

The effects of land use changes on streams and rivers in mediterranean climates

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Abstract We reviewed the literature on the effects of land use changes on mediterranean river ecosystems (med-rivers) to provide a foundation and directions for future research on catchment management during times of rapid human population growth and climate change. Seasonal human demand for water in mediterranean climate regions (med-regions) is high, leading to intense competition for water with riverine communities often containing many endemic species. The responses of river communities to human

alterations of land use, vegetation, hydrological, and hydrochemical conditions are similar in mediterranean and other climatic regions. High variation in hydrological regimes in med-regions, however, tends to exacerbate the magnitude of these responses. For example, land use changes promote longer dry season flows, concentrating contaminants, allowing the accumulation of detritus, algae, and plants, and fostering higher temperatures and lower dissolved oxygen levels, all of which may extirpate sensitive native species. Exotic species often thrive in med-rivers altered by human activity, further homogenizing river communities worldwide. We recommend that future research rigorously evaluate the effects of management and restoration practices on river ecosystems, delineate the cause–effect pathways leading from human perturbations to stream biological communities, and incorporate analyses of the effects of scale, land use heterogeneity, and high temporal hydrological variability on stream communities.

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Introduction

Profound and far-reaching impacts of humans on the environment are mediated through land use changes which destroy, degrade, and fragment habitat, and constitute the primary cause of losses in biodiversity

worldwide (Sala et al., 2000; Chapin et al., 2001). Human impacts on landscapes often diminish the capabilities for ecosystems to provide essential services for people, including clean air and water and natural products (Foley et al., 2005). Land use changes have had a large effect on terrestrial and aquatic environments in mediterranean climate regions (med-regions), that is the Mediterranean Basin, California-Baja California, southwestern Australia, southwestern South Africa, and central Chile, because of their long history with human activity, rapid urban and agricultural development, and large climatic and topographic variation (Underwood et al., 2009; Sirami et al., 2010). Because med-regions contain many endemic species, human impacts on these regions have a disproportionate effect on global biodiversity and make mediterranean ecosystems (med-ecosystems) among the “most imperiled” in the world (Myers et al., 2000; Horwitz et al., 2008; Underwood et al., 2009). Although considerable research has been conducted on the effects of land use change on terrestrial environments in med-regions, studies of land use impacts on aquatic habitats are more limited. Because water is seasonally scarce in med-regions, many rivers and streams have been dammed or diverted to provide water supplies for human activities (Gasith & Resh, 1999). Given the critical importance of water quality and quantity in med-regions and concerns about the impacts of human environmental alterations on aquatic systems and species, there is a need for more data and knowledge to guide management and policy decisions balancing human and environmental needs for sustainable water resources. Research in other temperate regions of the world has shown that land use changes denude native vegetation, increase runoff and erosion, alter stream geomorphology and substrata characteristics, modify flow regimes, and enhance the transport of nutrients, sediment, and contaminants from catchments to receiving waters, all with numerous implications for the stream and river biota (Paul & Meyer, 2001; Allan, 2004; Walsh et al., 2005; Johnson & Horst, 2010). Human land use impacts on med-ecosystems are predicted to be heightened by climate change which will result in warmer, drier, and more variable weather (Klausmeyer & Shaw, 2009).

In this article, we review and synthesize the literature on land use impacts on mediterranean rivers (med-rivers) to provide a foundation for evaluating current and predicted human effects on streams and

rivers in med-regions. Furthermore, we conduct comparisons to determine if land use effects on stream and river ecosystems are different in med and non-med systems, particularly given the large seasonal and interannual variability in rainfall, runoff, and stream discharge observed in med-basins, and end with research recommendations (Gasith & Resh, 1999).

The history of land use change in mediterranean regions

Humans have a long history with areas now occupied by med-ecosystems. Some of the earliest hominids occurred in South Africa and early African *Homo* species expanded into the Mediterranean Basin nearly a million years ago, with European populations following the retreat and advance of glaciers. From its African origins, *Homo sapiens* expanded through Europe and Asia, eventually reaching Australia by 40,000–60,000 years ago and the Americas by about 12,000 BP (Fagan, 2003; Hassell & Dodson, 2003). Early human populations in med-regions engaged in hunting, fishing, and gathering activities, up to the time of European conquest in the Americas, southern Africa, and Australia. Early hunter-gatherers in med-regions affected landscapes primarily by burning vegetation, but also by harvesting wood and maintaining paths and wells, creating a mosaic of patches in different successional stages after disturbance (Simonet & Cornejo, 1990; Keeley, 2002; Abbott, 2003; Arnesto et al., 2010; Bowman et al., 2011; Faivre et al., 2011; Gammage, 2011; Neumann et al., 2011). Furthermore, the Khoisan in southwestern South Africa focused on livestock husbandry beginning about 2,500 years ago, and the Mapuche of central Chile became small-scale farmers ~1,400–2,000 years ago, with both using fire to create pasture or farmlands. Although many med-regions originally contained extensive tree stands, frequent fires started by native peoples apparently destroyed and prevented the re-establishment of forests and, in extreme cases, even shrubs, promoting the expansion of grasslands and degraded scrub (e.g., *Acacia caven* Molina savanna in Chile, garrigue in the Mediterranean Basin) (Keeley, 2002; Gammage, 2011). These people created more open terrain which facilitated European settlement in Australia, South Africa, and the Americas (Hallam, 1975; Timbrook et al., 1982; Gammage, 2011). It

appears that ancient land use practices are reflected in today's landscape patterns (Keeley, 2002), with many areas continuing to be subjected to intensive livestock grazing or agricultural, mining, urban, and/or industrial development.

The most intensive and complex impacts of ancient humans on landscapes occurred in the Mediterranean Basin. Originally, much of the Mediterranean Basin was covered by pine, oak, and cedar forests; however, the expansion of human settlements and agricultural activity ~5,000–2,000 years ago resulted in declines in forests and the expansion of shrublands and grasslands (Hooke, 2006). These changes in native vegetation occurred at different times in different places, with the Minoans deforesting Crete by 1200–1500 BC with later, similar changes in ancient Greece, where sacred groves were protected. Land use changes accelerated during the expansion of the Roman Empire, resulting from the local use of forests for firewood, from their large-scale conversion into farming or grazing lands to produce wine, oil, wheat, wool, and meat, and from mining activities, such as in the pyrite belt of southwestern Iberia, which stripped soils of vegetation (Perlin, 1989; Sabater et al., 2003; Sirami et al., 2010). Associated with these activities were Roman, and later Muslim, alterations to the water cycle, including the construction of dams, weirs, wells, aqueducts, waterwheels, canals, and other features which changed river geomorphology, re-routed and de-watered stream channels, created barriers, and covered springs (Fagan, 2011). These ancient modifications of the landscape continued during Medieval times as trees were felled for firewood, and farming and grazing activities expanded, with erosion being especially exacerbated by terrace farming, resulting in some cases in the choking of stream channels by sediment (Hooke, 2006).

With the expansion of agricultural, industrial, mining, and urban activities after European colonization of South Africa, Australia, and the Americas, and particularly after the Industrial Revolution in all med-regions, med-landscapes and their drainage rivers became increasingly altered (Arnesto et al., 2010). Rivers were tapped to produce energy and water for cities, farmlands, and industrial colonies (Sabater et al., 2009). Historical land use changes denuded landscapes of native vegetation and promoted erosion, flooding, and downstream sedimentation, often producing the altered environments evident today.

Current land use patterns and changes in mediterranean regions

Human population growth and land use changes have greatly altered landscapes in med-regions. Average human population density ranges from 5 people/km² in Australia to ca. 50 people/km² for Chile and South Africa to ca. 120 people/km² in the Mediterranean Basin and 250 people/km² in California-Baja California (Underwood et al., 2009), with urban density exceeding 200 people/km² in southern Europe and over 2,000 people/km² in the Los Angeles River basin, California (Table 1). From 1990 to 2000, human population density and urban area increased by 13% and agricultural area by 1% in med-regions throughout the world, with the greatest proportional changes for population density occurring in Chile (19%) and for urban and agricultural expansion in the Mediterranean Basin (+17% change in urban areas, +1% in agricultural areas) (Underwood et al., 2009). Currently, average per cent coverage of the landscape by urban areas range from 0.3% for Australia to 9.2% for California-Baja California, with the other three regions (Mediterranean Basin, central Chile, southwestern South Africa) having urban coverages of 0.7–1.5% (Underwood et al., 2009). Percent coverage by intensive agricultural areas ranges from 6% in California-Baja California through 24 to 29% in Chile, South Africa, and the Mediterranean Basin to 37% in Australia. The present percentages of the sum of urban, farm, and pasture lands covering large mediterranean catchments (med-catchments) range from 15 to 69% in the Mediterranean Basin, from 10 to 62% in southern California, from 8 to 70% in central Chile, from 5 to 66 % in southwestern South Africa, and from 31 to 86% in southwestern Australia (Table 1). Land use changes have been greater at lower than higher elevations, with the only substantive development at elevations >1,000 m occurring in the Mediterranean Basin. The direct effects of past and current land use changes include the replacement of native vegetation by human-made structures (buildings, roads, concrete lots, plazas) and exotic plant species (e.g., in monocultures, plantations, orchards, ornamental gardens, lawns, golf courses). Projected land use and climate changes also indicate that med-ecosystems will be among those most negatively affected by changes in future environmental conditions (Sala et al., 2000).

Table 1 Percent coverage by human land uses (urban, farm, and pasture lands) and average human population density for watersheds largely occurring in mediterranean-type climates

Region and rivers	Urban + farm + pasture	Population density (#/km ²)
Mediterranean Basin		
Iberian Peninsula (Sabater et al., 2009)		
Ter	34	108
Ebro	50	34
Júcar	52	207
Segura	55	78
Guadalquivir	63	69
Guadiana	69	24
Tagus	48	136
Mondego	37	96
Duero	57	37
Rhône River Basin (Olivier et al., 2009)		
Upper Rhône	19	190
Main Rhône	45	141
Ain	40	61
Saône	63	94
Isère	23	82
Durance	24	22
Balkan Rivers (Skoulikidis et al., 2009)		
Kamchia	48	48
Evros	61	69
Axios	44	87
Evrotas	35	30
Pinios	55	54
Italian Peninsula (Gumiero et al., 2009)		
Tagliamento	18	50
Po	49	224
Arno	57	243
Tiber	55	238
Turkey (Akbulut et al., 2009)		
Seyhan	32	92
Ceyhan	38	91
Gediz	15	113
Southern California (Brinkman, 2007; Hunt, 2008)		
Santa Maria	13	25
Santa Ynez	10	32
Ventura	15	76
Santa Clara	13	83
Los Angeles	62	2022
San Gabriel	55	1291

Table 1 continued

Region and rivers	Urban + farm + pasture	Population density (#/km ²)
Santa Ana	45	673
Small coastal streams in southern Santa Barbara County	0–51	0.1–464
Small coastal streams in Santa Monica Mtns., CA	1–25	0.4–262
Small streams along Santa Catalina Gulf Coast	5–30	12–440
Southwestern West Australia (Stein, 2006; Walsh et al., 2007)		
Swan	86	11
Murray	61	2
Collie	31	9
Blackwood	81	1
Warren	33	1
Western Cape Province, South Africa (C. A. P. E. Estuaries Management Program, 2008; Dept. of Environmental Affairs and Tourism, 2001; Dept. of Water Affairs and Forestry, 2004; River Health Programme, 2003, 2007)		
Berg	66	48
Diep	82	71
Hout Bay	36	947
Lourens	72	252
Overberg West WMA (Palmiet, Bot, Klein, Uilkraal)	44–49	36
Overberg East WMA (Heuningnes)	41	7
Breede (Bree)	34	16
Duiwenhoks	47	
Goukou	36	
Gourits	5	9
Swart	42	
Knysna	31	
Central Chile (Little et al., 2009; Stehr et al., 2010)		
Purapel in Nirivilo	63 ^b	
Cauquenes en El Arrayan	59 ^b	
Vergara	70 ^b	
V de Valparaiso ^a	15	107
Metropolitana de Santiago ^a	29	429
VI de O'Higgins ^a	8	54
VII del Maule ^a	29	33
VIII del Bio Bio ^a	22	55

^a Regional averages^b Primarily pine or *Eucalyptus* plantations

Different med-regions also have different land use challenges. In southern Europe, farming, grazing, and silvicultural operations, particularly at higher elevations, have been abandoned with the subsequent return of shrublands or forest (Sirami et al., 2010); however, increased mechanization has allowed the expansion of cereal cultivation in North Africa. In southern California, many agricultural areas have been converted to urban land uses. The area covered by vineyards also has grown rapidly in California, South Africa, Western Australia, and Spain, whereas increasing amounts of natural habitat has been converted to pine plantations and avocado orchards in Chile and olive orchards in Greece (Aronson et al., 1998; Little et al., 2009; Underwood et al., 2009). Although Chile and South Africa have rapidly growing cities, including Valparaíso, Santiago, and Cape Town, most water and land in these med-regions is used for agricultural activities. Most med-rivers are heavily managed with flows being regulated through dams and other engineering works to ensure constant water supplies in a climate with highly seasonal rainfall. The construction rate of large dams in Spain peaked during the 1960s and 1970s and in California during the 1950s and 1960s, with currently more than 1,000 dams in Spain and over 1,600 in California (Goslin, 2005).

The growth of the wildland–urban interface in many med-regions has increased the intensity and frequency of wildfires and promoted the introduction of exotic species (Verkaik et al., 2012). Land use changes and fires have similar short-term effects on the physical, chemical, and biological characteristics of streams, but with fires constituting a pulsed perturbation, with rapid ecosystem recovery (Verkaik et al., 2012), and land use changes constituting a press perturbation with sustained, long-term impacts on ecosystems. Frequent clearance, burning, or intense grazing of med-vegetation, however, can produce a vegetation type conversion from forest or shrubland to degraded scrub or grassland (Keeley, 2002; Van de Wouw et al., 2011), with persistent effects on both terrestrial and aquatic ecosystems. Other human effects on med-rivers include straightening and building levees along river channels, decoupling rivers from their floodplains and destroying riparian vegetation as floodplains are converted to grazing, farming, industrial, or urban uses. For example, riparian vegetation cover in the middle section of the River Ebro declined from 40% in the 1950s to 4.5% today

(Ollero, 2007). Although land use changes have diminished riparian vegetation in many med-ecosystems, canopy cover by riparian vegetation can be as high or higher in human-altered basins as in undeveloped basins because of the protection or restoration of riparian bufferstrips, planting of ornamental trees, and increased return flows of irrigation or sewage water in some catchments. In addition, the construction of dams on rivers reduces floods in downstream areas, which may allow the encroachment of riparian vegetation; however, the net effects of dams on downstream riparian vegetation depend on the effects of dams on seasonal river flows (i.e., increased vs. decreased dry season flows).

