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Water Quality in the Agronomic Context: Flood Irrigation Impacts on Summer In-Stream Temperature Extremes in the Interior Pacific Northwest (USA)

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1. Introduction

European arrival to the Pacific Northwest (U.S.A.) in the late 1800's signalled the beginning of an era of change for watershed use and water quality of free-flowing stream systems. Historic uses centered around trapping of beaver, utilization of livestock forage, and harvesting of anadramous fish and timber resources (Beschta, 2000). Impacts of these land uses on stream systems were severe; impaired flood plain development with beaver removal, interrupted nutrient cycling with overharvest of anadramous fish, streambank degradation due to overgrazing, and severe flooding associated with abusive logging practices (Trefethen, 1985; Meehan, 1991; Beschta, 2000). Perhaps most dramatic was the conversion of ecosystem type from stream system to impoundment associated with dam construction for hydroelectric power production (Beschta, 2000). The US Clean Water Act (CWA) of 1965 was crafted and passed into law in response to these and other issues, providing a legal and regulatory framework for managing land use practices that impact water quality (Adams 2007). Subsequent amendments of the CWA in 1972 and 1977 helped expand this framework to include impacts of non-point source pollution and mandated that states develop water protection programs (National Research Council, 1995; Adams, 2007).

While the forgoing issues continue to attract popular and political attention, legislative and regulatory mandates have dramatically reduced the acute effects of these practices and attention has now shifted to less dramatic, but nonetheless important associations between topical land use practices and water quality. There is an established and growing awareness of the impact of agriculturally-related non-point source pollution on stream systems. In this chapter we will 1) examine the scope of non-point source pollutant issues relating to the water quality/agriculture interface for streams in the Interior Pacific Northwest (PNW), 2) present a case-study examining the relationship between flood irrigation and in-stream temperature, and 3) make the case that stream temperature and non-point source water quality issues are complex problems that are best addressed by considering variation of the problem in both time and space.

1.1 Modern agriculture and water quality of free-flowing streams in the Pacific Northwest

Excessive erosion associated with row-crop agriculture has reduced water quality over much of the PNW, particularly in the region dominated by the Columbia Basin and Columbia Plateau of Eastern Washington, North-central Oregon, and Northern Idaho (Schillinger et al., 2010). Similarly, pathogens and nutrients from agricultural operations can and have disrupted PNW stream ecology. Pathogens now rank second and nutrients fourth on the list of top pollutants in U.S. water bodies and agricultural lands are a significant contributor to both pathogen and nutrient loading (Richter et al., 1997; Parajuli et al., 2008; USEPA, 2009). Nutrient by-products from agricultural operations, particularly nitrogen, can cause eutrophication of stream systems and associated reduction of dissolved oxygen and reduced biodiversity (Vitousek et al., 1997). From a management standpoint the issues of sediment, nutrient and pathogen contaminants have collectively been addressed using stream-side vegetation buffer strips in an attempt to attenuate pollutant entry into stream systems (Castelle et al., 1994; Schmitt et al., 1999; Dosskey 2002; Dorioz et al., 2006; Mayer et al., 2007). Vegetation buffers have been effective at reducing nutrient (Yates and Sheridan, 1983; Lowrance et al., 1985), pathogen (Tate et al., 2004; Knox et al., 2007) and sediment loading (Lyons et al., 2000; Lee et al., 2003) in streams. However, development of effective policy concerning the use of buffer strips has been complicated by the fact that the efficacy of buffers in reducing pollutant loading varies strongly in accordance with a number of design and environmental factors including buffer width, cover and height of plant material, slope, and soil attributes (Pearce et al., 1997, Atwill et al., 2005, George et al., 2011). For example, a review by Dorioz et al. (2006) reported variation in sediment retention by vegetation buffers of 2.5 orders of magnitude across studies. Ultimately, the use of riparian buffer strips will need to be paired with spatially explicit models (e.g., Tim and Jolly, 1994) to assist managers in integrating land-use practices with conservation measures to reduce pollutant yield at the watershed scale.

