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A comparison of periphyton community structural and functional responses to heavy metals

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Abstract

This study evaluates the effectiveness of community production and respiration measurements as monitoring tools for environmental impact evaluations and compares these data to community structural data.

In Prickly Pear Creek, Montana, production and respiration rates were determined for periphyton communities in control, impact and recovery reaches using colonized granite substrates and sealed plexiglas chambers. Values for gross primary productivity (GPP), community respiration (CR_{24}), ash-free dry mass (AFDM) and chlorophyll *a* content (Chl*a*) were obtained for each granite slab. Of these, AFDM, Chl*a* and CR_{24} were statistically significant among sites ($P \leq 0.01$). Although mean values for GPP appeared to differ among reaches, statistical differences could not be inferred because of large variances associated with this measure. These data indicate that inherent variability may limit the use of community function measures in routine environmental monitoring. However, production/respiration methods provide valuable data about emergent properties of aquatic communities that cannot be derived from routine population censuses.

Introduction

The effects of chemicals or pollutants on aquatic ecosystems have been studied extensively over the last several decades (Warren, 1971; Bates & Weber, 1981; Sheehan, 1984). However, there remains considerable doubt regarding the capacity of aquatic ecosystems to withstand environmental perturbations and the biological mechanisms involved. Two major problems confront investigations in this area. First, how does one measure a 'change' in a system as complex and dynamic as an aquatic ecosystem when spatial and temporal variances associated with such systems are often high? This is especially true for contaminants which may exert only subtle effects that are difficult to detect at the levels of resolution afforded the community or ecosystem ecologist. A second and perhaps more difficult problem is interpreting the significance of changes observed in an aquatic system following the addition of chemical pollutants or other anthropogenic perturbations. The fact that an ecosystem responds to environmental stresses is certainly well documented. However, the capacity of a system to absorb and/or respond to these changes in less well understood. Change, variability, stimulus and response are part of the dynamic nature of aquatic ecosystems and, while difficult to measure, these attributes are certainly responsible for the maintenance of these systems through time. Therefore, it becomes imperative not

only to measure changes caused by an environmental perturbation but to analyze what these changes mean within the context of a living biological system.

This study is an attempt to address these two questions. The first objective was to examine two methods currently being used to detect the effect of environmental perturbations on an aquatic ecosystem. Periphyton community function (photosynthesis and respiration) and community structure were measured simultaneously in order to compare them as indicators of environmental impact due to heavy metals contamination in Prickly Pear Creek, Montana. The second objective was to evaluate the potential of these two methods (community structure and community function) for indicating the significance that a change in the measured parameters might have on the periphyton community of Prickly Pear Creek. Using these two methods as a basis for comparison we discuss the statistical sensitivity, routine biomonitoring potential and ecological significance of the measured variables. Finally, we discuss how and to what extent the heavy metals in Prickly Pear Creek are affecting the periphyton community.

Study area

The study area encompassed a 20-km reach of Prickly Pear Creek 32 km southwest of Helena, Montana (Fig. 1). The Prickly Pear drainage includes the Elkhorn Mountains and the Colorado Mining District and forms part of the headwaters of the Missouri River.

Spring Creek, which joins Prickly Pear Creek at Jefferson City, Montana, (Fig. 1) is contaminated with high concentrations of cadmium, copper, lead, silver and zinc, which are being leached from waste piles associated with mining, milling and smelting operations of the late 1800's (La Point *et al.*, 1983; MWQB, 1981). Prospecting was so intense in this area that mining prospects, abandoned mines and mine tailings are the prominent features in the hills along the Spring Creek drainage. Due to the heavy metals carried by Spring Creek, this area has been the site of intensive studies conducted by the U.S. Environmental Protection Agency (EPA) and the Montana State Water Quality Bureau (La Point *et al.*, 1983; Miller *et al.*, 1985; MWQB 1981).

