most impaired sites were not directly adjacent to local mining pits visible on aerial photographs, nor did the sequence of growth regime shifts suggest a pattern of channel incision progressing from these areas. The initiation of site growth declines was most typically associated with drought years, and the most impaired sites were spatially distributed to suggest the influence of local bedrock controls on soil depth. Climate in the Drôme basin and in the Mediterranean region is trending significantly toward hotter growing seasons with a decrease in summer river discharge, and this will increase both chronic and acute water shortage for riparian forests. This study shows that drought-prone riparian forests are vulnerable to hydrogeomorphological changes, but the severity of impacts is conditioned by interactions between drivers at different scales, including regional climate variability, reach-based geomorphic alteration, and local lithological controls.

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1. Introduction

Dynamic fluvial processes influence riparian forest and woodland ecosystems in numerous ways that affect the initial habitat conditions, resource fluxes, and ecological succession pathways. Important hydrogeomorphic processes include: (1) creating new growing space through bank erosion, horizontal channel migration, channel cutoff, and accretion of new floodplain surfaces (Piégay et al., 2005; Florsheim et al., 2008; Stella et al., 2011); (2) influencing the recruitment and survival of riparian seedlings through scour, burial and desiccation, all of which depend on the flooding regime and floodplain morphodynamics (Mahoney and Rood, 1998; Dufour et al., 2007; Moggridge and Gurnell, 2009); and (3) influencing groundwater table dynamics via channel bed incision and aggradation, which in general reduces or increases, respectively, water availability to the root zone (Scott et al., 2000; Pont et al., 2009).

In drought-prone regions, these hydrogeomorphic processes create riparian ecosystems that are particularly resource-rich, productive, structurally complex, and biodiverse compared with the surrounding landscape (Patten, 1998; Gasith and Resh, 1999; Underwood et al., 2009; Stella et al., 2012). Phreatophytic poplars and cottonwoods (Populus spp.), which dominate sub-humid riparian communities in much of the northern hemisphere (Karrenberg et al., 2002), are important foundational species (sensu Ellison et al., 2005) in providing or regulating water quality, microclimate, structural habitat for wildlife and fish, an energy base for the food web, and bank stability (Braatne et al., 1996; Lewis et al., 2009; Young-Mathews et al., 2010).

The distribution, composition and health status of Populus-dominated riparian forests are directly influenced by current and historical hydrology (Braatne et al., 1996; Rood et al., 2003) and geomorphic conditions (e.g., Cooper et al., 2003a; Stella et al., 2011). Over the last century,
Cooper et al., 2003b; Dufour et al., 2007. Temporary reductions in water availability can result in reversible changes such as slower growth and increased water use efficiency (Horton et al., 2001; Amlin and Rood, 2003; Rood et al., 2003; Lamb et al., 2006; Stella and Battles, 2010), but more severe and abrupt water table declines can cause permanent damage and high mortality (Tyrre et al., 1994; Scott et al., 1999; Rood et al., 2000; Stella et al., 2010). Plant water availability may be limited not only by modifying water supply, canopy demand and vapor pressure deficits (e.g., Cooper et al., 2003b), but also by altering geomorphological processes that affect groundwater depth and capillarity (e.g., Scott et al., 1999, 2000).

Negative impacts to riparian tree survival, growth and function are likely to continue as increasing human populations compete for water. Climate change will further alter the water balance in many areas (Steiger et al., 2005; Palmer et al., 2008; Perry et al., 2011). Because of its phreatophytic physiology and low tolerance to drought, sudden and severe damage to Populus stands often occurs when the water table declines beyond a threshold (Scott et al., 1999; Cooper et al., 2003b). Furthermore, local geomorphic changes in conjunction with water competition and climate change could act synergistically to injure Populus forests in areas where their separate impacts would be minor (Albertson and Weaver, 1945; Stromberg et al., 1996). The spatially and temporally discontinuous pattern of riparian forest decline after some channel incision events suggests that local geomorphic manipulation can moderate the impacts on phreatophytic floodplain trees (Graf, 1982; Scott et al., 1999; Amlin and Rood, 2003). However, few investigations of riparian forest decline have examined the simultaneous effects of hydrogeomorphic alteration, climate change, and local geomorphic factors (Palmer et al., 2009).

In this study, we investigated patterns of riparian tree growth over time, both for individuals and for whole stands, in the context of local setting, reach-wide geomorphic changes and climate change. The Drôme River in southeastern France provides an excellent opportunity to study these interactions at varying temporal and spatial scales. The Mediterranean region, including the Drôme basin, is undergoing climatic drying that is projected to increase in the 21st Century (Giorgi and Lionello, 2008; García-Ruiz et al., 2011). Along a 5-km reach downstream of the town of Luc-en-Diois, the river is situated in a narrow valley confined by marl and limestone deposits where the active channel was mined for gravel beginning in the 1970s and subsequently has incised along its length (Landon et al., 1998; Kondolf et al., 2002; Toone et al., 2012). Adjacent to the channel, riparian forest stands dominated by Populus nigra currently display patchy, local evidence of severe crown dieback (Gagnage, 2008; Dunford et al., 2009). This species is the dominant European poplar species and, because of its high ecological and economic value, has been the focus of international efforts to conserve its habitat and genetic diversity (LeFèvre et al., 1998; Hughes and Rood, 2003).