Livestock operations can have both direct and indirect effects on water quality of PNW stream systems. Direct effects can include increased nutrient and pathogen loading through fecal or urine additions (Nader et al., 1998; George et al. 2011); excessive nutrient loading can stimulate algal blooms leading to reduced dissolved oxygen concentrations (Belsky et al., 1999). Additionally, increased turbidity associated with hoof action can negatively impact aquatic organisms by decreasing primary production (Henley et al., 2000; Line, 2003). Strategic location of livestock attractants (e.g., salt, artificial water sources) can reduce livestock densities near riparian areas and associated nutrient inputs into stream systems (Tate et al., 2003; George et al. 2011). Indirect effects of livestock on water quality include impacts to streamside plant communities and physical damage to streambanks. Riparian vegetation plays a critical role in sustaining the biotic integrity of the stream ecosystem by anchoring bank material in place during high flow events (Kleinfelder et al., 1992; Clary and Leininger, 2000). Livestock grazing of this resource is generally sustainable under conditions of moderate utilization (e.g., graze to 10cm stubble height; Boyd and Svejcar, 2004; Volesky et al., 2011), but timing of grazing can also influence livestock impacts on water quality and riparian vegetation (Boyd and Svejcar, 2004). Severe utilization of streamside plants by livestock can lead to reduced plant biomass and altered stream channel conditions (e.g., wider and more shallow channel) that are associated with decreased water storage capacity, reduced filtration effects of vegetation buffers, and increased water temperature (Kauffman and Kruger, 1984; Winward, 1994; Toledo and Kauffman, 2001).

1.2 Agricultural impacts on stream temperature

Recently, concern has developed regarding summer in-stream temperature dynamics and agricultural practices that may be associated with elevated water temperature. Water temperature is an important attribute of the aquatic environment that has the potential to affect basic ecological processes such as nutrient cycling and can also modulate biotic ecology (Poole and Berman, 2001; Isaak and Hubert, 2001). Water temperature is a key driver of invertebrate demography and impacts fish species via temperature-dependent fluxes in dissolved oxygen content (Young and Huryn, 1998). In the Mountain and Northwest United States, concern over water temperature extremes and their impact on red band (*Oncorhynchus mykiss newberri*) and bull (*Salvelinus confluentus*) trout is an important driver of regulatory mechanisms governing agronomic and other land use practices.

Much of the interest surrounding the influence of agriculture on water temperature has focused on practices that reduce woody plants in the riparian area. A growing body of empirical evidence suggests that shade from woody plants can reduce daily maximum water temperature (Poole and Berman, 2001; Tate et al., 2005). Livestock grazing (as well as wildlife-associated herbivory) can reduce woody plant cover, particular when grazing occurs subsequent to the senescence of herbaceous vegetation (Holland et al., 2005; Clary et al., 1996; Matney et al., 2005). However, shade from woody plants is only one of a myriad of factor, including air mass characteristics, elevation, stream flow, stream gradient, adiabatic rate, channel width/depth, and groundwater inputs that can influence water temperature maxima in streams (Larson and Larson, 1996; George et al., IN PRESS). Diversion of stream water for purposes of irrigating agricultural crops can potentially impact a number of these processes including stream flow and groundwater dynamics.

2. Flood irrigation impacts on stream temperature: A case study

Flood irrigation is a common agricultural practice on interior PNW meadows used for hay and/or livestock forage. These meadows are typically low-to-mid elevation and contain, or are influenced by, a seasonal or perennial natural stream system. Water from the stream is diverted at the upstream end of the meadow into smaller irrigation ditches that parallel the stream at a distance on one or both sides. Water then disperses from the irrigation ditch across the meadow either passively through sub-surface flow, or actively via overland spillage from the irrigation ditch. Specific water management practices vary by and within drainage, however, the irrigation season generally begins in spring following the onset of mountain snowmelt. Efficiency and control of water usage is obviously low with flood irrigation, but it remains a popular form of irrigation due to low costs associated with the absence of power (i.e., electric or fossil fuel) and few major infrastructural requirements.

Excessive in-stream water temperature is the most frequent water quality impairment for streams in southeastern Oregon (Oregon Department of Environmental Quality 2002). Stream temperature regulations are associated mainly with concern over thermal requirements for cold water fish stocks (Boyd and Strudevant 1996, Gamperl et. al. 2002).

Many streams in this region lack or have minimal amounts of woody cover (i.e. shade); under such conditions, fluctuations in water temperature are strongly influenced by parallel fluctuations in air temperature (Stefan and Preud'homme 1993, McRae and Edwards 1994). Flood irrigation can potentially increase in-stream temperature through two mechanisms: 1) decreases in in-stream discharge, and 2) overland flow of warmer flood waters re-entering the stream. Conversely, flood irrigation could act to cool in-stream water temperature if sufficient flood water returns to the stream via groundwater input.