Three sites on Prickly Pear Creek (control, impact and recovery) were used in this study (Fig. 1). The study reaches are composed of unshaded riffles with gravel and cobble substrates. These sites correspond to three of the eight sites used in the 1982 EPA Prickly Pear Creek study (La Point *et al.*, 1983) and were chosen to minimize differences in abiotic factors shown to be important in periphyton colonization studies (Weber & MacFarland, 1981). Flow velocities at each site ranged from 16 to 20 cm sec^{-1} and water depths ranged from 15 to 30 cm. Typical mean discharges for Prickly Pear Creek and Spring Creek are 0.5 m^3 sec^{-1} and 0.16 m^3 sec^{-1}, respectively, although these discharges can triple during a summer storm (MWQB, 1981). The control site was located 1.5 km upstream from the confluence of Spring Creek; the impact and recovery sites were located 0.15 km and 18 km downstream from Spring Creek, respectively.

The Montana Water Quality Bureau has listed Spring Creek as 'the most severe water pollution problem in the Prickly Pear drainage' (MWQB, 1981). Chemical analyses of water samples taken from Prickly Pear Creek over the last several years are compiled elsewhere (La Point *et al.*, 1983, Miller *et al.*, 1985, MWQB 1981). These data indicate that although the absolute concentrations of metals vary seasonally and yearly, it is reasonable to assume that since the mining period at the turn of the century, Spring Creek has acted as a point source for heavy metals entering Prickly Pear Creek. In addition, instream concentrations for certain metals of concern in Prickly Pear Creek below Spring Creek have routinely exceeded U.S. EPA Water Quality Standards (Miller *et al.*, 1985). Water samples taken immediately prior to the production-respiration studies indicate that copper concentrations in control, impact and recovery reaches were 14, 25 and 12 $\mu g\ l^{-1}$, and zinc concentrations were 31, 1238 and 100 $\mu g\ l^{-1}$ (Table 1).

Materials and methods

Study Design

On June 9, 1982, 50 colonization substrates were placed into Prickly Pear Creek, 26 in the control