Effects of land use changes on mediterranean streams

Physical factors

Hydrology

Mediterranean ecosystems are defined by a common climate, which is reflected in the hydrological characteristics of their basins. Driven by seasonal rainfall patterns, flashy fall or winter floods are followed by a long dry season, leading to low or non-existent stream flows (Kennard et al.'s (2010) predictable winter intermittent pattern). The rivers and streams of central Chile and parts of the Mediterranean Basin and California also are fed by snow melt from high mountains, exhibiting a bimodal hydrograph reflecting runoff associated with fall or winter rains and spring snow melt (Gasith & Resh, 1999). As an example of the high seasonal variability in river discharge patterns, monthly averages for the Llobregat River in northeastern Spain range from <2 to $130 \text{ m}^3 \text{ s}^{-1}$ (Sabater & Tockner, 2010). If the fall-winter rains fail, even for just 1 year, supra-seasonal drought and low or non-existent flows can prevail (Lake, 2003, 2011).

During human development of mediterranean landscapes (med-landscapes), native vegetation is removed, soils are disturbed, impermeable surfaces are constructed, and open monocultures are cultivated, leading to increased, rapid runoff and flash floods during storms and decreased groundwater recharge, resulting in lower dry season flows or longer no-flow periods (Konrad & Booth, 2005; Walsh et al., 2005;

Marchi et al., 2010). Owing to extensive impervious land surfaces in urban areas, even minor storms may lead to increased runoff and small peak flow events in streams (Walsh et al., 2005). Ackerman & Schiff (2003) reported that increasing runoff coefficients (ratio of rainfall to runoff, normalized by drainage area) and a shift from broader to narrower discharge peaks reflected increases in impervious surfaces in urbanized sites in southern California.

Changes in runoff and groundwater pumping can diminish aquifers, exacerbating the effects of seasonal or longer term droughts on stream flows, particularly where wells are numerous (e.g., 1 million illegal wells have been reported in Spain) (Glennon, 2002; Sophocleous, 2002; Menció & Mas-Pla, 2008, 2010). Although human modifications of med-catchments generally intensify natural seasonal variation in discharge, these impacts can be modified by other human activities. Irrigation return flows or sewage treatment plant discharges during the dry season can augment river flows, in some cases maintaining flowing rivers which would normally dry (Prat & Munné, 2000; Burton et al., 2005; Riley et al., 2005; Klose et al., 2009). Irregular water flows (1–100 m³/s) in the Llobregat River of northeastern Spain are influenced by the 55 wastewater treatment plants (WWTPs) scattered through its drainage network, with WWTP discharge, at times, constituting most of the river flow.

Human-induced alterations in water yield in med-catchments are often associated with changes in vegetation and evapotranspiration rates. The replacement of native vegetation by exotic tree plantations can increase evapotranspiration rates and decrease runoff (Stipinovich, 2005), but the reverse is often observed when native forests or shrublands are replaced by degraded shrublands, crop lands, or grassland (García-Ruiz et al., 2011). The abandonment of upland agricultural lands in southern Europe has allowed natural vegetation to expand, which when combined with increased water consumption for domestic, industrial, and agricultural uses in lowland areas, has resulted in lower flows and longer dry periods for unregulated rivers (Benejam et al., 2008).

As in most arid and semi-arid regions, rivers have been extensively modified to provide supplies of water during times of need. The largest use of water in med and most other climatic regions is for agricultural purposes, particularly for the irrigation of farmlands.

For example, up to 80% of surface and ground water in the North African Maghreb is used for agriculture and irrigated areas are rapidly expanding (e.g., irrigated area increased from 65,000 ha in 1956 to 345,000 ha today in Tunisia; Djellouli-Tabet, 2010). Intensive use of water supplies for agricultural, urban, and industrial activities amount to 30–220% of the total resource, stimulating the construction of dams, delivery canals, wells, and facilities for inter-basin transfers (Sabater, 2008; Sabo et al., 2010a, b). Annual discharge in Balkan Rivers has declined by 48–79% (Skoulikidis et al., 2009) and discharge in the Ebro River of northern Spain has decreased by nearly 40% over the last 40–50 years (López-Moreno et al., 2011). The largest impacts on river flow are engendered by the operation of dams, which affect the timing and magnitude of flow regimes (López-Moreno et al., 2011). In the Mediterranean Basin, the largest number of dams is found in Spain and Turkey, and they have large effects on river discharge patterns in downstream areas, reducing floods, increasing low-flow periods, and changing flow seasonality (i.e., high flows for irrigation in summer, low flows in winter when runoff is stored, Belmar et al., 2010; Sabater & Tockner, 2010). Water from two large reservoirs on the headwaters of the Tagus River of Spain is now transported out of the basin, resulting in consistent low river flow below the dams which is further reduced during natural droughts (Lorenzo-Lacruz et al., 2010). In these regions of Spain, water demand has been driven by the conversion from dryland to irrigated farming.

To illustrate natural hydrographs for med-rivers and to examine the effects of land use conversions and dam construction on them, we applied spectral methods (Sabo & Post, 2008; Sabo et al., 2009; Grossman & Sabo, 2010; Sabo et al., 2010a, b; Sabo et al., 2012) to daily average discharge data for med-rivers on four continents: the Warren River in Western Australia, the Doring River in southwestern South Africa, the Salinas River on the California (USA) coast, and the Ebro River in eastern Spain. Discharge data were obtained from the Global Runoff Data Centre (http://www.bafg.de/GRDC/EN/Home/homepage__node.html) with the exception of data for the Salinas River, which were derived from the National Water Information Service (maintained by the United States Geological Survey). These rivers, which vary in drainage area by two orders of magnitude (Table 2) have long-term

Table 2 Spectral metrics of hydrologic variation and changes in these metrics between early and late periods of the record for four mediterranean rivers (drainage areas in parentheses in km²)

Metric	Description	Warren (3,933)		Doring (6,895)		Salinas (10,639)		Ebro (84,230)	
		1967–1981	1991–2005	1924–1932	1998–2010	1930–1949	1991–2010	1917–1930	1987–2000
All values in first section of table are for the time period indicated above, no referencing to historical record to assess changes in hydrology									
A_{rms}	Root mean squared amplitude	0.94	0.72	1.03	1.14	0.84	0.72	0.08	0.10
N_{rms}	Root mean squared noise	0.01	0.01	0.02	0.02	0.02	0.02	0.00	0.00
SNR	Signal to noise ratio	40.84	38.20	33.34	35.86	34.53	29.35	30.85	29.72
σ_{hf}	Standard deviation of high-flow anomalies	0.52	0.58	1.97	1.47	1.49	1.55	0.14	0.17
σ_{lf}	Standard deviation of low-flow anomalies	−0.33	−0.39	−0.58	−0.60	−0.99	−1.57	−0.12	−0.35
$2\sigma_{hf}$	Magnitude equivalent to 2 SD high-flow anomalies	1.04	1.16	3.94	2.94	2.98	3.10	0.28	0.34
$2\sigma_{lf}$	Magnitude equivalent to 2 SD low-flow anomalies	−0.67	−0.79	−1.17	−1.20	−1.98	−3.14	−0.25	−0.70
All italicized values in the following section reflect statistical properties of events referenced to historical hydrographs. First column for each river is self-referenced events to trend from the early period. Second column for each river is recent events referenced to trend from the historical record									
Frequency of high-flow events	Number of high-flow anomalies exceeding σ_{hf}	26.00	37.00	21.00	7.00	14.00	13.00	53.00	27.00
Frequency of low-flow events	Number of low-flow anomalies less than σ_{lf}	41.00	37.00	24.00	28.00	16.00	39.00	24.00	46.00
Timing of high-flow events	Modal month of high-flow anomalies exceeding σ_{hf}	3.00 ^a	4.00 ^a	4.00 ^a	1.00 ^a	5.00 ^b	4.00 ^b	2.00 ^b	4.00 ^b
Timing of low-flow events	Modal month of low-flow anomalies less than σ_{lf}	1.00 ^a	2.00 ^a	2.00 ^a	3.00 ^a	5.00 ^b	3.00 ^b	2.00 ^b	9.00 ^b
Duration of high-flow events	Maximum duration (day) of high-flow anomalies exceeding σ_{hf}	183.00	142.00	82.00	46.00	349.00	259.00	87.00	98.00
Duration of low-flow events	Maximum duration (day) of low-flow anomalies less than σ_{lf}	199.00	274.00	115.00	231.00	411.00	321.00	91.00	294.00

^a Water year for southern hemisphere rivers starts April 1; April = 1, March (following calendar year) = 12; water years used only for comparison of timing in this table (not in Fig. 1)

^b Water year for northern hemisphere rivers starts October 1; October = 1, September (following calendar year) = 1

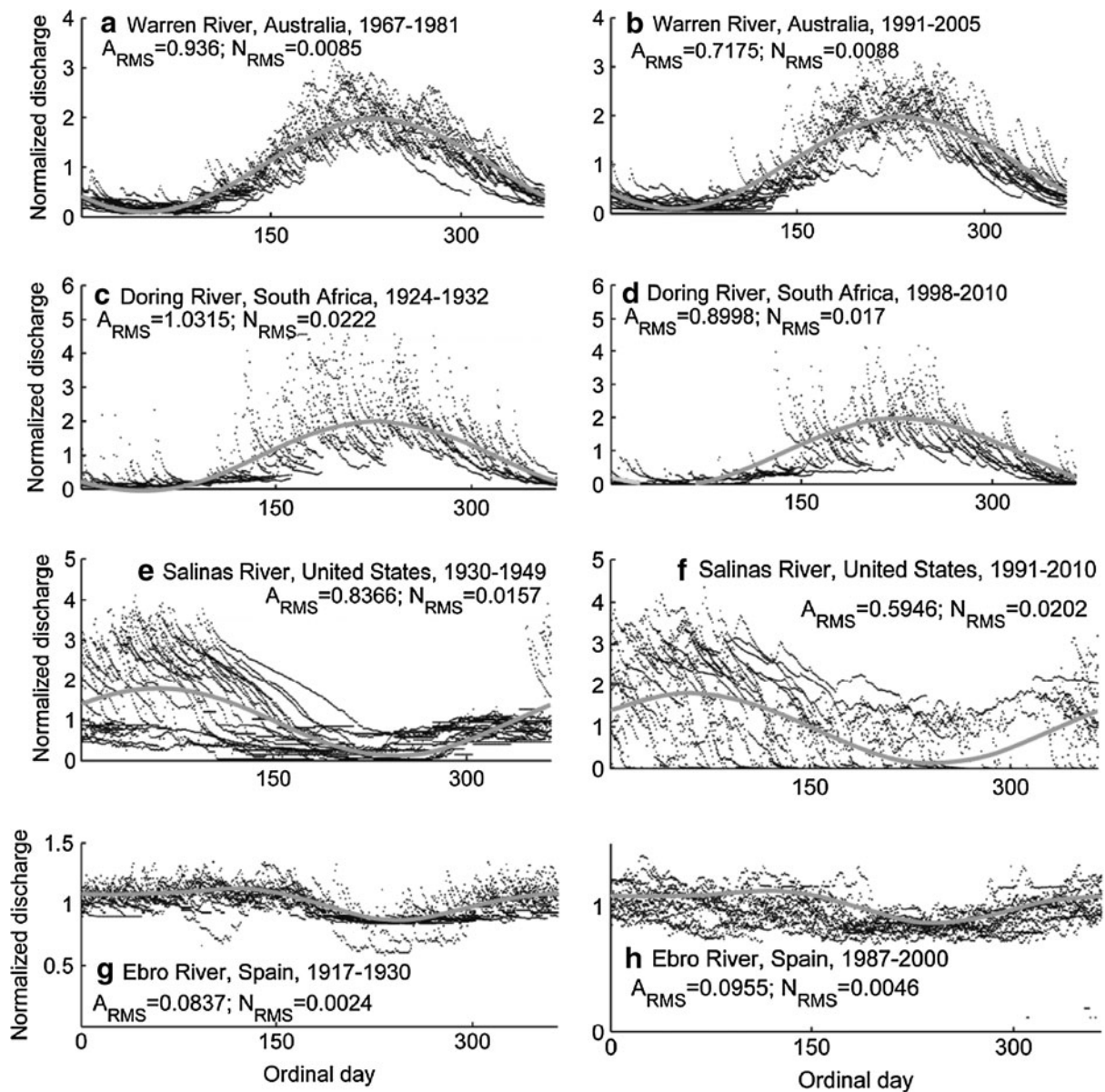


Fig. 1 Hydrographs for med-rivers on four continents. All panels show daily average discharge (\log_{10} transformed), normalized by average daily discharge over the same period of record (black circles). Dark gray lines are the reconstructed

signal extracted from the Fourier power spectrum. The amplitude of this signal is given by A_{RMS} , and the root mean squared noise around this signal is N_{RMS} (Sabo & Post, 2008)

discharge datasets, and drain basins with contrasting land uses, with the Warren River being the least affected and the Salinas and Ebro being the most impacted by dams, agriculture, and, in the Ebro basin, major population centers. Although we contrast river hydrographs before and after major dam construction, it should be remembered that the Ebro River basin was

greatly influenced by human activity before these records began.