Previous research has suggested that flood irrigation may act to moderate summer in-stream temperature extremes by elevating the groundwater table in the surrounding meadow and increasing groundwater inputs into the stream (Stringham et al. 1998). If such a moderating effect were to occur, one prediction would be that daily in-stream temperature maximums would be less sensitive to the controlling influence of air temperature. Our objective in this case-study was to determine the impact of flood irrigation on seasonal temperature dynamics of a meadow stream in southeastern Oregon. Specifically, we characterized depth to groundwater and discharge patterns in irrigated and non-irrigated years, and tested the hypothesis that air temperature would have less effect on stream temperature in an irrigated compared to non-irrigated stream reach.

2.1 Study site

Our study took place in the Lake Creek drainage in Grant Co., OR, U.S.A. (11T0370683 UTM4890874) at an elevation of approximately 1500 m. Lake Creek is a perennial Rosgen C class (Rosgen 1994) stream. Its position near the base of the Strawberry Mountains causes strong seasonal discharge fluctuations associated with snowmelt in April through early June. The local area receives approximately 330 mm of annual precipitation, most falling as snow during the winter months; maximum air temperatures (approx. 28^{o}C) occur in late July (Oregon Climate Service 2005). Soils were sandy clay loam underlain by a clay lens at depths ranging from 75 – 90 cm. We established a 3.5 km upper study reach and a 1.0 km lower study reach approximately 1.3 km downstream from the upper reach (Figure 1). The upper reach was within the irrigated portion of the meadow and the lower reach was not. Two tributaries entered the main channel of Lake Creek within the irrigated portion of the meadow. The beginning of the upper study reach was immediately downstream from the farthest downstream tributary. No tributaries entered within the lower study reach. Shade from woody plants was non-existent in either reach but the lower reach had limited topographic shading.

The meadow surrounding Lake Creek is seasonally flooded by two irrigation ditches flowing north to south along the east and west sides of the meadow (Figure 1). The West Ditch (0.08 m^3/sec^{-1}) and East Ditch (0.06 m^3/sec^{-1}) diversions were opened in 2004 from April 13 to June 30 and from April 21 to June 30 in 2005. Approximately 70% of the flow in West Ditch #1 was diverted into West Ditch #2. A lateral slope downward from the irrigation ditches to streamside elevations allowed for subsurface flow from irrigation ditches to the surrounding meadow. Diversions within the West Ditch were used to augment flood irrigation from subsurface flow. Water spreading from the East Ditch relied entirely on subsurface flow from the irrigation ditch.

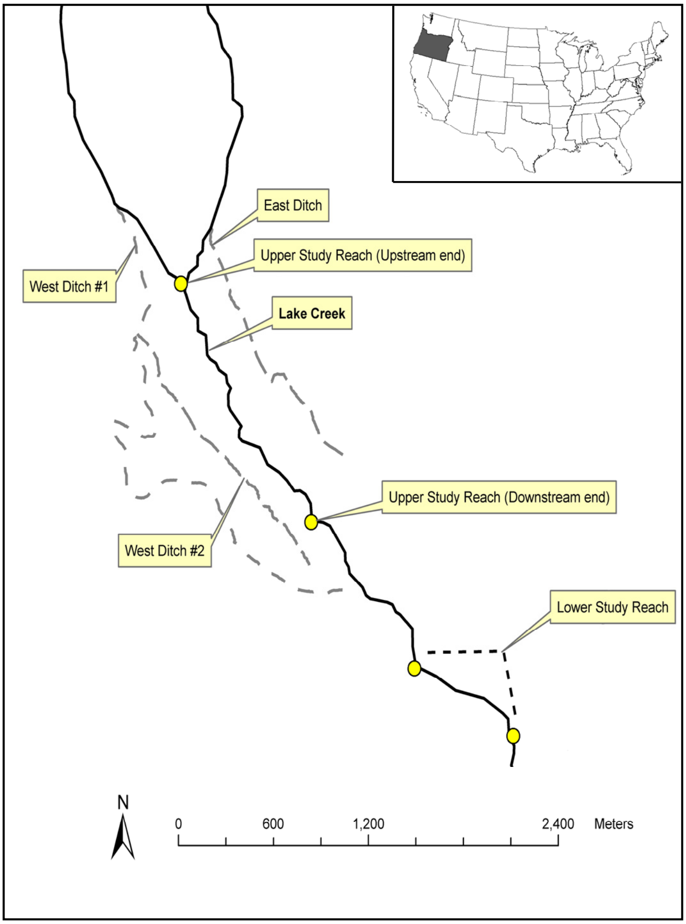

Fig. 1. Overview of study site on Lake Creek, southeast OR, U.S.A. depicting irrigation ditches and locations for upper and lower study reaches (inset shows location of Oregon within U.S.A.). The meadow surrounding the upper reach was flood irrigated from mid-April to June 30 of 2004-2005.