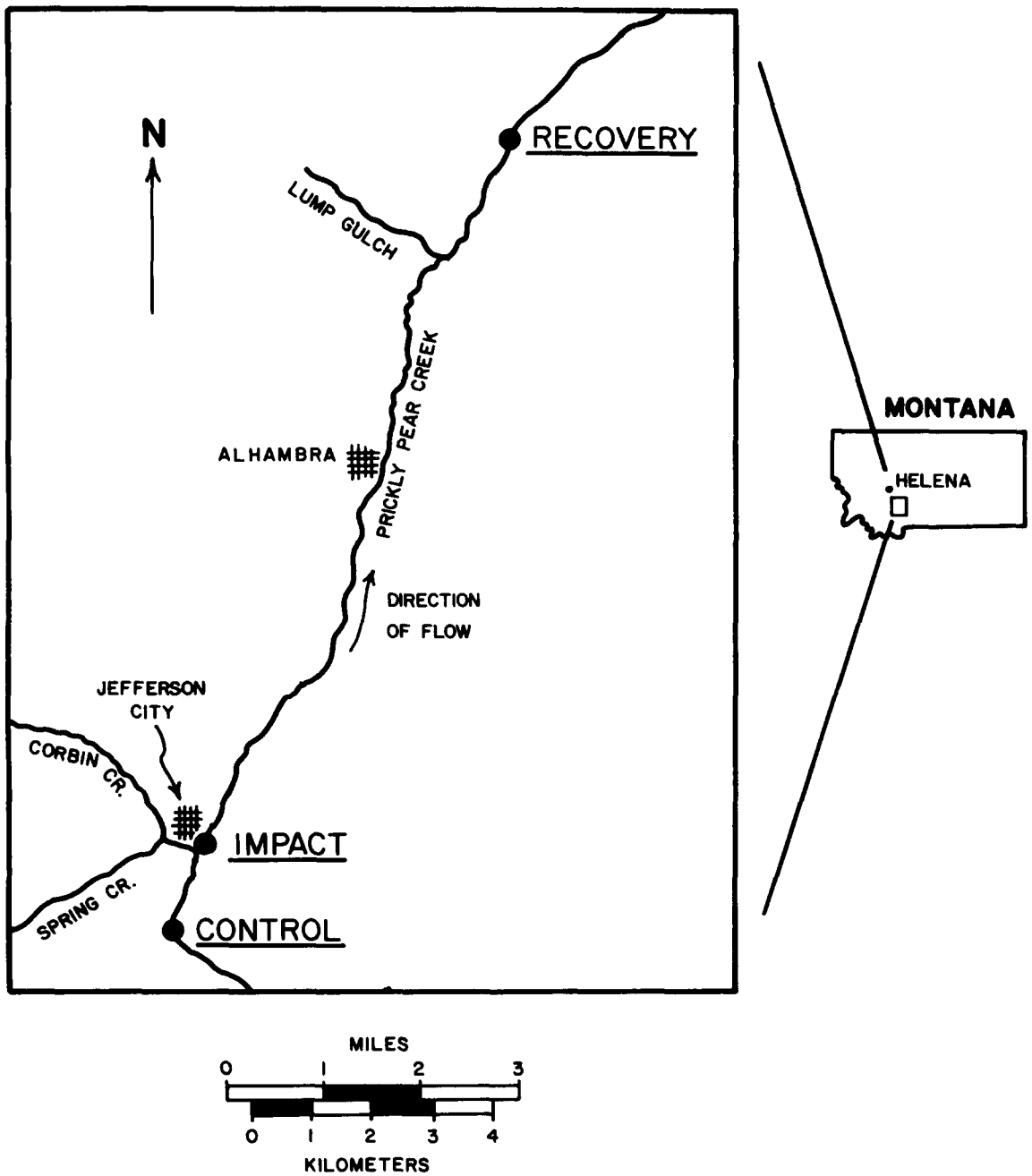

Fig. 1. Map of Prickly Pear Creek drainage, Montana. Control, impact and recovery reaches indicate areas where granite slabs were placed for the 66 day colonization period. For the purposes of this study, Spring Creek was treated as a point source for heavy metals entering Prickly Pear Creek.

reach, 12 in the impact reach and 12 in the recovery reach. Granite slabs, 8 cm×10 cm, were used as the colonization substrates. Production and respiration measurements were conducted over a period of 10 days beginning on August 14, 1982, after 66 days of colonization. The number of replicate slabs used to determine any given parameter vary slightly; however, in general the control values are based on 11

Table 1. Mean dissolved metal concentrations in control, impact and recovery reaches in Prickly Pear Creek, MT, July 1982. Values are means ± standard deviations of six samples (from La Point *et al.*, 1983).

Metal, μg/L	Reach		
	Control	Impact	Recovery
Cadmium	8.5± 9.9	17.5±14.8	7.7± 8.8
Copper	14.3± 9.4	25.3± 7.5	12.5± 8.0
Nickel	11.8±11.4	16.5±21.8	7.8±14.1
Zinc	31.5± 6.8	1238 ±99	100.8± 7.4

replicate substrates, and the impact and recovery values are based on eight and six replicates respectively (discrepancies due to missing data are noted in the figures and figure captions).

Production-respiration chambers

Plexiglas chambers used in this study were modified from those used by other investigators (Maki & Johnson, 1976; Bott *et al.*, 1978; Bott *et al.*, 1985). These sealed production-respiration chambers enabled us to isolate colonized granite substrates from the rest of the stream benthos, and to measure the oxygen production and consumption of this community during the test period. Each chamber (Fig. 2) consisted of a double-walled cylinder, 15 cm high and 25 cm in diameter (7.7-liter volume). The inner wall had 0.5 cm holes drilled through it to act as a baffle, distributing the flow of water evenly through the chamber. The outer cylinder had two 2.5 cm O.D. ports, which were connected using a 3.5 cm O.D. hose to a submersible pump. Flow velocity across the colonized granite substrate within the chamber was maintained at 16 cm sec^{-1}. Dissolved oxygen (DO) was monitored continuously with an in-line polarographic DO probe and meter (Yellow Springs Instrument Co., Model 54A). Prior to production and respiration measurements, substrates were removed from the river, placed into a basin containing control river water, and immediately transported to the control reach where production-respiration measurements were taken. All production-respiration measurements were made in the control reach using control water. In this way a better control over the abiotic conditions affected by chamber placement was maintained. However, these procedures did allow the possibility of routine chemical differences in test waters to mask the effect of differences in heavy metals concentrations. In order to assure that nutrient changes were not responsible for differences in metabolism we compared the general chemical composition of waters taken from the three reaches. Table 2 indicates that the ranges of nutrient concentrations overlap for all three reaches as do the

Table 2. Nutrient concentrations in control, impact and recovery reaches in Prickly Pear Creek, MT, July 1982. Values are means ± standard deviations; sample size is in parentheses.