In contrasting variation in these four med-river systems between early and recent periods in the discharge record (Fig. 1), we first note the strong seasonality (indexed by the amplitude of seasonal variation, A_{RMS}) and even stronger interannual

variation associated with the winter rains (indexed by N_{RMS} , Fig. 1, and σ_{hf} and σ_{lf} ; Table 2). Changes in hydrograph characteristics between early and late record periods are most obvious for the Salinas and Ebro Rivers, where numerous dams and extensive agricultural development have resulted in highly managed systems (Fig. 1). Nevertheless, there are detectable changes in early and late record periods for all four rivers, as quantified by changes in the standard deviation of high- and low-flow anomalies (σ_{hf} and σ_{lf}) and the number, timing (mode), and duration of high- and low-flow anomalies exceeding a spectral anomaly magnitude (SAM) equivalent to σ_{hf} and σ_{lf} . SAMs for high flows (σ_{hf}) increased in three of the four rivers whereas the (absolute) magnitude of low-flow anomalies (σ_{lf}) increased across all four rivers between early and late periods indicating stronger low-flow variation (Table 2). Finally, we quantified the frequency, timing, and duration of low- and high-flow anomalies in the early and late-period records, *but* referenced to the early record reference SAM (σ_{hf} and σ_{lf}), so that we could determine if aseasonal extremes had changed relative to historical patterns of variation. The number (frequency) and duration of high-flow anomalies ($>\sigma_{\text{hf}}$) tended to decline and the timing of these events was shifted by 1–3 months. By contrast, the number (frequency) and duration of low-flow anomalies ($<\sigma_{\text{lf}}$) tended to increase and the timing of these events was shifted by 1–7 months with the greatest shift in timing for the Ebro River in which the historical low flows of November were replaced by low flows in June (a shift of 7 months later in the water year for this river; Table 2). Although we cannot rule out the effects of climate change on hydrographic changes, alterations in land use and in the water cycle are likely to be strong contributors to the observed changes, especially for the Salinas and Ebro rivers. Among the most noteworthy changes are alterations in the timing and increases in the frequency and duration of low-flow anomalies.

Geomorphology and sediment

Human impacts on vegetation and soils in med-regions can have large effects on the erosion, transport, and deposition of sediment and, hence, stream geomorphology. For mediterranean forests, floods tend to occur late in winter after water tables have risen, are relatively muted, and have low sediment

loads (Poesen & Hooke, 1997; Garcia-Ruiz et al., 2008). In comparison, runoff from cleared land, whether used or abandoned farmland, causes increases in flood frequency and intensity, and in suspended sediment and bed loads (Poesen & Hooke, 1997; Garcia-Ruiz et al., 2008). In extreme cases, excessive sediment inputs to channels can choke streams with sediment, resulting in shallower depths and less differentiation between pools and riffles. The representation of sand and fine substrata on stream bottoms, with consequences for substratum embeddedness, commonly increases with increasing catchment development (Ode et al., 2010). Human-created reservoirs often fill with sediment, reducing sediment transport to downstream areas (Kondolf, 1997; Minear & Kondolf, 2009). Fire in med-regions can also increase catchment runoff, flood frequency and severity, and erosion and sediment deposition with rapid amelioration as terrestrial vegetation recovers (Candela et al., 2005; Mayor et al., 2007; Shakesby, 2011). The effects of fire management activities, such as fire suppression or fuel removal via mechanical means or controlled burns, can also influence runoff and erosion (Verkaik et al., 2012).

Flash floods in urbanized med-catchments can cause major erosion of stream channels, scouring of stream beds, and the transport of high loads of suspended solids (Booth, 1990; Poesen & Hooke, 1997; Cherifi & Loudiki, 1999; Chu et al., 2011). Straightening river channels and lining them with rip-rap, concrete, or other impervious materials causes the rapid transport of flood flows, inhibits groundwater recharge through the channel bottom, and reduces substrata, bed, and, hence, habitat heterogeneity (Burton et al., 2005; Brown et al., 2005a, b). In cases where channels with natural substrata receive runoff from areas covered by impervious surfaces, sediment inputs may be low and stream beds may be excavated by flood flows, resulting in incision and failed banks (Booth, 1990). Channels are often lined with levees or are designed with excess capacity to prevent flooding of adjacent areas, which allows development in historical floodplain areas, but which disconnects rivers and streams from their riparian corridors and floodplains.

Temperature and light

Land use changes in med-regions can affect temperature and light levels in streams, owing to modifications of both the hydrological cycle and, particularly,

the riparian vegetation (Burton et al., 2005). Land use changes tend to increase overland flow during storms and decrease hyporheic exchange and groundwater inputs to streams, resulting in warmer stream temperatures (Dallas & Rivers-Moore, 2011). By reducing dry season flows, land use changes often result in isolated, non-flowing pools, whose temperatures reflect ambient atmospheric temperatures. Dams affect thermal regimes, often creating thermally stratified reservoirs upstream and increasing or decreasing temperatures downstream seasonally, depending on whether reservoir releases come from the epilimnion or hypolimnion.

Increases in temperature and light levels occur in most streams where the riparian canopy has been reduced or eliminated (Naiman & Décamps, 1997; Rutherford et al., 2004; Burton et al., 2005; Mazor et al., 2011). Although reductions in riparian vegetation have been observed in many med-rivers and streams, such changes have not been observed in others, apparently depending on whether riparian corridors are protected or restored, exotic plants have been planted along streams, or water supplies to relatively intact riparian zones have been augmented by return flows. Working in southern California, Rehn (2010) recorded similar riparian canopy coverages along natural and degraded streams, but Mazor et al. (2011) recorded lower average canopy vegetation in urban than open or agricultural streams. Burton et al. (2005) reported greater bank shading in natural (87%) streams than concrete-lined channels (9%) and lower temperatures in natural streams than in channelized streams receiving urban runoff or wastewater. Although in-stream light levels often increase with reductions in riparian vegetation, this effect can be overridden by the effects of increased suspended sediment transport from landscape or channel erosion.

Hydrochemistry

Pollutants

Changes in land use are often associated with inputs of nutrients, metals, pesticides, herbicides, pharmaceuticals, oil products, plastics, and personal care products (PCPs), although loading rates depend on the presence of WWTPs and the specific land uses and lithology of catchments (Turgut, 2003; Farmaki & Thomaidis, 2008; Muñoz et al., 2009; Sánchez-Pérez et al., 2009;

Fernández et al., 2010; Ginebreda et al., 2010). Pesticides, pharmaceutical and PCPs, surfactants, flame retardants, and other groups of chemicals can attain concentrations from several ng to a few µg/l, particularly just below WWTPs and in downstream river sections (Mazor & Schiff, 2008; Petrovic et al., 2011). Metals can attain concentrations of several µg/l, with soluble forms most concentrated during low flows (Guasch et al., 2010; Mazor et al., 2011). Among 8 metals monitored during the dry season in 11 southern California catchments, Mazor & Schiff (2008) reported concentrations for most of between a few and 10 µg/l, but with higher average concentrations for dissolved manganese (41–238 µg/l) and selenium (7–78 µg/l). Although PCBs were not detected in this study, concentrations of a PAH (benzo(b)fluoranthene) reached nearly 9 ng/l at one site and specific pesticides had concentrations as high as ca. 20 ng/l.

Floods in Mediterranean rivers and streams can erode, transport, and suspend sediments and associated (hydrophobic) pollutants, such as sediment-associated metals, which reach highest concentrations during floods (Guasch et al., 2010). Ackerman & Schiff (2003) examined stream concentrations and fluxes of nutrients (ammonium, nitrate, phosphate), metals (cadmium, chromium, copper, lead, mercury, nickel, selenium, zinc), suspended solids, and pesticides (chlorpyrifos, DDT, diazinon) across a land use gradient (undeveloped open space, agriculture, residential, mixed urban, commercial, and industrial use) in southern California (Table 3). Agricultural storm water had the highest concentrations for all constituents (by factors of 1.1–81) except for total phosphorus and diazinon, which were slightly higher in streams in residential areas, and mercury, which was not detected in agricultural runoff. The geometric mean concentrations for all constituents, except mercury and nickel, were lower in streams draining undeveloped versus developed catchments. Concentrations of all chemical variables were similar and elevated across residential, commercial, industrial, and mixed urban areas (=urban areas), with streams in highly urbanized areas exporting 45–56% of the nutrients, 60% of the copper and zinc, and 67% of the chlorpyrifos and diazinon in this region. Such pollutants in rivers with low discharge and dilution rates, owing to abstraction or natural hydrological variation, can interact with other stressors to produce complex and rather

Table 3 Comparison of fluxes (expressed as kg km⁻²) in storm water for water year 1995 of several important constituents across a land use gradient (left to right: least to most

developed) (subset of data from Table 7 in Ackerman & Schiff, 2003, rounded to whole numbers except metals)

Constituent	Open space	Agriculture	Residential	Mixed urban	Commercial	Industrial
Ammonium	2	50	57	66	94	75
Nitrate	51	271	219	234	275	287
Phosphate	14	21	76	77	103	83
Suspended sediment	717	56,400	7,340	10,600	11,900	18,800
Copper	0.1	5.6	2.2	3.5	4.4	6.3
Zinc	0.1	8.3	9.3	20.4	33.6	43.5

Open space is undeveloped land

unpredictable effects on river communities (Stevenson & Sabater, 2010).

Salinity

A common effect of land use changes on streams is to increase the concentration of major ions, as reflected in conductivity (Burton et al., 2005; Boyacioglu, 2006; Brinkman, 2007; Ode et al., 2010; Mazor et al., 2011). Streams and rivers often exhibit increased salinity levels with increasing catchment development, particularly in areas with salty soils and groundwater (McFarlane & George, 1992; Halse et al., 2003). Massive land clearing in southwestern Western Australia has led to the replacement of deep-rooted native vegetation by shallow-rooted crops, mainly wheat, with saline groundwater tables becoming shallower (Halse et al., 2003; Morgan et al., 2003; Caccetta et al., 2010; Furby et al., 2010). In this area, damaging salt levels have sometimes reached the soil surface and the area of dryland salinity has increased annually by 14,000 ha. Rivers and lakes in drier parts of this region have become more saline because of increased saline groundwater inputs or storm flows over and through saline soils (Schofield & Ruprecht, 1989; Ghassemi et al., 1995; Halse et al., 2003). Salinity levels in affected streams increase during the first flushing flows of the winter rainy season reach lowest levels in late winter with high flows, and increase again as flows diminish in the dry season, often reaching high levels in persistent pools in heavily cleared catchments (Williams et al., 1991; Kay et al., 2001; Halse et al., 2003; Davis et al., 2010). Increased salinization of rivers and streams has been observed in other med-

regions, including southeastern Spain and southwestern South Africa (Millán et al., 2011).

Nutrients

Stream and river concentrations of nitrogen and phosphorous often increase with increasing human development in med-catchments (Burton et al., 2005; Busse et al., 2006; Brinkman, 2007; Klose et al., 2009; Song et al., 2009; Ode et al., 2010; Mazor et al., 2011; Fig. 2). These nutrients originate from increased erosion of denuded landscapes; fertilizer and manure runoff from agricultural lands, gardens, and golf courses; and sewage effluent. Nutrient peaks often coincide with or follow storm discharge peaks (e.g., Klose et al., 2009; David et al., 2011).

Robinson et al. (2005), working in the Santa Barbara area of southern California, found marked differences in nutrient export during storms across different land uses, approximating 1, 10, and 100 g nitrate-N ha⁻¹ mm⁻¹ and 0.8, 2–9, and 20–70 g phosphate-P ha⁻¹ mm⁻¹ for undeveloped uplands, residential/commercial areas, and greenhouse agricultural lands, respectively. Storm to storm variability was low. In a year with about 70 cm of rain, Robinson (2006) reported fluxes of nitrate (mol ha⁻¹ y⁻¹) of 21 from upland chaparral, 49 from residential areas, and 5,684 from greenhouse agriculture. In chaparral catchments in the San Bernardino Mountains of southern California, Riggan et al. (1985) reported nitrate export ranging from 3 to 714 mol ha⁻¹ y⁻¹ over 4 years and Fenn and Poth (1999) measured nitrate export of 257–829 mol ha⁻¹ y⁻¹ from 1995 to 1998. In three studies in rural areas with extensive tree

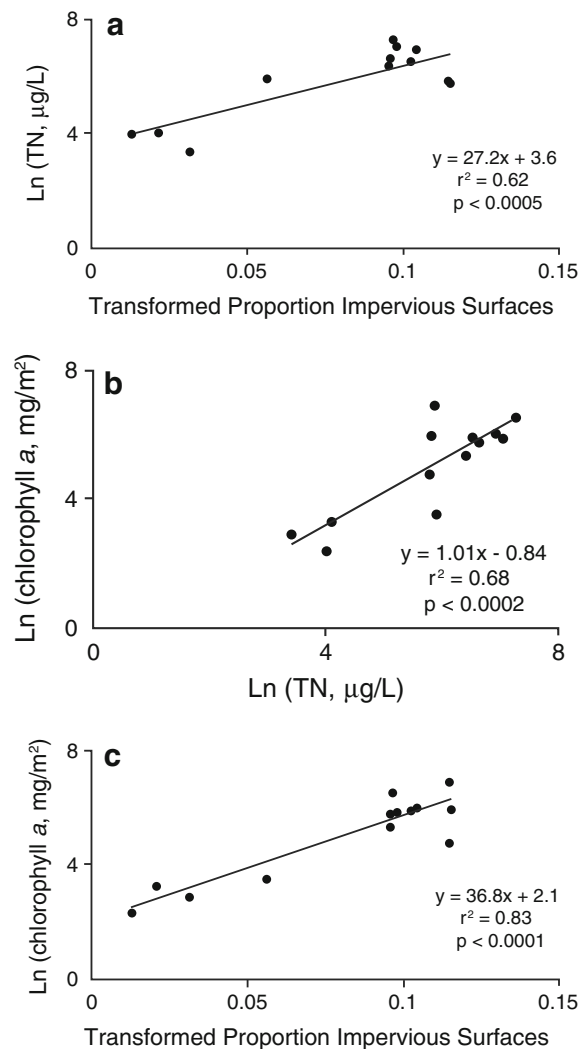


Fig. 2 Illustration of relationships among land use, nutrient concentrations, and benthic algal biomass (as chlorophyll *a* concentration) for a med-river system. The data are from 14 sites in the Ventura River catchment, southern California, USA, sampled in June 2008. Nutrient and chlorophyll data were log_e-transformed and proportion land use cover was arcsine square root transformed before analyses. Equations, best-fit lines, and overall r^2 and P values from least-squares linear regression analyses are shown on each panel. (Plotted from data in Klose et al., 2009, 2012)

cover and limited urban development in Catalonia, Spain, nitrate export was calculated to be $3.6 \text{ mol ha}^{-1} \text{ y}^{-1}$ (Avila et al., 2002), $47.1 \text{ mol ha}^{-1} \text{ y}^{-1}$ (Butturini & Sabater, 2002), and $15.7 \text{ mol ha}^{-1} \text{ y}^{-1}$, with 52–86% of the annual nitrate export occurring during storms in the latter case (Bernal et al., 2002).