2.2 Methods and materials

Data for a non-irrigated year were collected in 2002 and for flood irrigated years in 2004 – 2005. We measured point-in-time stream discharge at approximately 10-day intervals in June and July using permanent main channel cross-sections located at the upstream ends of the upper and lower study reaches. Discharge was estimated using a magnetic-head pygmy flow meter with a top set wading rod and an Aquacalc 5000 discharge recorder (JBS Energy, Inc; West Sacramento, CA). Measurements were made at 60% of total depth from the stream water surface. Regression analysis was used to estimate mid-month values for June and July. Depth to groundwater within the upper reach was measured at 10-day intervals in June and July using shallow PVC wells spaced at 15-m intervals along 4 transects run perpendicular to the stream channel from streamside to 135 m distance. Wells were constructed of perforated 1.9-cm diameter plastic pipe buried to 150 cm depth, or restrictive layer, whichever came first. Gravel (0.5-cm diameter) was packed around the well to within 15 cm of the soil surface and the remaining hole filled with soil. Transects were located at the beginning and end of the upper reach and at 2 intermediate points. The east or west side of the creek was randomly selected for placement of each transect; 3 transects were located on the west side and 1 transect on the east side. In 2002 we used a laser level to measure the elevation of each well relative to the deepest portion of the adjacent stream channel. Raw groundwater depth values were modified by adding the elevation of the well to generate values indicating the elevation of the meadow water table relative to channel elevation. We used regression analysis to estimate mid-month depth values, by year and distance from stream, for June and July. Values were averaged for each date across the 4 transects and within distance from stream.

In-stream, air and groundwater temperature data were collected in June and July of all years. Daily maximum stream temperature was measured using thermistors placed at the beginning, end and two intermittent positions within the upper reach and at the beginning and end of the lower reach. Thermistors were programmed for hourly readings and maximum temperatures were identified by selecting the highest hourly reading for the warming portion of the daily thermograph. Maximum in-stream temperature values were averaged within day and reach. Daily maximum air temperatures were determined as per methods for in-stream values using four stream-side thermistors interspersed throughout the upper and lower reaches. Because daily air temperatures did not differ across thermistors, we averaged values within day. Daily maximum groundwater temperature was monitored using thermistors placed in 7.62-cm diameter plastic pipe wells located 10 m from the active channel at the beginning and end of the upper study reach. Maximum daily temperatures were determined as per methods for in-stream temperatures. Values were averaged monthly, across locations and within year.

Because air temperatures vary across years, and because of the strong influence of air temperature on water temperature (Stefan and Preud'homme 1993, McRae and Edwards 1994), inter-annual variation in air temperature can obscure treatment effects on water temperature. Thus, we compared upper and lower reaches within year. A common range of maximum daily air temperatures was selected across years and within month (June or July). These data were regressed (within study reach) with the corresponding days maximum stream temperature. Within a year and month, slope and intercept values were compared between study reaches using Statistical Analysis Software (PROC SYSLIN, SAS 1999). Regression equations for maximum air and water temperatures were then used to generate predicted maximum water temperatures using air temperature values that approximated the highest observed values within month and across years (June = 26^{o}C, July = 31^{o}C). All mean values are reported with their associated standard error.

2.3 Results and discussion

In-stream discharge dropped sharply from the spring run-off (June) to post run-off (July) periods and was generally less in June for the upper reach as compared to the lower reach (Table 1). Discharge was similar between the upper and lower reaches in July of all years. Between-year differences in June were associated with differing run-off patterns between years and diversion of water in the upper reach. Data for groundwater elevation indicate that by June 15 of the non-irrigated year (2002) the elevation of the water table at 15 m distance from the stream was slightly lower than that of the stream-side well, a gradient that suggests the potential for water losses to the surrounding meadow (Figure 2); by July of the non-irrigated year this downward slope continued to 75 m distance from the stream.

	June 15		July 15	
	Upper reach	Lower reach	Upper reach	Lower reach
2002	0.75 +/-0.01	0.76 +/- 0.01	0.12 +/- 0.00	0.12 +/- 0.01
2004	0.67 +/- 0.02	0.76 +/- 0.01	0.16 +/- 0.01	0.16 +/- 0.00
2005	0.49 +/- 0.01	0.62 +/- 0.02	0.19 +/- 0.01	0.21 +/- 0.01

Table 1. Seasonal discharge (m^3/sec^{-1}) for upper and lower study reaches on Lake Creek, OR, U.S.A. Data from 2002 represent a non-irrigated year. The meadow surrounding the upper reach was flood irrigated from mid-April to June 30 of 2004-2005.