Parameter	Reach		
	Control	Impact	Recovery
Total organic carbon, mg/L	7.8 ±0.64 (n=2)	2.8 ±0.42 (n=2)	ND[a]
NH_3-nitrogen, mg/L	0.159±0.235 (n=6)	0.23 ±0.20 (n=12)	0.14 ±0.23 (n=18)
NO_2-NO_3, mg/L	0.38 ±0.55 (n=6)	0.59 ±0.19 (n=12)	0.53 ±0.39 (n=17)
Ortho-phosphorus, mg/L	0.007±0.005 (n=6)	0.004±0.003 (n=12)	0.009±0.002 (n=17)
Total phosphorus, mg/L	0.05 ±0.002 (n=6)	0.05 ±0.007 (n=12)	0.05 ±0.001 (n=16)
pH, S.U.	7.29 ±0.06 (n=5)	7.66 ±0.20 (n=10)	7.60 ±0.19 (n=15)

[a] No data.

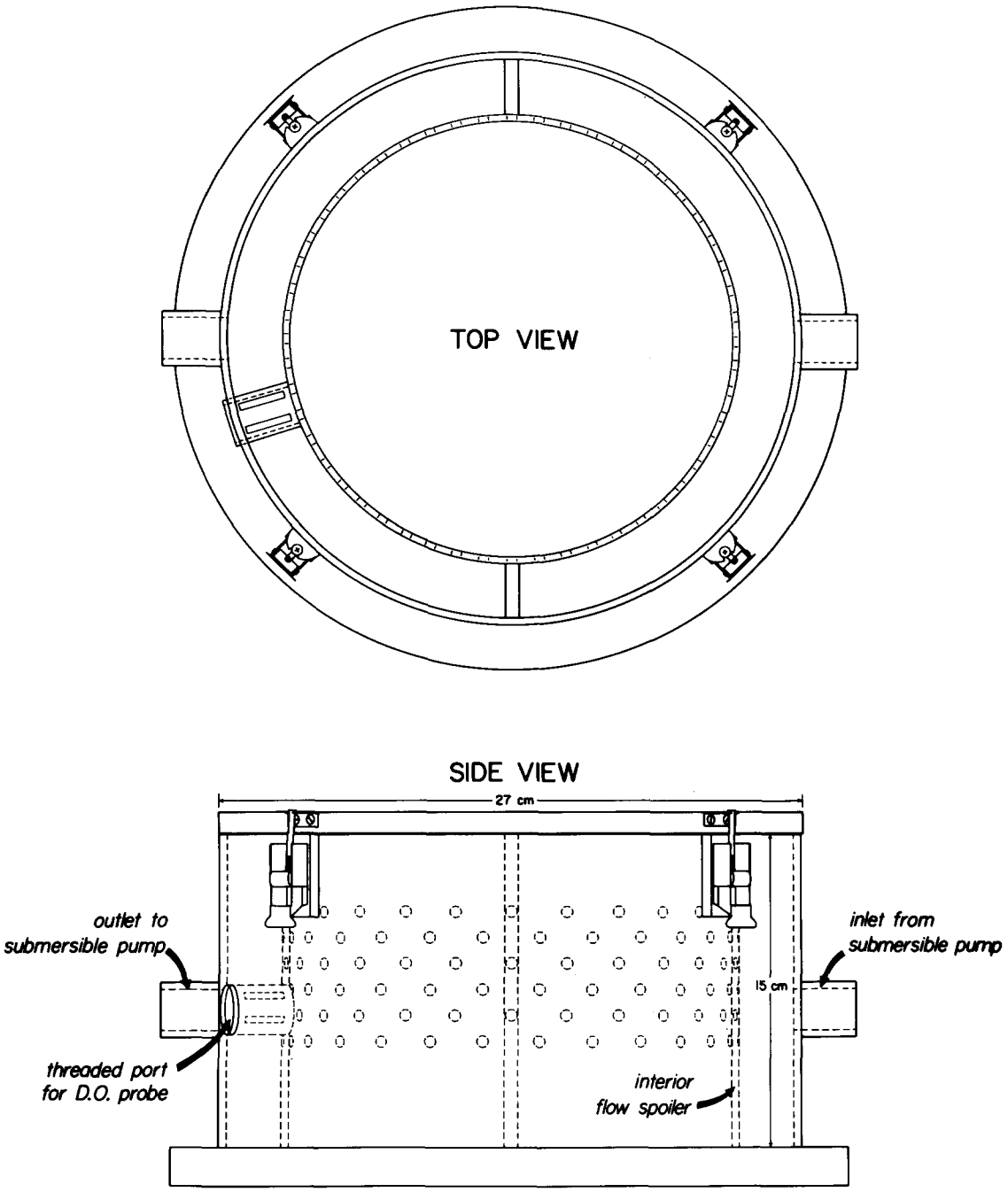

Fig. 2. Diagram of production-respiration chamber used in Prickly Pear Creek. Inlett and outlet ports connected to inline submersible pump. Volume of chamber when sealed is 7.5 L.

major cations and anions.

In the control reach, three chambers were placed into the stream such that, when the chambers were sealed the lids were 1 to 2 cm below the surface. We recorded the initial and final DO, temperature and duration of experimental period (usually 0.5 to 2 h). If gas supersaturation occurred, noted by bubble formation within the chamber, or if the measured

DO was greater than the calculated 100% saturation value, the chambers were opened and flushed with stream water to reinitiate the run. Periodic flushing with stream water also avoided CO_2 and nutrient depletion.

To measure community respiration we recorded the decline in DO within the chambers at night or in chambers covered by black plastic during the day. During these experiments, the chambers were also flushed periodically with fresh stream water. Time, temperature and DO were recorded as in the production experiments. The respiration values reported below are both daytime (dark chamber) and nighttime measurements. No clear differences could be discerned between day and night respiration readings despite the change in stream temperature (ca. 10 °C night and ca. 13 °C daytime). This could be due to the fact that the change in temperature had no effect or the noise inherent in the test system was greater than the effect due to temperature.