Goodridge & Melack (2012), working in the Santa Barbara (California) area, found that nitrate

concentrations changed with increasing discharge, decreasing in three agricultural catchments, increasing in the undeveloped catchment and an urban catchment, and remaining constant in another urban catchment. In contrast to the nitrate-runoff relationships, electrical conductance, a measure of total dissolved solids, decreased with increasing discharge in all catchments. Based on the hydrological modeling results of Beighley et al. (2008) and their own analyses, Goodridge & Melack (2012) argued that upland catchment areas with no development played an important role in regulating stream nitrate concentrations, resulting in little variation in peak storm nitrate concentrations across catchments and with nitrate concentrations in lower river reaches integrating upland catchment hydrology and nitrogen processing and nitrogen subsidies from lowland areas (also see Melack & Leydecker, 2005). Upland–lowland linkages became weaker in dry years and stronger in wet years. These patterns are likely to be common where headwater streams drain steep, undisturbed montane catchments, flowing into lowland areas with extensive agricultural and urban development. In some cases, such as in the Ventura River of southern California, storm hydrographs near the mouth are often bimodal, with an early peak associated with runoff from lowland developed areas and a later, higher, and broader peak associated with runoff from extensive, undisturbed, upland areas, with associated changes in material concentrations and fluxes (Klose et al., 2009).

Besides nutrients in runoff from developed landscapes, wastewaters are a major source of nitrogen and phosphorus in streams and rivers, particularly in urbanized catchments (Ortiz et al., 2008; Martí et al., 2010). During seasonal or longer term droughts WWTP discharge can be the major or only contributor to flow and nutrient flux in urbanized med-rivers, accounting for more than 50% of nitrogen and phosphorus loads (Carey & Migliaccio, 2009; Klose et al., 2009).

Nutrients may be transported to downstream floodplains or into estuaries and, thence, into the ocean (McComb & Davis, 1993; Obermann et al., 2009; Chu et al., 2011). Many estuaries in med-regions are poorly flushed during the dry season, becoming isolated from tidal fluxes and accumulating nutrients from the catchment, fueling eutrophication (Hodgkin & Hamilton, 1993; McComb & Davis, 1993).

Organic matter

Dissolved and particulate organic matter enters streams from upland and riparian zones, providing a carbon source for stream food webs. Land use changes affect organic matter inputs to streams by altering the sources, routing, and processing of terrestrial and aquatic carbon. For example, land use changes often change the abundance and composition of riparian vegetation (Fig. 3a), leading to alterations in the composition and levels of organic carbon in streams with effects on microbial and invertebrate communities (see below).

An extensive literature documents the impacts of organic carbon inputs in sewage on stream organisms (Cairns & Pratt, 1993; Ortiz et al., 2008). Depending on the comprehensiveness and sophistication of sewage and storm water treatment facilities and the volume of river flows, sewage and storm runoff inputs can produce high organic carbon inputs, ammonia levels, and microbial activity and low dissolved oxygen concentrations, sometimes eliminating most macroinvertebrate and fish species (Prat & Munné, 2000; Karaer & Kucukballi, 2006; Ortiz et al., 2008). Furthermore, inputs of nutrients (N, P) to mediterranean streams (med-streams) can increase the decomposition rates of particulate organic carbon and produce algal blooms which, in turn, can stimulate bacterial growth on algal exudates and detritus (Prat & Munné, 2000). In some cases, organic and nutrient inputs stimulate microbial and algal growth in open med-rivers reducing oxygen to low levels, particularly at night, sometimes causing fish kills (Klose et al., 2009, 2012). Similar considerations apply when med-streams have very low flows or become a series of isolated, stagnant pools during the summer dry season, when decomposition of allochthonous leaf litter produces high dissolved organic carbon (primarily polyphenol) levels, stained water, and low or no dissolved oxygen (Towns, 1985, 1991; Boulton & Suter, 1986; Bunn et al., 1986).

Biological communities

As is apparent from the above discussion, land use changes have numerous effects on physical–chemical conditions in streams and rivers, with many ramifications for biological communities. Many studies, including those from med-ecosystems, have shown

that land use changes affect the growth, development, reproduction, and behavior of individual organisms and the diversity, composition, abundance, and biomass of biological assemblages.

Riparian zones, allochthonous inputs, microbial communities, and decomposition

Land use changes have had large effects on the structure, cover, patchiness, and composition of upland and riparian vegetation in med-regions with subsequent effects on the wildlife it supports (Fig. 3a, b; Warner & Hendrix, 1984; Décamps et al., 1987, 1988; Le Maitre et al., 1996; Corbacho et al., 2003; Brown, 2004; Aguiar & Ferreira, 2005; Ferreira et al., 2005; Salinas & Casas, 2007; Hughes et al., 2010; Larsen et al., 2010; Fernandes et al., 2011). One general pattern is that the total number and relative proportion of exotic plant species increase with catchment development (Aguiar & Ferreira, 2005; Aguiar et al., 2006; Walker et al., 2009; Fig. 3a). In some cases, exotic plant genera, such as *Acacia* Mill. in South Africa and Chile, *Eucalyptus* L'Hér. in California and Europe, and *Arundo* L. in the riparian zones of South Africa and California, come to dominate plant communities (Le Maitre et al., 2011). Where land use development has reduced riparian vegetation cover, streams have higher temperatures and light levels, lower inputs of allochthonous carbon, and greater bank erosion; however, the effects of land use changes on riparian canopy cover and leaf litter inputs also will depend on riparian bufferstrip protections and possible compensation through planting efforts (Naiman & Décamps, 1997). In some cases, such as where there are dense growths of *Acacia* along streams in the Cape of South Africa, exotic riparian species may reduce light and temperature levels and confine or impede flows with repercussions for stream communities (D. W. A. F., 2004; C. A. P. E., 2008; Le Maitre et al., 2011; Samways et al., 2011).

Land use changes also can affect riparian vegetation by increasing nutrient and sediment inputs, and altering flow regimes, soil moisture levels, groundwater tables, and disturbance patterns (floods, drought) (Warner & Hendrix, 1984; Aguiar & Ferreira, 2005; Salinas & Casas, 2007). In some med-regions, increases in soil and stream salinity associated with land use changes may affect riparian vegetation. In southwestern Western Australia, the species richness,

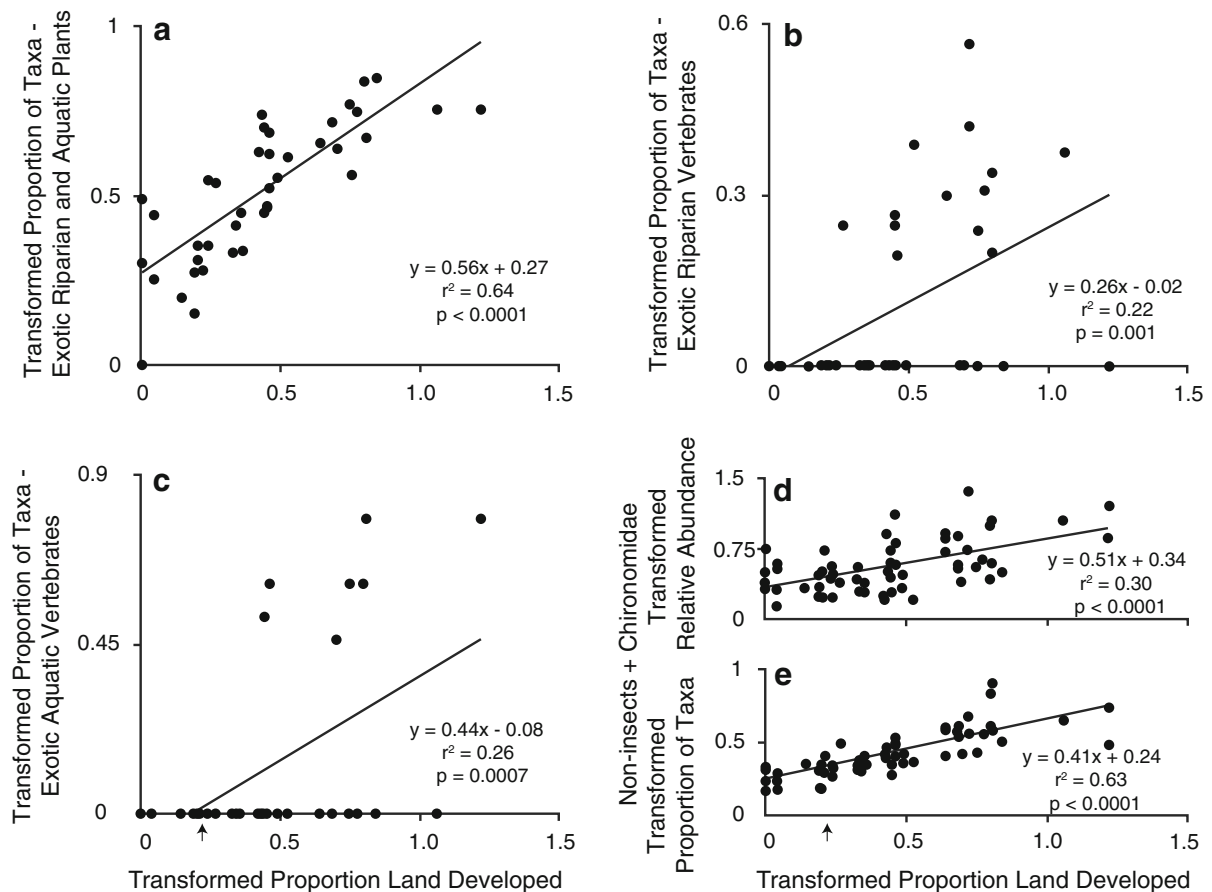


Fig. 3 Illustration of relationships between the proportions of (a) riparian and aquatic plant species, (b) aquatic vertebrate (fish, amphibian, aquatic reptile) species, and (c) riparian vertebrate (bird, mammal) species which were exotic and the proportion of catchments covered by developed lands (agriculture + urban + pasture) for a med-region. Plates d and e show the relationships between the proportions of all invertebrate individuals (d) and invertebrate taxa (e) which belonged to non-insect groups and the Chironomidae versus the proportion of catchments in developed land uses. Proportionate data were arcsine square root-transformed and equations, best-fit lines, and overall r^2 and P values from least-squares linear regression

but not functional diversity, of riparian plants, particularly perennial herbs, declined and species composition was altered by increasing soil salinity (Lymbery et al., 2003; Doupé et al., 2006). Studying riparian corridors in two rivers in southern Spain, Salinas & Casas (2007) found that the species richness of herbaceous and woody plants, and cover by woody species, declined downstream with increasing salinity.

The inputs and decomposition rates of litter, and composition and biomass of associated microbial

analyses are shown on each panel. Re-analysis of data from Brinkman (2007) (38–41 stream sites sampled over 1–3 springs (2000–2002) in the Santa Barbara, California, area). Variation in the relationships between proportions of exotic vertebrate species and land use patterns was high because of low numbers of exotic species in these groups (riparian vertebrates = 4 exotic and 81 native species, aquatic vertebrates = 2 exotic and 8 native species); however, exotic species were observed only at stream sites in developed catchments. Arrows The demarcation between undeveloped (>95% wildlands, 11–12 sites) and developed catchments (<95% wildlands, 27–29 sites)

communities, are affected by land use changes (González et al., 2010). Working in agricultural catchments with coastal urban areas in central Portugal, Feio et al. (2010) reported that the decomposition rates of oak (*Quercus robur* L.) and alder (*Alnus glutinosa* Gaertn.) leaves in coarse (1 cm) mesh bags, but not in fine (0.5 mm) mesh bags, were lower in streams affected by human activity than in relatively undisturbed, reference streams, apparently because of reduced densities of macroinvertebrate shredders in the

disturbed streams. Leaf decomposition rates were negatively related to ammonium and nitrite levels and alterations of the hydrological regime, and positively correlated with nitrate and dissolved oxygen concentrations. Menéndez et al. (2011) found that alder leaf breakdown rates increased with increasing nutrient inputs and shredder densities across five low-order Spanish streams, and also reported that fungal sporulation rates were positively related to nitrate but negatively related to ammonia concentrations. Field studies and laboratory experiments indicate that leaf decomposition and fungal biomass, community composition, and/or sporulation rates are affected by temperature and inorganic nutrient concentrations, i.e., factors which are often increased by human development (Gulis et al., 2006; Artigas et al., 2008; Ferreira & Chauvet, 2011a, b). In addition, land use changes often are associated with decreased or disrupted flows which can affect fungal biomass and sporulation rates and leaf decomposition rates (Bruder et al., 2011), but flow resumption stimulates bacterial and fungal growth (Artigas et al., 2009, 2011). By changing bottom sediment size distributions, human land use also affects the decomposition rates of buried leaf litter by affecting the access of invertebrate shredders to interstitial space (Navel et al., 2010).

Land use changes can alter leaf decomposition rates and the growth, species composition, and biomass of microbial communities by altering the composition of riparian vegetation. Although leaves from exotic plant species sometimes have slower decomposition rates than leaves from native species (e.g., Hart, 1975), differences in breakdown rates among leaf species appear to be affected more by leaf properties, such as structural or secondary compounds or nitrogen content, than by their geographic origin (Canhoto & Graça, 1996; Akanil & Middleton, 1997; Graça et al., 2001; Sampaio et al., 2001; Moline & Poff, 2008). Nevertheless, stream microbial and invertebrate communities can be altered by changes in riparian plant composition in developed catchments (Samways et al., 2011). Bärlocher & Graça (2002), for example, found that the richness and composition of fungal communities differed between streams bordered by native riparian forests and those bordered by pure *Eucalyptus* stands, but that decomposition rates of both *Eucalyptus* and native chestnut (*Castanea sativa* Mill.) leaves did not differ between the two stream types.