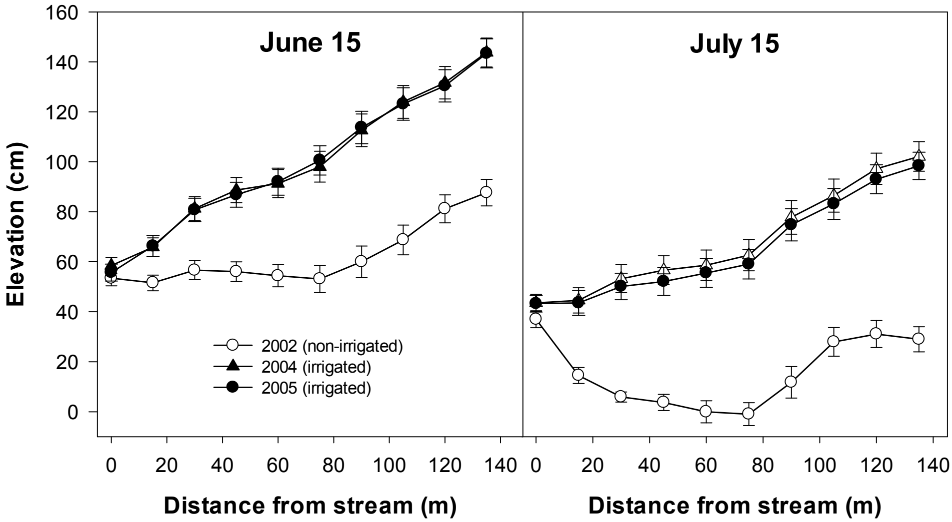

Fig. 2. Mean groundwater elevation and associated standard errors for the upper study reach as a function of distance from stream for Lake Creek, OR, U.S.A. Values represent the height for the surface of the groundwater table relative to the channel thalweg. The meadow surrounding the upper reach was flood irrigated from mid-April to June 30 of 2004-2005.

In contrast, during irrigated years the water table elevation had a positive slope with distance from stream until mid-July and this positive slope was maintained to 135 m distance from the stream. In irrigated years groundwater elevation dropped up to 50 cm (depending on distance from stream) after irrigation shut-off. A similar drop was noted in the non-irrigated year, but the absolute value of groundwater elevations after irrigation shut-off was much less as compared to the irrigated years (Figure 2). These data suggested a higher meadow water table with irrigation and a greater potential for groundwater inputs into the stream. Maximum daily groundwater temperatures increased from June to July and ranged from approximately 9 to 12°C across years and months (Figure 3). Increasing groundwater temperature in July was probably associated with increasing air temperature (Ward 1985). Groundwater temperatures were generally 8 to 14°C cooler than in-stream temperature maxima suggesting that any groundwater return flow into the main channel would have a buffering effect on maximum stream temperatures.

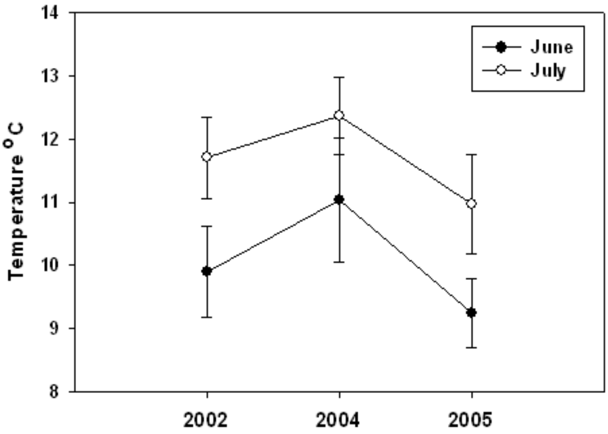

Fig. 3. Daily maximum groundwater temperature means and associated standard errors for the upper reach of Lake Creek, OR, U.S.A. The meadow surrounding the upper reach was flood irrigated from mid-April to June 30 of 2004-2005.