In this context we asked the general question, “How are patterns in tree growth and decline consistent with overall climate drivers versus local environmental conditions such as valley morphology and human-induced geomorphic changes?” Specifically, we investigated: (1) relationships between tree growth and independent measures of riparian stand health, and how these vary among sites within the reach; (2) trends in the spatial extent and timing of P. nigra growth decline within the reach; and (3) regional climate-based versus local site-based environmental correlates with growth patterns evident across the tree-ring chronologies. We expected that if gravel mining was the immediate cause of tree mortality, crown dieback and reduced growth, two predicted patterns in the tree-ring record should be evident. First, any growth declines should have occurred relatively suddenly and only after the initiation of gravel mining (approximately 1975–1980 from aerial photograph evidence; Landon et al., 1998). Second, floodplain sites closest to the pits should show the earliest and strongest declines, with more distal sites demonstrating later and/or more subtle declines. These spatio-temporal patterns are consistent with other studies of water table decline associated with in-channel mining (Scott et al., 1999; Amlin and Rood, 2003; Rollet et al., 2006) and channel bed incision from sediment starvation after dam construction (Kondolf, 1997).

In addition, the presence of both confling slopes on one side of the river channel and tributary confluences upstream of the reach may contribute to edaphic heterogeneity in the floodplain, with potentially local effects on riparian forest stands. Conversely, if tree growth along the Drôme River responds primarily to climate rather than local drivers, growth patterns should show a more regional signal that is coordinated with either short-term droughts (Singer et al., 2012), or long-term climate change in the Mediterranean basin (Giorgi and Lionello, 2008; García-Ruiz et al., 2011).

2. Materials and methods

2.1. Regional setting and study sites

The Drôme, which is a tributary of the Rhône River in southeastern France (Fig. 1), is a fourth-order stream with a mountainous pre-Alpine catchment of 1640 km² of Cretaceous age (elevation range 800–1400 m), that is underlain primarily by marl deposits in its valley with steep slopes rising to limestone ridges. Throughout the watershed, the climate follows a weakly Mediterranean pattern of warm, relatively dry summers and cool, moister winters. The study area receives 889 mm average annual precipitation (gauged at Luc-en-Diois; Fig. 1); at lower elevations in the watershed, winter precipitation falls as rain, but snow accumulates at higher elevations.

The catchment area for the study reach is approximately 225 km² (194 km² at the Luc-en-Diois gaging station) and annual discharge averages 2.4 m³ s⁻¹ (1961–2007 for Luc-en-Diois). Mean discharge in spring (March–May) is 3.8 (±1.7 SD) m³ s⁻¹ and exceeds the summer baseflow by an order of magnitude (0.6 ± 0.4 SD m³ s⁻¹ for July–Sept). Between 1948 and 2009, major floods occurred in 1951 (Q₁₀), 1978 (Q₇), 1994 (Q₁₀₀) and 2003 (Q₄). Major growing-season droughts, with multiple years of lower-than-average streamflow, occurred in 1964–65, 1989–91, and 2003–2006.

The study was undertaken within a 5-km study reach downstream of the town of Luc-en-Diois (Fig. 1). The channel in this reach is characterized by alternating straight and braided sections, an average longitudinal slope of 0.1%, an active width ranging from 10 to >200 m, and no significant bank protection (Toone et al., 2012). As a result, unconstrained channel migration occurs over the floodplain, which supports an extensive riparian forest that developed since the 1950s and is currently dominated by poplar (P. nigra), willow (Salix alba), ash (Fraxinus excelsior), and alder (Alnus glutinosa) (Pégay and Landon, 1997; Gagnage, 2008). The channel still experiences significant flood events (Landon et al., 1998; Kondolf et al., 2002), including one in December 2003 that resulted in failure of the bridge at Die.

The Drôme channel and floodplain were subjected to intensive gravel mining beginning in the 1970s. Gravel extraction through channel dredging and strip mining occurred at a rate of 250,000 m³ yr⁻¹ within and downstream of the study reach during this period, dwarfing the Drôme’s natural annual bedload flux, estimated at 35,000 m³ yr⁻¹ (Landon et al., 1998). Though the most concentrated activity was downstream of the 5-km study reach, several areas of intensive extraction within the study reach are evident on 1980 aerial photographs, but not on the previous series flown in 1971 nor the subsequent one in 1991 (Fig. 1; Toone et al., 2012). This activity was centered downstream of the Béoux confluence and downstream of the Recoubeau bridge. Geomorphic evidence of in-stream mining includes pits in the gravel bars, sections of straightened
low-flow channel, and local gravel levees built to protect the mining zones from flooding.

The study reach also experienced bed incision, as observed from comparing long profiles from 2003 to 2005 to one surveyed in 1928 by the Service des Grandes Forces Hydrauliques (Toone et al., 2012). Between 1928 and 2003, incision averaged 0.8 m but locally exceeded 2 m, and varied partly because of limestone outcrops that are discontinuous longitudinally and which anchor most of the single-channel reaches (Fig. 1;
Toone et al., 2012). Though the period 1928–2003 is too long to assess whether the recent gravel mining initiated channel bed incision, some of the most incised areas are associated with local mining sites, including the Recoubeau bridge, which was undermined during the mining period and subsequently protected by a concrete weir. Since the mid-1970s, this area experienced both channel incision that propagated upstream, and local aggradation (+ 0.33 cm between 1928 and 2003) near the weir after its installation in the early 1990s (Toone et al., 2012).