Feio et al. (2010) reported that sediment respiration rates were lower in undisturbed streams than in disturbed streams with higher organic matter, nutrient, and daytime dissolved oxygen levels, but they found no differences in fungal biomass on leaves or biofilm biomass (measured as ash free dry mass) on natural substrata in disturbed and undisturbed streams. Streams and rivers receiving high organic matter and/or nutrient loads have high bacterial biomass, sometimes becoming dominated by specific bacterial species (e.g., *Sphaerotilus natans* Kutzinger below sewage outflows) (Karrasch et al., 2006; Ortiz et al., 2008; Klose et al., 2012).

Much research on bacterial communities in med-regions has focused on indicators of fecal contamination (Cao et al., 2011a, 2011b). Fecal indicator bacterial concentrations increase with increasing urban development and usually peak during and after storms, leading to health advisories. Using rapidly developing molecular methods (Hagedorn et al., 2011), Sercu et al. (2011) observed that bacterial community composition in a southern California stream shifted from the wet to the dry season, with fecal indicator bacteria apparently derived from catchment runoff during rains but from point sources of human waste in the dry season.

Algal and aquatic plant biomass and communities

By affecting levels of primary resources, such as nutrients and light, land use changes have large effects on algal biomass and species composition (Burton et al., 2005; Busse et al., 2006; Simpson, 2006; Tornés et al., 2007; Delgado et al., 2012). Chlorophyll *a* values from med-rivers and streams range from 0.2 to 1,400 mg/m² and increase with catchment development (Table 4; Fig. 2; Busse et al., 2006; Klose et al., 2009, 2012; Ode et al., 2010; Delgado et al., 2012). Nutrient concentrations often determine algal biomass in med-rivers, although light can play a major role in small, headwater, or turbid streams (Busse et al., 2006; von Schiller et al., 2007; Klose et al., 2012). The addition of nutrients, however, can trigger increases in algal biomass even in shaded streams. For example, Sabater et al. (2011) reported that small windows of higher light availability were sufficient to produce a threefold increase in algal biomass in a forested stream following slight increases in nutrient concentration. Although small forested med-streams

Table 4 Chlorophyll *a* values in mediterranean streams and rivers

System	Chlorophyll conc. (mg m ⁻²)	Source
Mountain stream	6–11	Rolland et al. (1997)
Travertine streams	50–300	Rundio (2009)
Calcareous, non-travertine streams	100–500	Rundio (2009)
Middle mountain stream, low nutrient, shaded	5	Tornés & Sabater (2010)
Middle mountain stream, low nutrient, unshaded	4–7	Tornés & Sabater (2010)
Middle mountain stream, unfertilized, shaded	8–90	Sabater et al. (2011)
Middle mountain stream, fertilized, shaded	10–120	Sabater et al. (2011)
Undeveloped and rural streams	2–30	Busse et al. (2006)
Suburban streams	50–320	Busse et al. (2006)
Large river, agricultural	0.2–1,425	Urrea-Clos (2010)
Large river, agricultural	14–780	Sabater et al. (2009)
Agricultural streams	60–500	Hornberger et al. (1977)
Undeveloped, low-order streams	4–58	Klose et al. (2012)
River, residential and agricultural	90–417	Klose et al. (2012)
River, mixed use, below WWTP	226–1,037	Klose et al. (2012)
Temporary streams	4–112	Alvarez & Pardo (2007a, b)
Streams draining forested, agricultural, and urban basins	2–79	Von Schiller et al. (2008)

Data are the range observed in each system

have been postulated to be heterotrophic (Acuña et al., 2004), shifts toward increased autotrophy may occur when nutrients exceed threshold concentrations (Sabater et al., 2011). Additions of nutrients to open med-streams often promote the growth of filamentous macroalgae, such as *Cladophora* Kützinger, *Rhizoclonium* Kützinger, and *Enteromorpha* Link, sometimes resulting in dense floating mats. Diatoms also proliferate with increased nutrient loading in more shaded streams (Busse et al., 2006; Klose et al., 2012). Because different diatom species show different tolerances to inputs of organic matter, nutrients, metals, and biocides, diatom community composition is a sensitive indicator of water quality in med-streams and rivers (Tornés et al., 2007; Blanco et al., 2008; Blanco & Becarés, 2010; Martín et al., 2010; Ricart et al., 2010; Solak & Acs, 2011; Delgado et al., 2012; Kargioglu et al., 2012). In the Santa Ana River system in southern California, Burton et al. (2005) found that the relative representation of cyanobacteria and sensitive diatoms decreased, and that of nitrogen-heterotrophic, eutrophic, and motile diatoms increased, with increasing urban runoff and, especially, wastewater inputs with high ion and nutrient loads (similar to California results in Porter et al., 2008). The

biovolume of green algae was higher and the relative abundances and richness of diatoms were lower in concrete-lined channels with higher light and temperature levels than in natural streams or channels with natural bottoms. Working in 104 streams in southern California, Stancheva et al. (2012) found that the biovolume and richness of non-diatom benthic algae were positively correlated with temperature and light levels, and that heterocystous cyanobacteria and Zygnemataceae were negatively related to nitrogen concentrations and red algae were negatively related to TP concentrations.

WWTP discharge in some med-rivers during the dry season can be the major or only source of water and nutrients to downstream reaches, promoting the luxuriant growth of vascular plants or macroalgae (e.g., *Cladophora*), which can sequester large amounts of nutrients (Simpson, 2006; Martí et al., 2010; Klose et al., 2012). Nutrient uptake and transformation rates are often high, but nutrient uptake efficiencies are often low, in streams receiving substantial WWTP inputs, and nitrogen incorporated into plant and algae tissue is usually scoured from stream bottoms and transported downstream during storms (Klose et al., 2009; Martí et al., 2010; Merseburger et al., 2011).

Because aquatic macrophytes are sensitive to disturbance regimes, substrata characteristics, and light, nutrient, and contaminant levels (Simpson, 2006; Going et al., 2008), they are affected by human alterations to landscapes and the water cycle, often showing clear differences in species composition, diversity, and cover between disturbed and undisturbed sites (Abou-Hamdan et al., 2005; Bernez & Ferreira, 2007; Ceschin et al., 2010; Kargioglul et al., 2012).

Estuaries receive and accumulate nutrients from their drainage basins often resulting in eutrophication characterized by blooms of cyanobacteria (*Nodularia spumigena* Mertens) or green macroalgae (e.g., *Cladophora*, *Enteromorpha*) in spring and summer. For example, the large, shallow Peel-Harvey estuary of southwestern Australia receives inputs of phosphorus from several rivers, which fuel algal blooms causing anoxic conditions and fish kills (McComb & Davis, 1993; Ritchie & Weaver, 1993; Keipert et al., 2008). Total nitrogen concentrations in the Berg Estuary in the South African Cape have increased from <300 µg/l prior to 1980 to nearly 2,000 µg/l in 2005 and chlorophyll *a* concentrations have increased from 1.8 to 8.2 µg/l in winter and from 0.2 to 1.2 µg/l in summer over a similar time period, all concordant with increases in catchment agricultural development (C. A. P. E., 2008).

Invertebrates

Stream and river invertebrate assemblages show numerous responses to land use changes (Fig. 3d, e; Solimini et al., 2000; Alvarez & Pardo, 2007a, b; Mancilla et al., 2009b; Ode et al., 2010; Sánchez-Montoya et al., 2010; Waite et al., 2010; Cheimonopoulou et al., 2011; Mazor et al., 2011). The relative and absolute richness, density, and biomass of many stream insect orders, particularly the Ephemeroptera, Plecoptera, and Trichoptera (EPT), decline, whereas those of some dipterans (e.g., Chironomidae, Muscidae) and non-insects (nematodes, oligochaetes, leeches, microcrustaceans, snails) increase, with increasing catchment development (Fig. 3d, e; Fend et al., 2005; Alvarez & Pardo, 2007b; Brinkman, 2007; Dallas & Day, 2007; Mazor & Schiff, 2008; Song et al., 2009; Correa-Araneda et al., 2010; Theodoropoulos & Iliopoulou-Georgudaki, 2010; Waite et al., 2010). Most studies in med-regions have observed much larger

effects of urbanization than agricultural development on stream communities, but even moderate human influences, such as light livestock grazing or legacies in abandoned farmlands, can result in the loss of sensitive groups (e.g., Plecoptera) (Brinkman, 2007; Mazor et al., 2011). Because the invertebrate faunas of streams in most med-regions contain many endemic species which are vulnerable to environmental changes, land use alterations can have large effects on local and global biodiversity (Bonada et al., 2004; Samways et al., 2011). For example, in southwestern Western Australia the insect and crustacean faunas have high levels of endemism with many taxa which are sensitive to increases in stream salinity largely created by land clearing for the cultivation of crops (Bunn & Davies, 1990; Horwitz, 1997; Halse et al., 2003).

Among functional feeding groups, the richness and diversity of shredders, scrapers, and predators have been observed to be lower in urbanized than in undisturbed streams (Brinkman, 2007; Mazor & Schiff, 2008). On the other hand, agricultural and urban environments provide habitats, such as ornamental, farm, recreational, or sewage treatment ponds or lakes, storage reservoirs, water mains, tyre pools, and drainage channels which support specific insect taxa, particularly Diptera (Culicidae, Chironomidae, Chaoboridae), which can attain nuisance levels and transmit disease (Resh & Grodhaus, 1983).

Although numerous studies have shown clear invertebrate responses to land use changes, it is often difficult to pinpoint why land use changes have these effects, given the numerous ways that land use influences physical–chemical factors. In southern California, Brinkman (2007) found that many invertebrate metrics, such as EPT richness, were strongly related to a habitat index which incorporated information on bank and bottom stability, substrata size, and disturbance to channels and riparian vegetation. Ode et al. (2010) and Mazor et al. (2011) reported that invertebrate biomonitoring metrics were related to many abiotic variables, but particularly substratum size (%sand + fines), nutrient concentrations (TN, TP), and in-stream habitat (channel alteration). Working in central Chile, Mancilla et al. (2009b) reported that macroinvertebrate distributions were related to conductivity, bank integrity, and benthic habitat and Burton et al. (2005) reported that bottom habitat conditions, rather than stream chemistry, appeared to

have the greatest impacts on invertebrate community structure in the Santa Ana River system of southern California.

Changes to river hydrological regimes and sediment dynamics owing to land use changes also alter invertebrate communities. Land use changes often increase scouring flood flows, removing and transporting invertebrates downstream and fostering the dominance of quick-colonizing, r-selected species, with immediate or short-term effects on invertebrate densities, species composition, and richness (Gasith & Resh, 1999). At the other extreme, land use changes can decrease stream flows and increase the duration of dry periods during the dry season, selecting for species that can withstand or recover quickly from droughts (Lake, 2011). If dry periods become too prolonged, however, it is unlikely that most invertebrate species can survive. Finally, in some cases, return flows from agricultural and urban areas can increase dry season flows, both disrupting the life cycles of species adapted to seasonal drought and increasing the survival and densities of exotic species which require perennial flow, sometimes resulting in the demise of native species through their interactions with exotic species (Brown et al., 2005a; Riley et al., 2005). The construction of dams, in particular, has numerous effects on aquatic invertebrates by converting upstream lotic to lentic habitats and altering downstream hydrology, hydrochemistry, and geomorphology (Kondolf, 1998; Bredenhand & Samways, 2009; Sabo et al., 2012). For example, sediments are often intercepted and stored by reservoirs, with increased scouring of downstream reaches during high-flow releases, resulting in increased incision, decreases in turbidity, increases in N_2 and O_2 levels, and the armoring of downstream areas (Kondolf, 1997; Sabo et al., 2012). Bredenhand and Samways (2009) reported that EPT taxa were diverse and abundant in flowing sections above a dam in the Cape of South Africa, but were almost absent in the river below the impoundment, with chironomids reaching high abundances below the dam.

Because riparian vegetation provides shading, bank stability, and allochthonous inputs to streams and filters sediment and contaminants from upland areas, human-induced changes to riparian zones can have far-ranging effects on invertebrate communities by altering environmental conditions, food sources, and adult habitat (Aguiar et al., 2002; Samways &

Sharratt, 2009; Magoba & Samways, 2010; Theodoropoulos & Iliopoulou-Georgudaki, 2010; Samways et al., 2011). By increasing flood flows, land use changes result in the increased dislodgement, transport, and loss of leaf litter and associated invertebrates. Stream channelization eliminates depositional habitat, decreasing the retention of leaf litter and reducing shredder densities (Gasith & Resh, 1999). In addition, human-induced changes in the composition of riparian vegetation can affect the quantity and quality of inputs of woody and leaf debris, with repercussions for invertebrate habitat structure and food supplies (Mancilla et al., 2009a). Valdovinos (2001) reported that decomposition rates and shredder (particularly Plecoptera: Gripopterygidae) densities were much higher for native *Nothofagus pumilio* Krasser leaves than for the leaves of introduced Monterey pine (*Pinus radiata* D. Don) in woodland streams in central Chile. In Central Portugal, Abelho and Graça (1996) found that leaf decomposition rates, and the richness and densities of aquatic invertebrates, in stream reaches with riparian zones dominated by deciduous trees were higher than those in reaches surrounded by *Eucalyptus* monocultures. Magoba and Samways (2010) found that stream reaches bordered by native vegetation or cleared of invasive alien trees (primarily *Acacia*) had a higher richness of EPT and Odonata (EPTO) taxa and density of adult odonates than reaches lined by invasive alien trees in the Cape of South Africa. The ingestion, growth, and survival rates of shredders appear to be determined by the structural and chemical characteristics of litter from different plant species rather than by whether or not plants are native or exotic (e.g., Going & Dudley, 2008).