Maximum daily air temperature was associated positively with maximum daily water temperature in both irrigated and non-irrigated years (Figure 4). R^2 values ranged from 0.55 to 0.81, with the exception of July, 2002 which had values of 0.30 and 0.32 for the upper and lower reaches, respectively (Table 2). Explanatory power during this time period was decreased by several outlying points that were associated with cool nights followed by warm, but overcast days. If these days are excluded (n = 3), R^2 values for the upper and lower reaches increase to 0.48. In the non-irrigated year (2002) maximum daily water temperature in both study reaches responded similarly to air temperature although the intercept was slightly higher for the lower reach in June (P = 0.003, Figure 4). In the irrigated years intercepts differed (P < 0.001) between the upper and lower reaches and slopes were different (P < 0.071) for all but July of 2004 (P = 0.159, Figure 4). At a given maximum air temperature, water temperature was slightly less in the upper reach suggesting that flood irrigation helped moderate maximum daily water temperature. Although the y intercept was slightly higher for the lower reach in June of the non-irrigated year (2002), the lines of best fit for irrigated and non-irrigated reaches converge at higher air temperatures. In contrast, lines of best fit for the study reaches diverge at higher air temperatures during irrigated years (2004-2005), providing further evidence for an irrigation-related buffering effect.

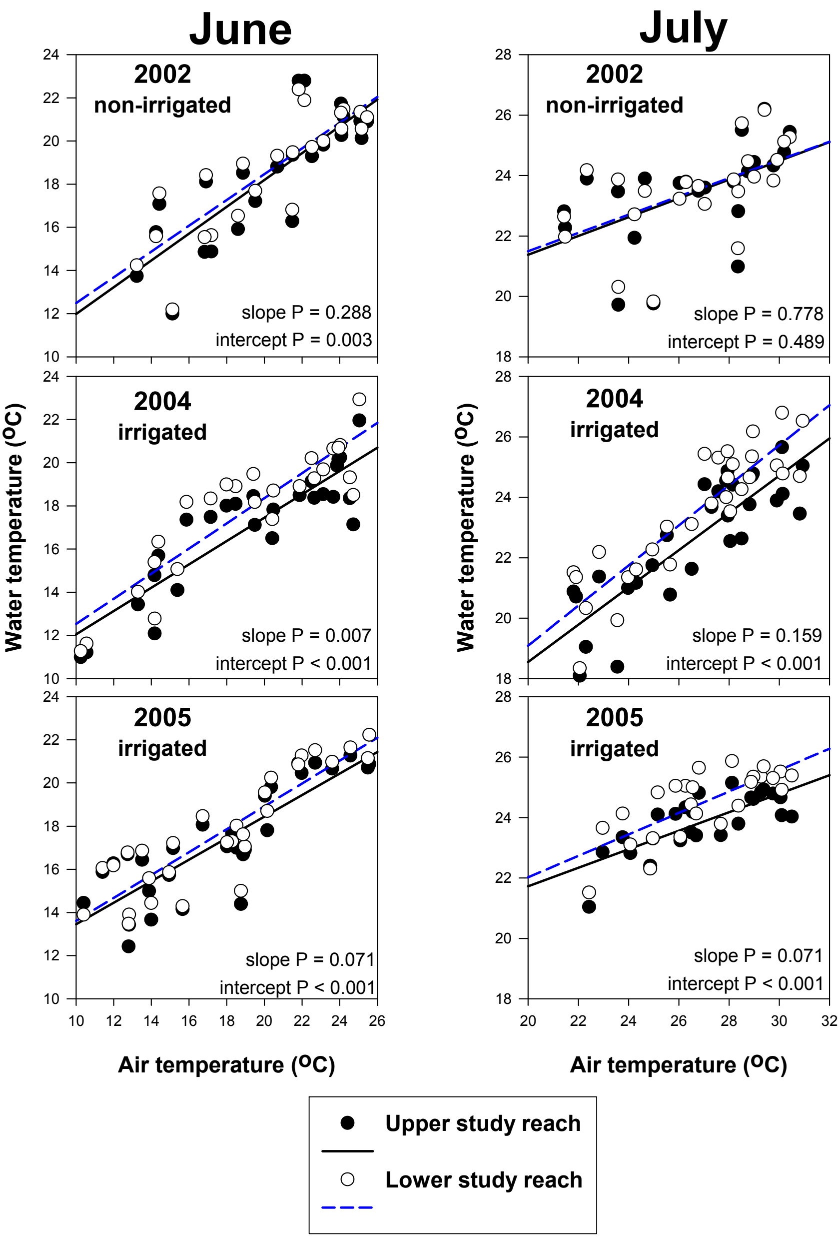

Fig. 4. The relationship between daily maximum air and stream temperature values for June and July within the upper and lower study reaches of Lake Creek, OR, U.S.A. The meadow surrounding the upper reach was flood irrigated from mid-April to June 30 of 2004-2005. Probability values for differences in slope and intercept values between irrigated and non-irrigated reaches are reported within graphs.