2.2. Field sampling

We sampled trees in seven floodplain stands located throughout the study reach (Fig. 1). These stands were selected by first examining a chronosequence of aerial photograph series to locate floodplain surfaces that had been created prior to 1971 but were still young enough to be dominated by P. nigra and other pioneer riparian trees. Conifer-dominated patches, which indicated xeric, high-elevation areas outside the riparian zone, were filtered out using an NDVI index on recent aerial photographs (Quickbird image flown 7 Dec. 2001). This process identified 11 mid-aged sites that ranged from 36 to 58 years old. From this group we selected seven sites for field sampling that were relatively well-distributed within the reach and that supported deciduous riparian forest stands dominated by P. nigra (Fig. 1; Gagnage, 2008; Lejot et al., 2011).

In each stand we surveyed the relative elevation of its floodplain surface above the current baseflow elevation using DGPS (Trimble 5800 Differential Global Positioning System) with a vertical resolution of < 10 cm, at 1–7 points (mean 3.4) per stand. Surveys were conducted from May to July 2008 on days when the river discharge was < 2 m³ s⁻¹. Using a soil corer, we measured the fine sediment thickness at 6 to 14 locations distributed across the stand.

At the center of each stand, we surveyed a circular plot of 500 m² to sample tree composition, density and crown health. Canopy cover was estimated visually for the entire stand using a range of values to represent the high heterogeneity in cover present. All canopy trees within the plot were measured and the species composition was recorded. From the center of each plot, we selected 20 dominant trees (i.e., those structuring the canopy) for which we estimated the proportion of live crown; these measurements were aggregated across the sample to assess the health of the stand (Scott et al., 1999).

The estimates of crown integrity were complemented using low-elevation aerial photograph analysis of the sites to identify the area of dead branches within the canopy (Gagnage, 2008; Dunford et al., 2009). In this method, digital photographs taken from an unmanned aerial vehicle were orthorectified to the project basemap and dead branches were identified using the object-oriented classification approach detailed in Dunford et al. (2009). Dead branch area was summed and divided by the sampling area for each of the stands to estimate the proportion dead canopy.

2.3. Dendrochronological analysis

Of the live trees in each stand, we selected the largest dominant P. nigra individuals for coring, regardless of health status. A total of 55 trees were cored across the seven sites; samples ranged 6–9 trees per site. Cores were extracted at breast height (1.4 m) using a 5-mm increment borer, with two cores collected per tree wherever possible (N = 114 cores).

Following standard dendrochronological procedures, we mounted cores and clarified ring anatomy by sanding each core with a series of progressively finer sand papers (Stokes and Smiley, 1968). Ring widths were measured to 0.001 mm resolution using a sliding stage micrometer (Velmix Inc., Bloomfield, NY) and Acu-Rite encoder (Heidenhain Corp., Schaumberg, IL). Potentially misdated cores were identified with the software program COFECHA (Holmes, 1983) and checked for false rings, missing rings, and other sources of dating errors. All ages reported are minimum ages, as they do not account for time for the tree to reach coring height or the time between the innermost ring on the core and the pith.

To examine long term trends in tree growth, we converted the raw ring widths to basal area increments (BAI) using the formula

\[
\text{BAI} = \pi \left( r_n^2 - r_{n-1}^2 \right)
\]

where \(r\) is the radius at the end of year \(n\). BAI typically increases as trees’ crowns expand or understory trees are exposed to higher light levels, and stabilizes or continues to increase in healthy, undamaged, mature trees (Biondi and Qeadan, 2008). The BAI series for multiple cores per tree were averaged, and then site chronologies were constructed by taking the arithmetic mean of all BAI series from the site for each year.

We analyzed the BAI chronologies using Rodionov’s regime shift detection (RSD; Rodionov, 2004) to explore the possibility of rapid and sustained shifts in the site mean growth rates, and to determine the timing of any shifts. The algorithm uses sequential t-tests to identify shifts in the mean without an a priori hypothesis of timing, and uses an index based on normalized anomalies to determine if the shifts are maintained. We applied the algorithm with the RSD add-in for Microsoft Excel 2002 (Rodionov, 2006a) with significance level = 0.05, cut-off = 10 years, Huber’s weight parameter = 2, and autocorrelation estimated by the inverse proportionality method on 7 year subsamples (Rodionov, 2006a,b).

To examine correlations with growing season climatic and hydrological variables, we used a data-adaptive detrended procedure within the program ARSTAN (Cook, 1985) to produce a chronology that emphasizes high-frequency variation (Cook and Peters, 1981). This series was more appropriate to use than the BAI series, because lower-frequency signals such as decadal growth trends due to site conditions would be preserved in the BAI chronologies and would obscure the high-frequency climate correlations (Cook et al., 1980). Nine of the 114 total cores were removed from this analysis due to poor cross-dating: in some cases, short series may have compromised the ability of statistical correlations to identify misdated cores (Grissino-Mayer, 2001). A Friedman variable span smoother (alpha = 5) was fit to each remaining series (Friedman, 1984), which was then divided by its fit curve to produce a time series of dimensionless index values; the autocorrelation was then estimated and removed from each series. All residual series were averaged by calendar year using a robust bi-weight mean to generate the final residual chronology. The chronology was truncated where the expressed population signal, a measure of chronology strength with maximum of 1, fell below 0.85 (Wigley et al., 1984).

We tested differences in recent growth among sites using analysis of covariance (ANCOVA). Recent growth, the response variable, was calculated by summing the BAI for the most recent four years (2004–07); this metric was log-transformed in the ANCOVA model to satisfy residual assumptions. Predictor variables in addition to site included tree age and early tree size, which was represented as basal area in 1980. This latter variable was included in order to control for differences in tree size at the start of the growth decline period, when larger trees would be expected to have a more extensive root system and therefore potentially greater access to water. To estimate individual tree basal areas at the time when growth declines might have begun, we summed for each core all ring widths prior to 1980, then averaged the radius estimates for all cores within trees, and converted these to basal areas.