Some studies have reported that scraper densities decline with increasing human disturbance, presumably because the effects of sediment deposition override the effects of increased algal biomass or because scrapers, which consume microalgae, cannot handle or ingest the filamentous macroalgae often promoted by high nutrient and light levels (Dudley et al., 1986; Dudley, 1992; Burton et al., 2005). Although microalgae (primarily diatoms) can grow as epiphytes on macroalgae, many scrapers remove diatoms from hard substrata and are ill-equipped to deal with epiphytes. On the other hand, some collector-gatherer taxa, such as chironomids or baetid mayflies, can remove epiphytes from macroalgae and are abundant where filamentous algae proliferate

(Dudley et al., 1986; Dudley, 1992). Collector-gatherer taxa are often abundant at moderately enriched sites with abundant food sources (Ortiz & Puig, 2007), but then decline at high levels of enrichment owing to reductions in nighttime dissolved oxygen levels or to the excessive loading of toxins (Resh and Grodhaus, 1983; Sánchez-Pérez et al., 2009). Most invertebrate species are absent below WWTPs releasing large quantities of organic matter because of low dissolved oxygen concentrations, but invertebrate communities recover downstream, showing a longitudinal succession of invertebrate species of increasing sensitivity to low oxygen levels (Johnson et al., 1993; Prenda & Gallardo-Mayenco, 1996; Ortiz et al., 2005, 2008). Finally, collectors often reach highest abundances below dams, where filter-feeders intercept plankton flowing out of reservoirs and deposit feeders benefit from high fine particulate organic matter (<1 mm, FPOM) quality; however, shredders are almost absent because of low coarse particulate organic matter (CPOM) levels (Casas et al., 2000).

Stream macroinvertebrates also respond directly to increases in temperature and to inputs of contaminants, such as metals, pharmaceutical products, biocides, and other organic pollutants (Paul & Meyer, 2001; Muñoz et al., 2009; Ginebreda et al., 2010; Dallas & Ketley, 2011; Hamon, 2011; Dallas & Rivers-Moore, 2012). The effects of temperature increases and pollutant inputs on the biota can be multiple, from an additional physiological stress to the biota to extirpation of sensitive species to selection for tolerant species (Ricart et al., 2010), with a variety of food web implications (Muñoz et al., 2009). An extensive literature documents the effects of toxin and nutrient inputs in wastewater effluent on invertebrate populations and communities, with effluent concentrations as low as 5% altering macroinvertebrate assemblages (Johnson et al., 1993; Prat & Munné, 2000; Ortiz & Puig, 2007; Grantham et al., 2012). Muñoz et al. (2009) reported that concentrations of some pharmaceuticals (anti-inflammatories, beta-blockers) were positively related to the densities and biomass of *Chironomus* L. spp. and oligochaetes, but negatively related to other taxa, in the Llobregat River of northeastern Spain. Burton et al. (2005) found that EPT and non-chironomid Diptera richness was higher and the relative abundances of oligochaetes and Orthocladinae (especially *Cricotopus* Wulp) lower in natural streams than in channels receiving

wastewater effluent. Bollmohr & Schulz (2009) reported that sensitive insect species, mainly mayflies and caddisflies, were less abundant at a site containing high concentrations of organophosphorus pesticides associated with suspended sediment than at less contaminated sites, particularly in the dry season, in a South African med-river.

High salinity also can have a large effect on invertebrate communities in some med-streams. For example, taxon richness was low and densities of chironomid larvae and ostracods were high in the Hotham River (salinity 4.4–6 ppt) of southwestern Australia (Bunn & Davies, 1992). In contrast, the invertebrate community of the saline Blackwood River was not influenced by salinity levels, suggesting that its fauna was adapted to high salinities (Williams et al., 1991; Kay et al., 2001). Hyporheic invertebrate richness across 13 streams in southwestern Western Australia was not correlated with salinity levels, but there were significant changes in assemblage structure with increasing salinity, with dominant taxa appearing to be halotolerant (Boulton et al., 2007). In contrast, some species, such as the endemic freshwater mussel *Westralunio carteri* Iredale, have undergone considerable range retractions associated with rising salinity (Beatty et al., 2010).

Finally, exotic aquatic invertebrates, such as the New Zealand mud snail [*Potamopyrgus antipodarum* (J. E. Gray)], the Asiatic clam [*Corbicula fluminea* (O. F. Müller)], the Louisiana red swamp crayfish (*Procambarus clarkia* Girard), and others, as well as many exotic fish, have been introduced into or invaded med-rivers and are often associated with rivers altered by human activity (Gil-Sánchez & Alba-Tercedor, 2002; Riley et al., 2005; Brown et al., 2005a, b). Investigations of the effects of exotic species on the native biota have produced results ranging from trivial to profound, indicating that much more research is needed to determine the characteristics of exotic species and recipient communities which determine the impacts of exotic species on recipient communities (Múrria et al., 2008; Lowe et al., 2010; Klose & Cooper, 2012a, b).

Fish and amphibians

Owing to past isolation, high variation in flow regimes, and frequent droughts, freshwater fish communities in mediterranean climates are usually depauperate, but often with a high proportion of endemic

species (Swift et al., 1993; Bianco, 1995; Economidis, 1995; Vila-Gispert et al., 2002; Lin & Ambrose, 2005; Marchetti et al., 2006; Ferreira et al., 2007; Impson, 2007; Zogaris et al., 2009; Clavero et al., 2010; Marr et al., 2010). Fish species richness in these regions has been considerably augmented by the introduction or invasion of exotic species, often associated with reservoirs and altered flow regimes (Bianco, 1995, 1998; Economidis et al., 2000; Corbacho & Sánchez, 2001; Elvira & Almodóvar, 2001; Marchetti et al., 2004; Innal & Erk'akan, 2006; Kara, 2011; Hermoso et al., 2012). Most, but not all, studies in southern California have reported a positive relationship between the number of exotic fish species and urbanization (Fig. 3c; Brown et al., 2005b; Lin & Ambrose, 2005; Riley et al., 2005). Across California, Marchetti et al. (2004) found that exotic fish species distributions were related strongly to human-induced changes to the landscape (agricultural and urban development) and river systems (dams, diversions, aqueducts). Currently, fish community composition in urbanized areas of California varies greatly from site-to-site, primarily attributable to sites of exotic species introductions and limited fish dispersal powers (Marchetti et al., 2006). In the Guadiana River system in Spain, Corbacho and Sánchez (2001) reported that the richness of native fish species declined and of exotic fish species increased with increasing channelization and river regulation. Similarly, the ten native freshwater fish species in southwestern Australia, including eight endemics, were found primarily at wildland sites, but exotic fish species were mainly observed at sites affected by human activity (Morgan et al., 2003). In addition, endemic Mediterranean Basin fish species have been introduced outside their natural ranges with individual cases of hybridization and genetic dilution (Bianco, 1998).

Clavero et al. (2010) reported that exotic species and waterscape alterations (primarily water extraction) were the dominant threats to endemic species in the Iberian Peninsula, central Italy, and parts of northwestern Turkey, but that agricultural activity and pollution were the major threats to endemic species in the southern Balkan Peninsula and along the Mediterranean eastern rim. Analyses at regional or basin scales suggest that exotic species have had negative effects on native species (Light & Marchetti, 2007; Hermoso et al., 2011, 2012). Other researchers have argued that evidence for the magnitudes and

mechanisms of exotic fish species impacts on native species is largely circumstantial and inconclusive (Ribeiro & Leunda, 2012). In the Western Cape Province of South Africa, the introduction of exotic fish species, particularly piscivorous species like the smallmouth bass (*Micropterus dolomieu* Lacepede), has resulted in the decline and sometimes local extirpation of native fish species (Clark et al., 2009; Lowe et al., 2010; Woodford et al., 2010). Although fish faunas in the med-regions of the world were originally distinctive, there is increasing homogenization as native species dwindle and introduced species become widespread and dominant (Clavero & García-Berthou, 2006; Marr et al., 2010; Clavero & Hermoso, 2011; Hermoso et al., 2012).

The life cycles and behavior of native aquatic vertebrates are well adapted to the habitats and seasonal flow patterns found in med-streams, whereas many introduced species are not, perhaps accounting for their absence in wildland streams and their dominance in streams altered by human activity (similar to Meffe, 1984). The distribution of aquatic vertebrate species also may be affected by the construction of in-stream barriers such as dams, road crossings, bridges, and culverts, which inhibit the movement of native fish species. Such barriers prevent the dispersal of fish to areas where their local populations have been extirpated by disturbances (e.g., fire, toxin or oil spills) (Verkaik et al., 2012) and block the migrations of migratory species (e.g., the anadromous salmonids of California, the catadromous eels of Spain, Portugal, and North Africa) (Kettle et al., 2011; Moyle et al., 2011; National Marine Fisheries Service, 2012). The construction of dams appears to be the major cause of severe declines or local extinctions in migratory fish populations in med-regions (Innal & Erk'akan, 2006; Moyle et al., 2008). In a review on the rapid loss of most freshwater fish species from California, Moyle et al. (2011) concluded that the greatest threats were exotic species, agricultural development, and dams.

Stream fish often show sensitive physiological, demographic, and community responses to inputs of contaminants, including metals, biocides, PCBs, PAHs, pharmaceuticals, and many others, with fish often being used as sentinel species for monitoring pollutant loads (Hamon, 2011; Petrovic et al., 2011). Balik (1995), Bacha & Amara (2007), and Vila-Gispert et al. (2002) identified pollution as a major

influence on freshwater fish communities in Turkey, Algeria, and Spain, respectively, whereas fish assemblage structure in an Israeli stream was related to organic loads (Gafny et al., 2000). Fish also respond to changes in the concentrations of major ions engendered by land use alterations. The forested tributaries of the lower Blackwood River in southwestern Western Australia are fed by freshwater springs and harbor populations of four of the six freshwater species endemic to this system, whereas the saline upper tributaries and main channel are dominated by invasive mosquitofish (*Gambusia holbrooki* Girard), followed by several estuarine species and two native species (the cobbler *Tandanus bostocki* Whitley and the western minnow *Galaxias occidentalis* Ogilby) (Morgan et al., 2003; Beatty et al., 2006). Salinity tolerances for one exotic and three endemic fish species closely mapped the distributions of these fish species within this catchment and suggested that its endemic fish fauna is threatened by projected increases in salinity associated with land use changes, increased groundwater pumping, reduced groundwater inputs, and climate change (Beatty et al., 2010, 2011).

Amphibian communities also show substantial levels of endemism in many med-regions (Turner & de Villiers, 2007; Vidal et al., 2009; Pascual et al., 2011) and have been affected by habitat destruction and degradation, alterations to the hydrological cycle, pollution, frequent fires, and the introduction of exotic plant and animal species (Turner & de Villiers, 2007; Lillo et al., 2011; Mokhatla et al., 2012). Although many amphibian species are tied to temporary and permanent ponds and other wetlands, some are found in or along streams and rivers (such as the ghost and stream frogs of South Africa (*Heleophryne* Sclater, *Strongylopus* Tschudi), with stream breeding species being among the most threatened (Turner & de Villiers, 2007; Mokhatla et al., 2012). Riley et al. (2005) reported that native amphibian species were common in wildland streams but that most were greatly reduced in streams draining urban areas in southern California. Amphibian assemblages may be particularly sensitive to land use changes because they are directly affected by perturbations in both terrestrial and aquatic environments (Ficetola et al., 2011). Some exotic amphibian species, such as the American bullfrog (*Lithobates catesbeianus* Shaw) and the African clawed frog (*Xenopus laevis* Daudin), have

invaded several med-regions with potentially important effects on native amphibian species (Lobos & Measey, 2002; Lillo et al., 2011).

Land use and climate change

Climate change models indicate that many med-regions will become warmer and drier, with an increase in the frequency, intensity, and duration of droughts (Burke et al., 2006). More specifically, Mediterranean Europe (e.g., Lehner et al., 2006), southwestern North America (Seager et al., 2007), the eastern Mediterranean rim (Chenoweth et al., 2011), central Chile (Urrutia et al., 2011), and southwestern West Australia (e.g., Mpelasoka et al., 2008; Van Ommen & Morgan, 2010) are projected to receive less rainfall and undergo an increase in drought severity. Although invertebrates in med-streams may extend their ranges to higher latitudes with warming (Bonada et al., 2007a, b), invertebrates in streams in southwestern Australia and southwestern Africa may be particularly vulnerable because there are no adjacent lands at higher latitudes (Davies, 2010; Dallas & Rivers-Moore, 2012). Furthermore, med-stream invertebrates which are cold or cool stenotherms, occupying streams at high elevations or fed by springs, may become locally extirpated as temperatures increase and groundwater inputs decrease (Dallas & Ketley, 2011; Dallas & Rivers-Moore, 2012). The largest effects of climate change on biological communities in med-streams and rivers likely will be mediated through effects on hydrological regimes which have already been altered by dams, water extraction, and land use changes. Increases in the frequency and duration of droughts will alter habitat and the densities of many stream species unless flows are artificially maintained (Bêche et al., 2009; Hermoso & Clavero, 2011; Lake, 2011; Moyle et al., 2011). Because of a drying climate and growing human populations in med-regions, competition between humans and in-stream resources for water are destined to increase (Gasith & Resh, 1999). Finally, land use changes may affect climatic trends directly. For example, the loss of native vegetation will reduce surface roughness in southwestern Western Australia, decreasing the interception of clouds and reducing regional precipitation (Pitman et al., 2004; Kala et al., 2011).

Similarity of the effects of land use changes on rivers and streams in mediterranean and other systems

Many of the effects of land use change on streams and rivers in med-regions, as reviewed above, are similar to those observed in other climatic regions of the world (Paul & Meyer, 2001; Allan, 2004; Walsh et al., 2005; Johnson & Horst, 2010; Feld et al., 2011; Clapcott et al., 2012). The effects of land use change on ecological communities, then, are likely to differ more in magnitude than kind in mediterranean compared to other climates. Because climate, by definition, distinguishes mediterranean from other regions, any differences between mediterranean and non-mediterranean systems will have their roots in climatic conditions. In general, differences in the impacts of land use changes on mediterranean versus other streams ultimately stem from the benign thermal environment and high seasonal and interannual variation in runoff and flow (Gasith & Resh, 1999). The benign med-climate has fostered a long history of human–environment interactions, continued human population growth, and large-scale changes in land use and cover (Hooke, 2006; Underwood et al., 2009; Sirami et al., 2010). In fact, some med-shrublands and grasslands may be a product of past human alterations to landscapes and the high diversity and endemism of some groups may have been generated and maintained by a mosaic of human-induced disturbances (Sirami et al., 2010; Gammage, 2011). Because most med-regions have had long periods of aridity and isolation from other regions, they contain many endemic species adapted to seasonal wet and dry periods and to supra-seasonal perturbations (fires, extreme floods, sustained droughts). Human activity has extensively modified most lowland med-areas and has encroached into some foothill and montane areas, producing intense and frequent fires and high erosion rates.