Year	Month	Reach	Slope	y intercept	R^2	Predicted maximum water temperature °C
2002	June	Upper	0.62	5.77	0.66	21.92
		Lower	0.60	6.51	0.71	22.03
2002	July	Upper	0.31	15.16	0.3	24.80
		Lower	0.30	15.45	0.32	24.82
2004	June	Upper	0.54	6.64	0.79	20.71
		Lower	0.58	6.71	0.81	21.84
2004	July	Upper	0.62	6.20	0.73	25.33
		Lower	0.66	5.82	0.79	26.37
2005	June	Upper	0.50	8.47	0.72	21.44
		Lower	0.53	8.31	0.78	22.11
2005	July	Upper	0.31	15.60	0.57	25.09
		Lower	0.36	14.93	0.55	25.93

Table 2**.** Predicted monthly maximum water temperature values for the upper and lower study reaches of Lake Creek, OR, U.S.A. The meadow surrounding the upper reach was flood irrigated from mid-April to June 30 of 2004-2005. Equations were in the form of *max water* = *max air(x)* + *b* and were generated for each year/month/location combination based on data for regression equations in Figure 4. Air temperatures of 26°C and 31°C were used to approximate observed maximum temperatures for June and July, respectively.

Without irrigation (2002) predicted maximum water temperatures in the 2 reaches were very similar. However, with irrigation (2004 and 2005) there was divergence with cooler temperatures in the upper reach. For irrigated years predicted water temperature maximums were approximately 0.9°C warmer in the lower reach as compared to the upper reach (Table 2). Temperature reduction in irrigated years was recorded under varying discharge levels including the late snow-melt run-off period in June and lower flow conditions in July (Table 1). Stringham et al. (1998) reported maximum stream temperatures of 1 to 3°C cooler for a flood irrigated vs. non-irrigated reach. The relatively smaller thermal response in our study may be associated with the presence of old irrigation ditches between the currently used ditches (Figure 1) and the active stream channel. These ditches intercepted some of the sub-surface flow and provided additional water surface area for evaporative losses. Our results are in contrast to those of Tate et al. (2005) who reported increased stream temperatures with flood irrigation. This discrepancy may be explained by the relatively low percentage of total stream discharge (about 20%) that we diverted for irrigation as opposed to that of Tate et al. (30 to 70%).

In summary, we found that air temperature exerted a measurable and strong influence on water temperature maximums, explaining up to 80% of the variation in this variable (Table 2). For the stream we studied, flood irrigation appeared to moderate the influence of air temperature on daily maximum water temperature; this effect was observed during irrigation and continued for at least 1 month after cessation of irrigation. Our results agree with earlier work from Stringham et al. (1998) and suggest that flood irrigation can help buffer daily maximum water temperature during summer air temperature extremes. In the present study, the magnitude of the predicted temperature reduction was generally < 1^{o}C. However, stream size, percentage of water diverted, and proportion of surface/subsurface flows may all influence the effect of flood irrigation on water temperature.

3. Stream temperature as a complex problem

Boyd and Svejcar (2009) presented the notion that those problems facing natural resources managers today differ, in fundamental ways, when compared to those of previous generations. Specifically these authors argued that many of the issues facing managers today are "complex", in that the environmental factors associated with these issues vary in both space and time, making it difficult to generalize management prescriptions or characterize ecosystem responses. These same hindrances make it challenging to formulate biologically-realistic regulatory statues for stream temperature and other water quality parameters. In terrestrial systems, simultaneous consideration of space and time is difficult given that space has x, y and z (i.e., elevation) dimensions. Measurement of water temperature (and water quality in general) is somewhat different in that, practically speaking, space has only one dominant dimension (i.e., upstream or downstream). Thus the riparian ecosystem offers a unique opportunity for simultaneously considering environmental characteristics in space and time.

Our preceding examination of water temperature dynamics on Lake Creek is a good example of how variation in space and time can interact to in influence water quality parameters. Our approach in this case was to monitor water temperature over both space and time, in irrigated and non-irrigated years and to use this information to make inferences regarding the influence of flood irrigation on water temperature maxima. To further characterize variation in water temperature over both space and time, we plotted daily water temperature maximums for a non-irrigated year (2002) from June 1 (Julian day 152) to September 30 (Julian day 273; Figure 5). Data were interpolated using negative exponential smoothing and displayed in a contour plot (SigmaPlot 12.0; Systat Software Inc.; San Jose, CA) that allows the user to view change in temperature over time at a given point in the stream and change over space for a given day.