2.4. Climate data analysis

Climate data for the period 1961–2007 were obtained from several sources in order to evaluate the climate signal within the tree-ring chronologies. Monthly maximum and minimum temperature data (i.e., daily highs and lows averaged by month) were extracted from the E-OBS dataset (Haylock et al., 2008), which are gridded (0.25° cell) interpolations from regional station data with missing data estimated and stations with > 20% days with missing data excluded. Daily precipitation totals

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Table 1: Characteristics of the *Populus nigra* stands studied along the Drôme River.

<table>
<thead>
<tr>
<th>Site Number</th>
<th>Site Floodplain</th>
<th>Floodplain channel (m) mean ± SD</th>
<th>Fine sediment depth (cm, mean ±1 SE)</th>
<th>Dead canopy branches %</th>
<th>Density of canopy trees (trees ha⁻¹) mean ± 1SE</th>
<th>Diameter of cored trees (cm, mean ± 1SE)</th>
<th>No. trees cored</th>
<th>Age of cored trees (years, median, min and max)</th>
<th>Basal area increment (cm², mean ± 1SE)</th>
<th>Recent BAI (cm² year⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>1.55</td>
<td>2.09</td>
<td>70.3 (±14.5)</td>
<td>70%</td>
<td>740</td>
<td>47.4 (±16.4)</td>
<td>8</td>
<td>25%</td>
<td>47.4 (±16.4)</td>
<td>8</td>
</tr>
<tr>
<td>2</td>
<td>2.09</td>
<td>3.13</td>
<td>46.5 (±6.9)</td>
<td>1%</td>
<td>35.5 (±14.1)</td>
<td>1.55 (±5.4)</td>
<td>6</td>
<td>25%</td>
<td>35.5 (±14.1)</td>
<td>6</td>
</tr>
<tr>
<td>3</td>
<td>1.45</td>
<td>4.65</td>
<td>61.0 (±10.2)</td>
<td>0%</td>
<td>47.4 (±4.6)</td>
<td>1.55 (±5.4)</td>
<td>9</td>
<td>25%</td>
<td>47.4 (±4.6)</td>
<td>9</td>
</tr>
<tr>
<td>4</td>
<td>2.79</td>
<td>5.42</td>
<td>47.1 (±16.4)</td>
<td>80%</td>
<td>35.5 (±14.1)</td>
<td>1.55 (±5.4)</td>
<td>9</td>
<td>25%</td>
<td>35.5 (±14.1)</td>
<td>9</td>
</tr>
<tr>
<td>5</td>
<td>0.73</td>
<td>3.42</td>
<td>54.2 (±11.2)</td>
<td>0%</td>
<td>47.4 (±4.6)</td>
<td>1.55 (±5.4)</td>
<td>9</td>
<td>25%</td>
<td>47.4 (±4.6)</td>
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</tr>
<tr>
<td>6</td>
<td>1.87</td>
<td>2.50</td>
<td>25.0 (±5.8)</td>
<td>0%</td>
<td>47.4 (±4.6)</td>
<td>1.55 (±5.4)</td>
<td>9</td>
<td>25%</td>
<td>47.4 (±4.6)</td>
<td>9</td>
</tr>
<tr>
<td>7</td>
<td>1.09</td>
<td>1.17</td>
<td>15.5 (±3.7)</td>
<td>0%</td>
<td>47.4 (±4.6)</td>
<td>1.55 (±5.4)</td>
<td>9</td>
<td>25%</td>
<td>47.4 (±4.6)</td>
<td>9</td>
</tr>
</tbody>
</table>

were obtained for the Luc-en-Diois station (Météo France station #26167001) and daily river discharge values from the Luc-en-Diois gauging station (WV4214010, French National Office for Water and the Aquatic Environment). Monthly metrics were calculated by averaging the daily discharge values and summing the daily precipitation totals.

We used Pearson correlations to assess the relationship between annual radial growth, as indicated by the high-frequency chronology, and both monthly and growing season climate and discharge variables. Because conditions during the year prior to ring formation often influence ring width (Fritts, 1976), we used a 19-month window beginning in the previous April and extending to the current October to assess annual growth correlations with monthly values for discharge, precipitation, and minimum and maximum temperatures. The best-correlated months were further aggregated as a single period (i.e., growing season) to compare the sensitivity of tree-ring growth among climate metrics. The monthly discharge values had substantially unequal variances, therefore we transformed all data metrics to standard scores (i.e., z-scores) prior to averaging months together for the growing season. Standardization was necessary because the unequal variances could lead to biased growing season discharge estimates whereby only one or two months dominated the seasonal signal. The discharge standard scores were calculated

\[ z = (x - \mu) / \sigma \]  

where \( \mu \) and \( \sigma \) are the overall mean and standard deviation, respectively, for a particular month throughout the period of record and \( x \) is the monthly value in a given year. Following transformation, monthly scores were then averaged for the growing season, June through September.

We used a modified Mann–Kendall test proposed by Yue et al. (2002) to estimate the significance of trends in monthly and seasonal climatic and discharge variables over the period 1961–2007. In that procedure, trend is first estimated as the median slope of all possible point pairs (Theil, 1950a–c; Sen, 1968) then removed from the time series. Next, autocorrelation is modeled as a lag-one autoregressive process and also removed from the time series. Finally, the trend is mixed back into the time series, and significance of the trend is assessed by the Mann–Kendall test (Mann, 1945; Kendall, 1975). We completed all trend analysis with the Zyp package in R (version 2.12.1, R Development Core Team).