Based on our review, land use changes have exaggerated hydrological variation in med-rivers. Land use changes increase peak flood flows, decrease lag times between precipitation events and runoff, enhance the sharpness of hydrograph peaks, and decrease dry season flows (Konrad & Booth, 2005). Although organisms in med-streams are resilient to natural flow variation (e.g., Gasith & Resh, 1999; Acuña et al., 2005; Magalhães et al., 2007; Artigas et al., 2009; Boix et al., 2010; Gaudes et al., 2010),

land use changes create more intense hydrological disturbance, perhaps exceeding the capabilities of stream organisms for recovering from intense floods or, especially, from prolonged periods of drying (Magalhães et al., 2007; Bêche et al., 2009; Lake, 2011). As for streams in other climates, however, flows can be augmented by aseasonal dam releases, return flows from irrigation, or sewage treatment plants, which may allow exotic species to become established and expand (Riley et al., 2005).

The semi-arid climate and large human populations of med-regions have created intense competition between humans and natural environments for clean water (Gasith & Resh, 1999). Numerous dams have been built on rivers to store water during wet seasons for use in dry seasons and dry years, resulting in a scarcity of natural, free-flowing rivers in many med-regions. Furthermore, because of high human demand for water, the hydrological cycle has been extensively modified by surface and groundwater abstractions, water diversions, inter-basin transfers, and water conveyance facilities. In general, the amount of reservoir habitat has increased, the timing, magnitude, and frequency of downstream flows have been changed, and species have been transported to non-native habitats, often reducing native species and facilitating the establishment and expansion of exotic species (Marchetti et al., 2004, 2006; Clavero & Hermoso, 2011; Fig. 1). For example, low flows in winter can create frequent and severe unnatural winter droughts in stream channels below dams which “may be the principal threat for the conservation of freshwater biodiversity” in some streams (Belmar et al., 2010), and aquatic habitat is destroyed by dams which cause prolonged periods of low flow or drying in the dry season. The extraction of water from med-streams which produces unnaturally low flows can simplify the structure of invertebrate (e.g., Boix et al., 2010) and fish assemblages (e.g., Benejam et al., 2010; Clavero et al., 2010). Alterations to riverscapes and landscapes often proceed hand-in-hand, such as where dams protect downstream areas from flooding thereby, often in tandem with the construction of levees or other structures, allowing human development on downstream floodplains. The construction and operation of dams have particularly severe effects on migratory species by blocking migrations and changing flow patterns (Moyle et al., 2008).

With the exception of Australia, most med-rivers and streams have their headwaters in mountainous

areas. The steep slopes and highly erodible soils of many med-regions, combined with intense fall or winter rainfall, often produce high erosion rates and increased sediment inputs to streams. Milliman and Farnsworth (2011), using a worldwide database of 1,534 rivers, reported that the highest concentrations of suspended sediments tended to be located in subtropical arid to semi-arid regions, including those in med-regions. For example, suspended sediment concentrations averaged $45,000 \text{ mg l}^{-1}$ in the Miliane River (Tunisia) and $15,000 \text{ mg l}^{-1}$ in the Santa Clara River (California, USA), compared to a global average of 500 mg l^{-1} . Small rivers draining into the Mediterranean Sea had high sediment concentrations before dams were built, and the construction of dams on many med-rivers has reduced sediment fluxes in downstream reaches (Milliman & Farnsworth, 2011).

Land use changes exacerbate erosion and sedimentation processes, sometimes clogging and widening stream channels (Booth, 1990). High flows and sediment transport during floods scour substrata and abrade organisms, reducing algal, invertebrate, and fish populations, whereas increased sediment deposition during falling hydrograph limbs can smother coarse substrata, reducing oxygen penetration, covering and burying organisms, and decreasing habitat heterogeneity. Conversely, by covering landscapes with impervious surfaces, flood peaks are heightened while sediment inputs are reduced, resulting in downcutting and bank erosion in soil-lined channels (Booth, 1990; Konrad & Booth, 2005). In many developed areas, rivers and streams are channelized and lined with concrete which eliminates most natural habitat, severely reducing species richness (Burton et al., 2005).

High human population densities and/or alterations of landscapes and rivers in mediterranean and other climatic regions have increased contaminant loading to rivers and streams, with effects on organisms ranging from enrichment to lethality. Evidence from other climatic regions indicates that nutrient-enriched organic matter derived from sewage may increase primary consumer productivity and shift communities to shorter food chains dominated by fast-growing species (Singer & Battin, 2007). On the other hand, excessive organic matter inputs can deplete oxygen and inputs of metals, biocides, pharmaceuticals, PCPs, and a variety of additional organic compounds can have a variety of sublethal and lethal effects on stream

organisms (Johnson et al., 1993; Muñoz et al., 2009; Ginebreda et al., 2010; Hamon, 2011; Petrovic et al., 2011). Although such effects should be similar in mediterranean and non-mediterranean regions, the effects of these contaminants on organisms may be exacerbated by mediterranean flow regimes. Contaminant loading is usually associated with winter floods with diluted contaminant concentrations, but contaminants may become concentrated during low or intermittent flow periods (Gasith & Resh, 1999; Guasch et al., 2010). Because contaminant-laden return flows or sewage effluent may account for most, if not all, of the discharge in some med-rivers during the dry season, pollutant concentrations may become high at these times, with sublethal or lethal effects on stream organisms (Prat & Munné, 2000; Burton et al., 2005; Ortiz et al., 2005, 2008; Ortiz & Puig, 2007; Klose et al., 2009; Petrovic et al., 2011).

Recent analyses allow comparisons of nutrient export from mediterranean and non-mediterranean catchments. As reported above, nitrate yields from med-catchments ranged from 3 to $829 \text{ mol ha}^{-1} \text{ y}^{-1}$ (excluding subcatchments with greenhouse agriculture). Lewis (2002) reported that nitrogen export from 19 US catchments with natural land cover and low to moderate atmospheric nitrate deposition averaged $39 \text{ mol ha}^{-1} \text{ y}^{-1}$; Campbell et al. (2000) found that nitrate export from 15 upland, forested catchments in New England ranged from 7 to $171 \text{ mol ha}^{-1} \text{ y}^{-1}$; and Sickman et al. (2003) reported average nitrate yields of $49 \text{ mol ha}^{-1} \text{ y}^{-1}$ for 15 Sierra Nevada sites and $79 \text{ mol ha}^{-1} \text{ y}^{-1}$ for 13 Rocky Mountain sites. In a review of nitrogen export from 946 river basins throughout the world, Alvarez-Cobelas et al. (2008) found that annual nitrate export ranged from 0.01 to $7,558 \text{ mol ha}^{-1} \text{ y}^{-1}$, with croplands exporting more nitrogen than pastures or forests and xeric watersheds exporting less nitrogen than mesic watersheds. Based on a review of 685 river catchments throughout the world, Alvarez-Cobelas et al. (2009) reported that human population density and runoff were the major predictors of phosphorus export. They reported that P export was most variable in mediterranean and temperate shrublands and grasslands, and speculated that the wide variety of habitats and long history of land use change may be sources of this variability.

Most nutrient loading is associated with high runoff from drainage basins during floods. After wet seasons with high flows, California streams often have spring

blooms of green filamentous algae (e.g., *Cladophora*), presumably because of increased nutrient loading and decreased grazing and in-stream shading, and are dominated by vagile, opportunistic invertebrate taxa which quickly colonize denuded substrata, such as baetid mayflies, blackflies, and chironomids (Cooper et al., 1986; Feminella & Resh, 1991). During the dry season, med-streams are characterized by long periods of low or intermittent flow when temperatures, photosynthetic and respiration rates, biological oxygen demand, plant and microbial biomass and production, and silt deposition are all high, nutrient dilution is reduced, and detritus and algae can accumulate to high levels, provided allochthonous inputs or nutrient and light levels are sufficient (Gasith & Resh, 1999). Algal biomass can be particularly high in open rivers receiving augmented nutrient inputs from return flows or sewage treatment plants (Prat & Munné, 2000; Klose et al., 2009). Low or intermittent flows in open rivers combined with high nutrient inputs can fuel algal blooms which result in supersaturated oxygen levels in the daytime, owing to high algal photosynthetic rates, and depleted dissolved oxygen levels at night, owing to algal and microbial respiration, sometimes to levels which are deleterious to fish and invertebrates (Klose et al., 2012). Many dry season conditions stemming from low flows will persist through the winter into the following spring in perennial streams during droughts, so human impacts on sedimentation, nutrient loading, and riparian vegetation will continue to accumulate in those years. Open, polluted, downstream reaches of med-rivers often accumulate high growths of algae or aquatic plants across drought years or below dams (Klose et al., 2009). Except where dams reduce flood flows, this accumulated primary producer biomass is exported downstream during subsequent floods in high rainfall years.

As in many arid and semi-arid regions, riparian vegetation and canopy cover play an especially important role in ameliorating temperature extremes, providing carbon sources and cover, stabilizing banks, and filtering sediment and contaminants out of upland runoff in med-streams (Naiman & Décamps, 1997; Sabater et al., 2000, 2006; Sponseller et al., 2001; Roy et al., 2007). Because humans have destroyed, degraded, fragmented, and altered the composition of riparian habitat in many med-regions (Corbacho et al., 2003), they have changed the cycling of

nutrients and processing of organic matter, as well as modified the composition, biomass, and productivity of communities in med-streams. Where humans reduce riparian cover and promote exotic plant species, they affect the quantity, quality, and timing of allochthonous inputs to streams, making microbial composition, biomass, and production and the densities of invertebrate shredders sensitive indicators of riparian perturbations (Moline & Poff, 2008).

Because med-ecosystems exhibit high levels of endemism, human impacts on these systems have the potential to extirpate native species and reduce local, regional, and global native biodiversity (Clavero et al., 2010). Due to the ubiquity and intensity of human impacts on med-streams and rivers, many endemic species in med-ecosystems are threatened (Moyle et al., 2011). On the other hand, overall diversity in med-regions may remain high because of the introduction or invasion of exotic species, which benefit from the construction of reservoirs, changes to hydrological regimes, enrichment, and other human-induced changes to riparian and aquatic environments (Marchetti et al., 2004, 2006; Hermoso et al., 2011, 2012).

In summary, although human-induced changes to hydrological, geomorphological, and hydrochemical conditions have had similar effects on mediterranean and non-mediterranean stream and river ecosystems, human impacts on med-ecosystems may be more extensive and intensive because of a long history of human settlement and high levels of landscape and river alteration. Land use impacts on med-rivers and streams are exacerbated by high variation in flow regimes, leading to greater flood and drought disturbances, increased erosion and sedimentation, increased contaminant loading, higher pollutant concentrations during low-flow periods, and enhanced impacts on riparian vegetation. The extensive destruction and alteration of med-environments may reduce native species, which were adapted to natural environmental conditions, but favor exotic species, which thrive under altered conditions (Hermoso et al., 2012). Because aquatic faunas have high levels of endemism in med-regions, these multifarious, intensive, and ubiquitous impacts of humans on mediterranean aquatic environments have repercussions for the survival of species and global biodiversity. Land use and climate changes are proceeding in tandem, so future research needs to address how predicted climate change will interact with land use changes to affect

natural ecosystems and their ability to deliver ecosystem services (Underwood et al., 2009). Because a number of native riverine species, such as the southern California steelhead, are near their thermal range limits, increasing temperatures and decreasing flows may extirpate such species from some med-regions (Dallas & Rivers-Moore, 2012; National Marine Fisheries Service, 2012). The combination of climate and land use changes, then, are likely to have far-reaching effects on ecological communities in regions with med-climates (Bêche et al., 2009; Klausmeyer & Shaw, 2009).

Management and future directions

From this review, it becomes apparent that land use changes, and associated alterations of hydrological regimes, hydrochemistry, aquatic habitat, and sediment mobilization, transport, and deposition, have profound, extensive impacts on aquatic-ecosystems which will accelerate in the future. Because the impacts of land use changes on aquatic ecosystems primarily address issues of environmental assessment, management, and policy, we believe that future research should focus on providing information needed to make effective management and policy decisions.

Biomonitoring

Owing to a variety of directives, such as the Clean Water Act in the US, the Water Framework Directive in Europe, and the National Water Act in South Africa, there has been an explosion of interest and efforts in stream biomonitoring in med-regions (Iliopoulou-Georgoudaki et al., 2003; River Health Programme, 2003; Dallas & Day, 2007; Magalhães et al., 2008; Mazor & Schiff, 2008; Hermoso et al., 2009, 2010; Poquet et al., 2009; Ode et al., 2010; Aparicio et al., 2011; Friberg et al., 2011; Mazor et al., 2011; Munné & Prat, 2011). Because of the distinctive characteristics of med-streams and rivers, however, standard biomonitoring practices will need to be modified for application to med-ecosystems (Gasith & Resh, 1999; Alvarez-Cobelas et al., 2005). Of particular importance for biomonitoring and management efforts in med-regions is a consideration of the high temporal and spatial variation in the physical, chemical, and biological characteristics of streams and rivers.

The studies reviewed above show that many species of microbes, algae, plants, and animals in med-ecosystems show sensitive responses to land use changes, so the choice of biological metrics for assessing human impacts on these systems often depends on logistical considerations, such as the ease and cost of sampling, identification, and analysis (Bonada et al., 2006; Resh, 2008). Beyond measurements of hydrological and hydrochemical parameters, most monitoring efforts have concentrated on macroinvertebrate assemblages, with some attention to algae and fish (Rosenberg & Resh, 1993; Fitzpatrick et al., 2001; Brown et al., 2005a, b; Bonada et al., 2006; Hermoso et al., 2010; Yates and Bailey, 2010, 2011). There have been many calls, however, to expand the repertoire of groups and variables monitored, including more emphasis on process variables, such as primary production, benthic respiration, leaf decomposition, fungal sporulation, and nutrient uptake rates. By increasing the repertoire of stream characteristics examined, managers will have increased confidence in the robustness of conclusions about human impacts across a range of land use and natural gradients (Bunn et al., 1999; Hughes et al., 2009; Walters et al., 2009; Larsen et al., 2010; Peru & Dolédec, 2010; Clapcott et al., 2012). Regardless of the biological group examined, however, most biomonitoring approaches rely on the characterization of assemblages in reference water bodies, which are minimally affected by human impacts (reviewed in Carter & Fend, 2005). Because many med-ecosystems have been extensively modified by humans, the search for reference systems can become problematic, often leading to the use of systems that have had some level of human impact. In addition, many med-ecosystems are extensively modified at lower, flatter elevations, but remain relatively undisturbed at higher, steeper elevations (Underwood et al., 2009). Because most river systems show natural changes in physical, chemical, and biological characteristics from their montane headwaters to their lowland mouths, it may not be appropriate to use communities at high elevations as reference sites for perturbed lowland sites (Fend et al., 2005). The major approaches for dealing with these problems are to show that elevation, and associated changes in stream physical factors, do not influence the variables of interest, to target reference sites whose natural features are matched to those in disturbed sites, or to account for abiotic

effects statistically (Dallas & Day, 2007; Yates & Bailey, 2010; Munné & Prat, 2011).