When viewed in this way, daily maximum water temperature is set within a background of strong intra-annual and spatial variation. From a management standpoint, these data imply that capturing variation in space and time should be an integral part of a water quality assessment strategy. For example, if we were to have measured temperature at one point within the irrigated reach (stream distance 0 – 3.5 km), and one location within the non-irrigated reach (3.5 to 5.9 km) then our conclusions regarding the influence of flood irrigation on stream temperature would have been strongly influenced by choice of sampling location, as compared to the multiple location sampling strategy that we employed. Similarly, our use of a two-month sampling window helped overcome undue influence of time at shorter sampling intervals. This same logic applies equally to determining compliance within a regulatory framework. For example, the Oregon Department of Environmental Quality stipulates that average daily maximum water temperature for cold-water fisheries should not exceed 20^{o}C for any seven-day period, and that number drops to 12°C for streams containing bull trout spawning and juvenile rearing habitat (ODEQ, 2008). Determination of compliance with either standard for the section of Lake Creek we studied would depend strongly on when and where samples were taken (see Figure 5). The spatial and temporal variability of maximum water temperature for Lake Creek could also have strong relevance to the ecology of aquatic organisms. For example, while water temperatures at a given point/time on Lake Creek may exceed tolerance levels of cold water fishes, variability in temperature over space may act to provide thermal refugia that allow affected species to escape potentially harmful temperatures (Ebersole et al., 2001).

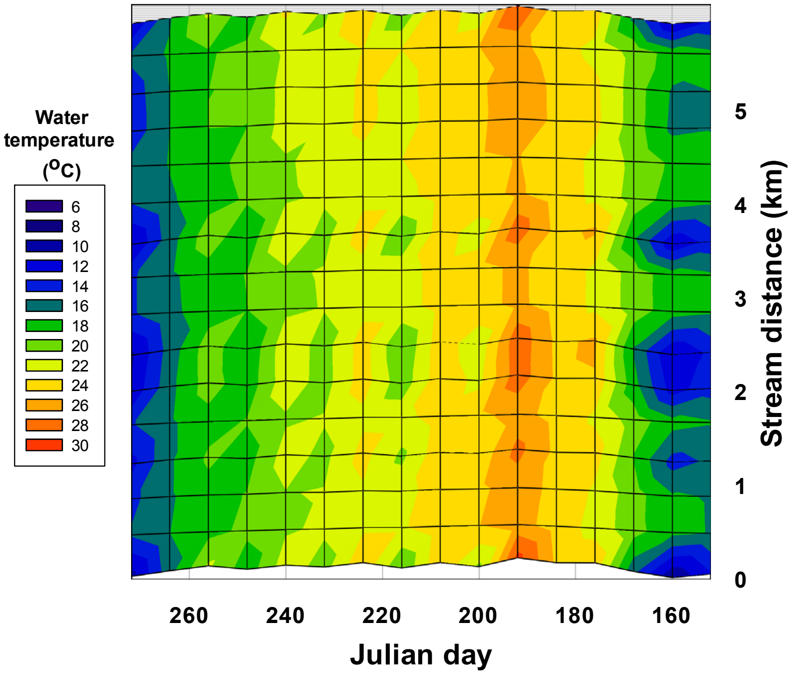

Fig. 5. Daily maximum water temperature values over space and time for Lake Creek, OR, U.S.A.

4. Conclusions

In conclusion, we suggest that complex problems such as water temperature management may not be solved with the broad, sweeping regulatory statues (e.g., the Clean Water Act and various amendments) historically used to solve more easily identifiable problems, but instead require an understanding of the inherent ecological uniqueness that defines complex problems on a case-by-case basis. We suggest that management and regulation of in-stream water temperature is a complex problem that will vary across both space and time and that natural resources professionals must recognize the dynamic nature of this relationship in designing management plans, regulatory policy, and future research. Such recognition is often at odds with contemporary policies and paradigms that focus attention on discreet temperature (or other water quality) values, thus creating tension between managers and regulators. Incorporating spatial and temporal variation into the concept of stream temperature will allow for a more representative characterization of water temperature regime and promote a more comprehensive understanding of the potential impacts of water temperature on the stream ecosystem and its inhabitants.

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