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**Fig. 2.** Boxplots of recent average growth (basal area increment means for 2004–07) for *Populus nigra* trees at the seven Drôme River floodplain sites. Thick solid lines indicate site means, boxes indicate 25th and 75th quartiles, and whiskers span the range of the data. Different letters at the top denote significant differences between sites based on the Tukey Honestly Significant Difference test and a threshold p < 0.05.
3. Results

3.1. Site conditions, tree health, and recent growth

The seven sites are located on floodplain surfaces 0.7 to 2.8 m above the river channel (Table 1), with fine sediment depth above the coarse gravel layer ranging from 15.5±3.7 to 70.3±14.5 cm (mean±SE). Tree density among the seven sites ranged from 240 to 1340 trees per hectare, corresponding to canopy cover ranging from sparse (0–25%) to closed (90%). Tree diameters ranged from 31.5 (±2.9 SE) to 54.5 (±5.6 SE) cm dbh, and median minimum tree age ranges from 36.5 to 58 years among the sites. Healthy *P. nigra* trees (i.e., those with <50% of branches dead) were uncommon at some sites, but represented over 75% of individuals at other sites. The number of trees with severe crown damage (>50% branches dead) ranged from 6% to 65% among sites. Cores from *P. nigra* trees showed substantial site-based variability in current growth. Mean basal area increment for the last four years varied by an order of magnitude among sites, from 7.7±1.3 to 63.9±15.2 cm² year⁻¹ (mean±SE). The great range in recent radial increment includes three sites sustaining high growth (Sites 1, 2, and 5), two maintaining moderate growth (Sites 4 and 7), and two others (Sites 3 and 6) yielding substantially lower average growth among trees (Fig. 2). The ANCOVA model indicated that growth differences were significant among sites (*F*₆,₄₆ = 8.56, *p* < 0.0001), and post-hoc Tukey tests of 95% family-wise confidence levels indicate that Sites 3 and 6 were significantly different from Sites 1, 2 and 5 (Fig. 2). The continuous covariates in the ANCOVA were...
not significant, including tree age ($F_{1,46} = 0.35, p = 0.56$) and tree size normalized to estimated 1980 basal area ($F_{1,46} = 2.36, p = 0.13$); nor were any variable interactions significant. Therefore site-based growth differences were not influenced by variation in the intrinsic biotic factors of tree age and size early in stand development.

At the site level, recent radial growth was positively correlated with the proportion of healthy (i.e., minimal branch mortality) *P. nigra* trees ($r = 0.56$; Fig. 3a), and negatively correlated with the proportion of trees with >50% dead crown ($r = -0.72$; Fig. 3b), and the proportion of the stand’s aerial image comprising dead branches in the upper canopy ($r = -0.69$; Fig. 3c). Metrics of stand vigor were positively correlated with tree density, with the sparsest stands supporting the least productive and healthy trees. For example, stand density was positively correlated with tree diameter ($r = 0.70$; Fig. 4a), and recent growth ($r = 0.77$; Fig. 4b), and negatively correlated with the proportion of trees with severe crown damage ($r = -0.83$; Fig. 4c).

Of the geomorphic variables measured, elevation above the channel was correlated positively with areal proportion dead canopy in the stand ($r = 0.74$), indicating greater damage in higher stands. However, there was no relationship evident between recent growth and elevation above the channel ($r = -0.16$), nor between recent growth and maximum bed incision in the adjacent channel calculated from the 1928 and 2003 long profiles ($r = -0.09$, $t = -0.21$, df = 5, $p = 0.84$).

### 3.2. Dendrochronological and growth results

Variability among the BAI site chronologies increased over the time period examined (1961–2007), and different sites show contrasting trends (Fig. 5). Prior to the mid-1970s, the chronologies show either little trend in growth or an increase, which is typical of younger trees still undergoing crown expansion. All sites sustained comparable growth during this period of 30–100 cm$^2$ average BAI per tree. After that time, variance...
among chronologies increased and sites diverged steeply in average growth, with Sites 1, 2, and 5 maintaining 50–100 cm² BAI per tree to the present, and the other sites showing growth declines at various times.

Regime Shift Detection (RSD) analysis identified significant shifts in annual growth rate throughout the study period, both in individual trees and in site chronologies (Fig. 6). Chronologies for sites 1, 2 and 5 exhibited no regime shifts, and growth was relatively steady through time. The RSD analysis detected significant declines at Site 3 (in 1978), Site 7 (in 1990), Site 4 (in 1995), and Site 6 (in 1997), as well as an increase at Site 7 in 1999 following its decline nine years earlier. At all sites, individual trees displayed either upward or downward shifts, but not both (Fig. 6).

3.3. Climatological correlations with annual growth and climate trends

P. nigra radial growth responded positively to monthly mean stream discharge during the growing season, especially June through September ($r = 0.26–0.32$; Table 2). Responses to discharge during the dormant season are generally negative ($r = −0.18–0$), but are much weaker than the response to growing season discharge. Correlations with monthly precipitation totals produce a similar pattern to discharge, but the growing season response varied more among months ($r = 0.09–0.28$; Table 2). The correlation analysis detected no significant relationships between growth and either monthly mean minimum or maximum temperature.