Similar considerations apply to the temporal dimension (Mazor et al., 2009). Because med-streams are characterized by high variation in environmental conditions related to flow, biomonitoring metrics should be tailored to particular seasons (wet vs. dry season) or consider hydrological conditions (wet vs. dry years, perennial vs. intermittent vs. ephemeral streams) (Acuña et al., 2005; Bêche et al., 2006; Bonada et al., 2007a, b, 2008; Sánchez-Montoya et al., 2007; Feio et al., 2010; Munné & Prat, 2011). Monitoring programs should be conducted over long time periods to establish the range of or trends in natural variability and to examine shifting differences between perturbed and matched reference systems, particularly given the complexities of shifting baselines (Bêche & Resh, 2007; Lawrence et al., 2010). Over longer time scales, the use of reference systems can be compromised by legacy effects, given the long time periods over which med-rivers have been altered by human activity and recent switches in land use, such as the abandonment of farming, plantation, and grazing lands in southern Europe (Harding et al., 1998; Sirami et al., 2010). Insofar as land use histories can be reconstructed, they can allow an evaluation of past legacies and current effects of land use on ecological systems (Maloney & Weller, 2011).

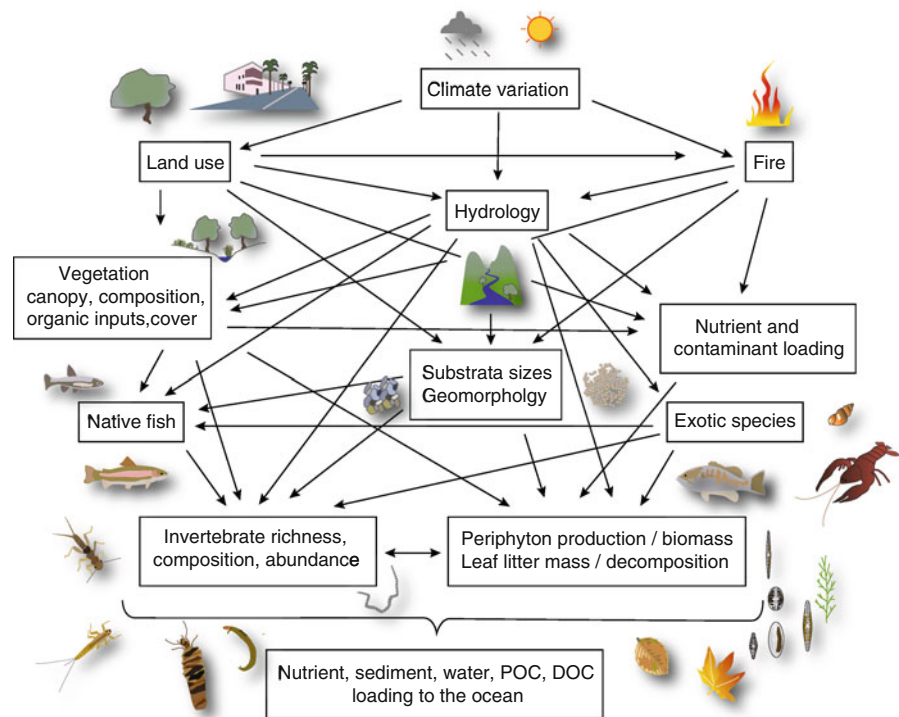
A particular focus of many biomonitoring efforts is the development and application of one or a few indices of the status (or health or integrity) of river ecosystems. Common metrics include indices of biotic integrity (IBI) which incorporate information on multiple bioindicators and RIVPACs observed/expected (O/E) measures, where relationships between biological and abiotic variables at reference sites are used to predict biological variables at perturbed sites, with subsequent comparisons to observed data (Wright et al., 1984; Ode et al., 2005; Poquet et al., 2009). Although IBI indices and O/E models are useful in presentations to governmental agencies and the public because they distill complex data into simple messages, they have been criticized because it is not clear if they add appreciably to information already contained in simple, biologically interpretable metrics (e.g., EPT richness), weighting procedures are often arbitrary, statistical properties are often obscure, and the underlying biological data are hidden (Norris & Georges, 1993; Carter & Fend, 2005;

Downes, 2010; Friberg et al., 2011; King et al., 2011; Yates & Bailey, 2011). From a scientific viewpoint, however, it is useful to present at least some of the underlying biological data to further our knowledge of ecological systems, to refine biomonitoring metrics, and to trace the distributions and abundances of specific taxa given concerns about sensitive species and biodiversity losses. New approaches to biomonitoring make extensive use of multivariate or agglomerative statistical techniques and comprehensive analysis of the responses of individual taxa or groups to perturbations, both assessing the status of sampled sites and delineating the taxa or groups responsible for responses (King et al., 2011; Clapcott et al., 2012). Rather than focus on composite indices, we encourage the inclusion of biological data in scientific papers, such as data on specific taxa or groups, as well as the further development of on-line repositories of biomonitoring data (e.g., those used in Sabo et al. 2010a, b). Furthermore, although it is possible to construct biotic indices which are indicative of human perturbations which are based on sampling a single habitat (e.g., riffles; Rehn et al., 2007), a comprehensive description of species composition and diversity will require the sampling of all major habitats (e.g., pools and riffles; stone vs. vegetated vs. fine sediment habitat) because of habitat-specific differences in the faunas of med-streams (Dallas, 2007; Bonada et al., 2008).

Future directions: pinpointing the causes of land use effects on the stream biota

Although it is clear that land use changes, and associated changes in hydrological regimes and pollution, alter stream biological communities, it is not always clear why they have these effects given the complex pathways of interaction among processes and factors which affect the stream biota at different scales of space and time (Fig. 4; Burcher et al., 2007). Nevertheless, knowledge of the pathways whereby land use or climate changes affect stream communities is often necessary for designing effective mitigation or restoration measures, particularly because different taxa show different responses to different perturbations (Downes et al., 2002; Downes, 2010; Lawrence et al., 2010). Although manipulative experiments often provide the clearest evidence for the effects of specific perturbations on ecological communities (Townsend et al., 2008; Lange et al., 2011), they can

Fig. 4 Path diagram showing cause–effect relationships leading from land use changes to stream communities and material outputs to the ocean. The diagram is based on this literature review and includes depictions of specific med-species. The *vegetation box* includes upland, riparian and aquatic vegetation and the *native fish and exotic species boxes* encompass the composition, richness, and abundances of these groups



become difficult to conduct, interpret, and replicate if more than a few factors are manipulated and extrapolations from the results of experiments in small areas to landscape scales are problematic (Cooper & Bar-muta, 1993; Englund & Cooper, 2003). In some cases, the strength of our inferences about the specific factors mediating land use effects on biological variables can be improved by monitoring and survey designs which follow the rudiments of rigorous experimental approaches, including a priori hypotheses, appropriate control sites which primarily differ from other sites in one factor (as opposed to reference sites which may differ in many ways from perturbed sites), and the random selection of sampled sites within each “treatment” (Downes et al., 2002; Downes, 2010). Although rarely conducted, follow-up work after development projects have been completed could test the accuracy of our models (environmental impact hypotheses or predictions), providing knowledge for determining and predicting the responses of systems to future human perturbations (Kondolf et al., 2007). Detailed experimental and descriptive information on the levels and interactions among abiotic and biotic ecosystem components can allow the construction of models,

which both represent the systems of interest and allow prediction of the effects of specific perturbations. Data on the covariance of system components can be used to delineate the strengths of pathways of interaction among those components (e.g., through path analysis, structural equation modeling) (Grace, 2006; Light & Marchetti, 2007; Hermoso et al., 2011, 2012); however, these approaches are still correlational and require verification with appropriate hypothesis-driven analyses, long-term observations of drivers and responses, or manipulations. As in most branches of ecology, then, any examination of the effects of specific perturbations on ecological systems requires a combination of experimental, analytical, modeling, and observational approaches so that systems can be described, parameter estimates can be refined, and predictions can be made and tested using experiments and observations at appropriate scales.

Scale is a particularly important issue in analyses of land use impacts on ecosystems because different perturbations, processes, and responses operate at different scales (Allan and Johnson, 1997; Allan et al., 1997; Cooper et al., 1998; Townsend et al., 2004). Correlational analyses, for example, indicate that the

effects of land use patterns on stream nutrient and sediment concentrations often operate at large drainage basin scales, particularly during the wet season, whereas riparian influences on light levels and leaf litter inputs or WWTP impacts on nutrient and algal concentrations often are apparent at much more local scales (Sponseller et al., 2001; Busse et al., 2006; Klose et al., 2009, 2012). Although studies have examined the effects of the scale of analysis on land use influences on river systems (Fitzpatrick et al., 2001; Sponseller et al., 2001; Wang et al., 2001; Strayer et al., 2003; Townsend et al., 2003, 2004; Pan et al., 2004; McBride & Booth, 2005; Urban et al., 2006), it is apparent that many more studies need to be done in med-ecosystems to delineate the processes which drive land use impacts at different scales of space and time (e.g., Fend et al., 2005).

Land use impacts are also spatially variable, with varying intensities and extents in different parts of drainage basins and between riparian and upland zones (King et al., 2005; Urban et al., 2006). Future research in med-ecosystems could address the effects of the spatial configuration of land use patterns on rivers (Sponseller et al., 2001; Strayer et al., 2003; Fend et al., 2005; McBride & Booth, 2005; Peterson et al., 2011). Because the inputs of sediment, nutrients, and other contaminants to streams are intimately related to the routing and magnitude of water flows through landscapes, spatially explicit eco-hydrological models have considerable potential for deciphering patchy land use effects on streams (Tague et al., 2009; Walsh & Kunapo, 2009).

Future directions: management tools and stream restoration

Managers and policy makers in med-regions have many tools or measures for preventing or mitigating the effects of land use changes on stream and river ecosystems. These measures include:

- zoning ordinances, land use restrictions, and the elimination of inappropriate subsidies to prevent development in sensitive areas,
- restrictions and treatments to reduce the use or inputs of contaminants,
- water management practices which provide appropriate river flows,
- policies and practices which protect or restore native vegetation and in-stream habitat.

Although studies in other regions have examined the effects of a variety of management practices on streams and rivers, few have examined their impacts on lotic ecosystems in med-regions. Although it is apparent that WWTP discharge affects river organisms through toxin and nutrient inputs, there have been few studies on how differences in sewage treatment levels (primary, secondary, and tertiary treatment) or storm water treatment (e.g., the use of bioswales or other managed wetlands) affect receiving water bodies in med-regions. Furthermore, determinations of maximum daily loads for contaminants require knowledge of the effects of these contaminants on biological resources and on the sources, routing, and loading of contaminants over a number of years. In general, long-term monitoring is necessary to detect trends and sources of degradation given high interannual hydrological variation. The failure of extensive management efforts to mitigate land use effects on aquatic ecosystems suggests that we are a long way from effectively modeling and controlling the impacts of human perturbations on ecosystems (Summers et al., 2001; Keipert et al., 2008).

New and sometimes large dams are still being planned and constructed, especially for hydropower generation, eliciting concerns about their social and environmental impacts. In cases where dams outlive their usefulness, owing to sediment infilling and infrastructural deterioration, there is increasing interest in removing dams to restore riverine ecosystem structure and function. Although hundreds of dams have been removed in the upper midwestern and northeastern US (Stanley & Doyle, 2003), most dam removal plans for med-regions are in their early stages. Furthermore, because of the crucial importance of water supplies in these regions, ongoing conflicts among different users, stakeholders, citizens' organizations, and governmental agencies over the construction, management, or possible removal of dams, specifically, and over the use and distribution of water resources, in general, are inevitable.

Finally, considerable funds and efforts have been directed to stream and river restoration projects, often involving the removal of exotic riparian plants, revegetation, addition of coarse elements for in-stream cover and habitat, installation of fish ladders, recontouring of banks and channels to resemble more natural configurations, and many other measures to produce more natural flow and geomorphological

conditions and increased habitat heterogeneity (Kondolf et al., 2001; Purcell et al., 2002; Bernhardt et al., 2005; Feld et al., 2011). In many cases, it appears that restoration efforts are conducted at too small a scale to be effective (Wang et al., 2003; Wohl et al., 2005; Roy et al., 2006; Bernhardt & Palmer, 2011; Herbst et al., 2012). Often restoration projects arise in response to local pressures without attention to priority planning at catchment scales. Restoration projects often do not have clear targets and do not include post-treatment monitoring programs, although such data are necessary to determine the effectiveness of restoration measures and to refine approaches for restoring river ecosystems to more natural conditions (Palmer et al., 2005; Palmer & Allan, 2006; Bernhardt & Palmer, 2007; Kondolf et al., 2007). The evaluation of the impacts of restoration projects, specifically, and of land and water management practices, in general, on river ecosystems constitutes a critical area for future research, guiding practice, policy, and adaptive management efforts (Booth et al., 2004; Kondolf et al., 2007; Feld et al., 2011).

Although it is clear that there are severe, pervasive impacts of land use changes and water development on stream and river systems in med-regions, these impacts are likely to increase given continuing population growth, urbanization, agricultural development, and the construction of tourist facilities and projected changes in climate (Underwood et al., 2009). Med-ecosystems have become a flashpoint for charting a course which balances sustainable resource use and the protection or restoration of native biodiversity.

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