Of the aggregated growing season climate variables, river discharge and precipitation correlated significantly with growth ($r = 0.40$; Table 2). Over the 1961–2007 period, the Drôme River basin became significantly hotter and river discharge has decreased. Average discharge between June and September at the Luc-en-Diois gauge has declined by an average of 0.09 m³ per second per decade (τ = −0.21, $p = 0.04$; Fig. 7a), mean June–September precipitation has been stable (τ = 0.0087, $p = 0.94$; Fig. 7b), daily minimum temperature has increased by 0.16 °C per decade (τ = 0.27, $p = 0.008$; Fig. 7c), and daily maximum temperature has increased by 0.40 °C per decade (τ = 0.36, $p < 0.001$; Fig. 7d).

4. Discussion

4.1. Patterns of poplar decline along the Drôme River

Riparian forest stands within the Drôme River have experienced sharp growth declines of their P. nigra trees in some areas, in conjunction with

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**Fig. 6.** Regime shift detection (RSD) analysis for each site. Each panel shows the site-averaged basal area increment chronology (thin line, left vertical axis), the RSD curve for the site mean (thick line, right vertical axis), and the RSD break points for individual trees (circles, right vertical axis).
poor stand health. Long-term growth trends showed strong site-based differences, varying among steady growth (Sites 1, 2 and 5), sustained low growth and/or sudden decline following dry years (Sites 3, 4 and 6), and initial decline and recovery (Site 7). Canopy and stand health were strongly correlated with basal area increment (Fig. 3); therefore these patterns do not merely reflect random variation in radial growth.

These symptoms of forest stand decline are coherent with impairments from water limitation, and this finding is consistent with research on other declining riparian Populus stands throughout the Northern Hemisphere (Stromberg and Patten, 1990; Scott et al., 1999; Amlin and Rood, 2003; Lamb et al., 2006; Hultine et al., 2010). No available evidence supports an alternative cause of the decline in tree health on the Drôme floodplain apart from water scarcity. Patterns of decline are not consistent with biotic factors such as intrinsic life history, stand dynamics, or herbivory, disease or pathogens. Trees at all sites were too young for natural declines in vigor to have occurred across so much of the study reach (Table 1; Braatne et al., 1996), and recent growth was unrelated to tree age or their size near the time when declines began. Therefore, variation in recent growth was not a function of inherent development or tree architecture. Density-dependent competition for resources (e.g., light or soil nutrients) was not likely a limiting factor either, because all available signs of stand health were positively, not negatively correlated with tree density (Fig. 4). The densest stands sustained the largest trees, highest annual productivity and best canopy health. These relationships suggest that higher tree density is a result, rather than a cause of different stand conditions, and that these differences are driven by density-independent controls such as abiotic stress rather than density-dependent resource competition among individuals (Braatne et al., 1996; Stromberg and Patten, 1996).

4.2. Evidence for local and reach-scale drivers of tree decline

Broad-scale deterioration in stand health within this reach of the Drôme River is consistent with rapid changes in water availability, including hypothesized groundwater decline associated with the channel incision that occurred over the last several decades (Landon et al., 1998; Kondolf et al., 2002). Other reaches of the Drôme also experience tree stress responses to hydrogeomorphic changes, namely severely reduced diameter growth in ash (F. excelsior) trees several kilometers upstream of the study reach in areas with local aggradation and presumably higher levels of flooding and anoxic stress in the root zone (Pont et al., 2009; Rodríguez-González et al., 2010).

Within the present study, significant growth reductions (i.e., negative regime shifts) occurred only since the initiation of intensive gravel extraction activity in the early 1970s (Fig. 7; Landon et al., 1998). Though the RSD method has a limited ability to detect declines in the early years of a chronology, visual inspection of the site growth series indicates no major reductions in growth prior to 1975 (Figs. 5–6). In-stream gravel mining introduces several changes in channel morphology and hydrogeomorphic processes (Kondolf, 1997). These include a local deepening of the channel with direct effects on the low flow levels and associated groundwater, an increased slope at the upper edge of the mining zone, and a loss of bedload supply within the reach. These changes initiate a disequilibrium between sediment transport capacity, which is increased, and sediment supply, which is reduced,

<table>
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<th>Month</th>
<th>Mean discharge (m$^3$ s$^{-1}$)</th>
<th>Cum. precipitation (mm)</th>
<th>Mean minimum daily temperature ($^\circ$C)</th>
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and typically results in local channel bed incision that propagates downstream of the mining site, and a regressive erosion upstream that eventually stabilizes the local channel slope. Cascading effects of channel incision on floodplain habitats due to mining has been also showed by Rollet et al. (2006) on the Doubs River, in which incision propagated both upstream and downstream of mining sites. Channel bed incision also tends to propagate downstream locally from dams for similar reasons (Ronald, 1997).

Though channel incision and associated groundwater decline would be expected to have system-wide effects (i.e., affect all sites), they are also expected to generate coherent spatial and temporal patterns in the relative response of individual sites. For example, we expected floodplain stands adjacent to gravel extraction areas on the Drôme to decline most strongly and earlier as they experience the effective deepening. Scott et al. (1999) documented that widespread cottonwood mortality and crown damage occurred within two years of intensive sand mining in the adjacent channel, and that effects of depressed groundwater and tree decline were most severe closest to the excavation site. Amlin and Rood (2003) observed extensive leaf senescence, abscission, and reduced stomatal conductance in a stand of cottonwoods adjacent to an excavated gravel pit that depressed the water table by 2.5 m, but saw no effect in a reference stand farther away. On the Drôme, we expected not only this proximity effect, but also that the sequence of impacts would progress from the near-pit sites to farther ones upstream and downstream over time with propagation of channel bed incision and associated water table lowering.

Contrary to expectations, only one stand’s regime shift coincided with incision following mining (Site 3, with a negative regime shift in 1978), whereas the others occurred more recently in the 1990s (Fig. 6). Site 3 is not the closest to a mining site, which would have explained its early and severe growth decline, and it is adjacent to Site 4, which did not experience a regime shift until 1995 (Fig. 6). The two most upstream areas, Sites 1 and 2, are adjacent to a mining site (Fig. 1), yet they experienced no decline and currently support healthy, dense forest stands with vigorous tree growth (Figs. 5–6). The downstream stands closer to other gravel pits had more variable growth, with one sustaining stable growth (Site 5), one in severe decline (Sites 6), and one that experienced decline then recovery (Site 7). Overall, the sites with the most severe declines were distributed throughout the reach, and their sequence of negative growth regime shifts started with Site 3 (1978), followed by Site 7 (1990), Site 4 (1995) and Site 6 (1997) (Fig. 8). These examples cumulatively indicate that, in contrast to the expected sequence of impacts, growth declines did not occur exclusively, or earliest, at sites close to the gravel extraction activities, nor were impacts sequenced as would be expected with incision propagating upstream and downstream of local mining (Ronald, 1997; Scott et al., 1999; Fiorsheim et al., 2001; Marston et al., 2003). Therefore, gravel mining alone cannot explain the finer-scale (within-reach) stand deterioration patterns, and suggests that other factors contributed significantly to the pattern of local impacts.

However, other localgeomorphic influences in addition to gravel mining are operating within the study reach, and these do show coherence with the spatial pattern of growth declines. For example, it is notable that three of the four most heavily-impaired sites are on the right bank of the river (Sites 3, 6, and 7) in an area with shallow alluvium and marl bedrock outcrops near the surface (Fig. 9). All of these sites maintained relatively low baseline growth (<50 cm² year⁻¹ average BA; Fig. 5) during their most productive years, and experienced negative regime shifts between 1978 and 1997 (Fig. 6). Site 4, on the left bank in this reach, had maintained substantially higher baseline growth than the other three and also experienced a sharp decline (Fig. 5). Thus, the sites with the shallowest depth to bedrock were the most vulnerable, which is consistent with incision and water table recession occurring generally throughout the reach and preventing shallowly-rooted stands (due to bedrock constraints) from accessing perennial groundwater. Additionally, the positive regime shift and complete recovery in growth rate that occurred at Site 7 after 2000 is compelling evidence of local geomorphic drivers, as it occurred adjacent to the recent aggradation of the channel bed following construction in the 1990s of the protective weir at the Recoubeau bridge (Figs. 1 and 9). This suggests that, as river stage recovered, nearby trees were able to access a more regular groundwater source and thus grow vigorously again.

Populus and other phreatophyte pioneer species have some ability to compensate for water scarcity by growing roots deep into alluvial soil horizons where the groundwater supply is dependable (Rood et al., 2011; Singer et al., 2012). However, when water scarcity is sudden and prolonged, or else when bedrock or other confining soil layers (e.g., hardpan) prevent roots from accessing the perennial water table, the trees’ high canopy demand and poor drought tolerance mechanisms often result in increased xylem cavitation, crown dieback and tree mortality (Scott et al., 1999; Cooper et al., 2003b; Rood et al., 2003). Conversely, when water resource conditions improve, for example by raising the groundwater level or increasing the root zone through sediment deposition, previously stressed riparian trees are capable of full recovery (e.g., Hultine et al., 2010). This process is apparent at the downstream site, which recovered following base-level rise at the new weir. If groundwater recovery is so great as to inundate the rooting zone for prolonged periods, trees can also experience water excess and slow growth (Kozlowski, 2002; Pont et al., 2009; Rodríguez-González et al., 2010).

4.3. Climate and discharge trends, and relationship to tree growth

Climate and river discharge also appear to have exerted an influence on riparian tree growth on the Drôme River over the last 50 years. Correlations between the detrended ring-width chronology and physical variables were strongest for growing season months (June–September), and significant for both discharge and precipitation (Table 2). Thus these trees are sensitive to hydroclimatic variables even though they grow in a habitat where water is abundant relative to the surrounding landscape (Stromberg and Patten, 1996; Stella et al., 2012). In Mediterranean-type regions, streamflow and water table levels are at their annual minima during the growing season, and as a result, abiotic stress on aquatic organisms is highest during this period (Gasith and Resh, 1999). Therefore, riparian trees experience stress that varies annually with hydroclimatic water supply, similar to sympatric upland species (e.g., pines and oaks) that are adapted to drier habitats and that are more typically used for dendrochronological studies (e.g., Cherubini et al., 2003; Meko and Woodhouse, 2005). In contrast to discharge and precipitation,
growth correlations with temperature were poorer and not significant (Table 2). Though riparian tree phenology in Mediterranean-type regions is sensitive to seasonal temperature differences (Stella et al., 2006), temperature is not typically limiting to growth for most of the leaf-on period, except for high daytime vapor pressure deficits in mid-summer that induce stomatal closure (Lambs et al., 2006; Hultine et al., 2010). Therefore, temperature is likely limiting primarily through its effects on water demand.

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Despite a somewhat limited sample size, the temporal pattern of growth declines in individual trees and sites suggests a general correspondence with severely dry years or longer droughts (Fig. 8). The trees at Site 3 initiated a growth decline in 1978, a year with average summer discharge, but this followed severe droughts in 1974 and 1976 and very high annual variability throughout the 1970s (Fig. 8). This period also coincides with the intensive gravel mining activity and, assuming that channel incision had occurred, trees would have been predisposed to water stress. Negative regime shifts for individual trees also tended to start in drought years, though in some cases they began the year after, and in other cases do not appear related to discharge.

Factors other than climate and geomorphic change may also have influenced the stand-level pattern of growth regime shifts. One methodological issue is that the regime shift detection method recognizes step changes in time series, which can mask the timing of a more gradual and sustained trend. Most trees and sites show fairly sudden rapid shifts in growth rate (e.g., Sites 3, 4, and Site 7 recovery in 1999), but others (Sites 6 and 7 declines) show some evidence of a more gradual decline that initiated in the late 1970s or early 1980s (Fig. 5). Another reason for the asynchrony in timing of site growth declines with climate and geomorphic change may have to do with the differential response of individual trees at a site. There is great variability in both early growth rate and onset of slow growth among trees at a site (Fig. 5), suggesting that background site conditions were quite varied spatially, and that following the onset of multiple environmental stressors, site conditions for trees reached negative thresholds for growth at different times.

4.4. Synergistic drivers and global change

Though neither channel and groundwater alteration from gravel mining nor the sequence of short-term droughts is sufficient alone to explain the fine-scale patterns of forest stand decline along the Drôme River, there is compelling evidence that these factors acted jointly to influence the timing and location of the strongest impacts. First, the recentness of all stand-level declines (i.e., since the early 1970s) suggests that channel bed incision and associated groundwater table decline in adjustment to mining contributed to overall forest vulnerability throughout the reach. Secondly, dry years following the early 1970s appeared to provide environmental triggers that induced major growth declines. For example, the first decline at Site 3 (1978) occurred after several major dry years (1974 and 1976), and the declines at Sites 4, 6 and 7 were associated with individual dry years that occurred subsequent to the mining period (Fig. 8). Finally, the positive regime shift at Site 7 in 2000 following construction nearby of the Recoubeau bridge weir (Figs. 1 and 9c) suggests that local geomorphic manipulation can reverse growth declines by improving sediment/groundwater conditions. Together, these examples demonstrate the high sensitivity of forest stands to resource availability from a combination of drought and local geomorphic factors that include reach-wide incision, locally-induced aggradation, and bedrock controls on soil depth.

It is clear that, in addition to these stressors, the Drôme basin is undergoing long-term climate change to a hotter, drier growing season and discharge regime (Fig. 7). Though precipitation trends were not significant at the Luc-en-Diois station (Fig. 7b), precipitation has declined significantly over the basin as a whole (E- OBS gridded data not shown). These trends are expected to continue and accelerate in the future throughout the Mediterranean Basin (Giorgi and Lionello, 2008; García-Ruiz et al., 2011). For mature riparian forest stands, it is likely that local water availability will decrease as the river stage and groundwater table recede farther, faster, and more frequently during the growing season. These processes will stress mature trees that have developed root distributions, crown size and other adaptive morphological traits in response to wetter conditions such as higher water tables and less variability (Rood et al., 2011). Climate change impacts can be expected to be most severe at sites with local geomorphic and/or geologic controls. Existing pioneer stands may decline or transition to other vegetation types faster and irreversibly under these conditions, and new stands may have trouble establishing (Shafroth et al., 2000). Riparian studies in other water-limited ecosystems show a common response of forest decline following rapid hydrologic change (Stromberg and Patten, 1996; Scott et al., 1999; Dufour and Piégay, 2008), and in some cases an irreversible shift to alternative system states (Stromberg, 2001; Palmer et al., 2008). This process has occurred locally on the Drôme (Sites 3, 4, and 6 in particular), in which stands that sustained high and variable growth initially switched to permanently suppressed growth with little variation annually or amongst trees. Though there remains some uncertainty as to the specific ecological trigger(s) that would precipitate widespread mortality or growth declines in any one location (Goffman et al., 2006), it is clear that the climate sensitivity and accelerating hydroclimatic change, combined with multiple local stressors due to geologic setting and human-induced changes, will make drought-prone riparian zones exceedingly vulnerable in the future (Rood et al., 2008; Perry et al., 2011).

5. Conclusions

In the case of Drôme River poplars, major declines in growth were coincident with a period of reach-based geomorphic alteration from instream gravel mining, and these impacts were most severe on floodplain surfaces with thin alluvial substrates overlaying shallow bedrock. Furthermore, these declines were triggered for many trees in drought years, and the climate in the region is trending significantly toward hotter growing seasons with a decrease in summer river discharge. These trends, which are projected to continue, will increase both chronic and acute water shortage for riparian trees in drought-prone ecosystems, with phreatophytic species such as P. nigra particularly vulnerable. Chronic stress from climate change may induce growth declines sooner and more severely in the future.

This study has shown that tree-ring growth is a sensitive indicator of water stress for riparian trees, and is a useful tool to understand the effects of hydrogeomorphic controls and human modifications on river ecosystems. Riparian forests in semi-arid systems are sensitive to many physical drivers and can demonstrate threshold responses to multiple stressors. Pioneer trees respond to decreased water availability through greater mortality and crown loss, and decreased radial growth. However, the severity of these impacts depends on interactions with physical drivers at multiple scales, and not all floodplain sites are affected equally. Site-based variation in environmental conditions needs to be taken into account to understand specific vulnerability to hydrogeomorphic alteration.

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References


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