For production measurements, we measured the ambient photon flux using a Licor LI-500 integrator with an LI-1025 underwater quantum light sensor. The sensor was submerged in the stream adjacent to the three chambers. This provided time-averaged quantum light measurement during each experimental period. Substrate net oxygen production was measured until a predetermined amount of total light quanta was received. Hence, we could compare production data from different days. The units of light energy measured by this instrument were converted to 'photosynthetically active radiation' (PAR), which is the amount of light as μEinsteins m^{-2} in the 400–700 nm wavelength range (Westlake, 1980).

Following each production and respiration measurement, substrates were scrubbed with a coarse toothbrush and washed with distilled water into a plastic bowl to collect all material that had accumulated on each substrate during the colonization period. This material was then filtered onto preashed and preweighed glass-fiber filters (Whatman GF/F). Each filter was placed into a separate petri dish, wrapped in aluminum foil and frozen. Chlorophyll *a* (Chl*a*) and ash-free dry mass (AFDM) were determined for each sample upon return to the laboratory.

Chlorophyll *a* was determined using a modified high performance liquid chromatography (HPLC) method (after Brown *et al.*, 1981). This method separates chlorophylls and phaeophytins, and Chl*a* concentration is determined using purified standards. Glass-fiber filters containing the scraped material were placed into aluminum foil-covered 20 ml scintillation vials. An extraction solvent (20 ml of 90% HPLC-grade acetone saturated with $MgCO_3$; Vollenweider, 1974) was added, and each vial was placed into a sonic bath for 3 minutes. Sample vials were then placed into a freezer for a 24 h extraction period.

Following extraction, 50 μl aliquots were injected into the HPLC for Chl*a* determination. Chromatography for this procedure includes an isocratic separation using a *C18* reverse-phase column (Waters Associates), 2 ml min^{-1} flow rate with 75% methanol/25% acetone solvent mixture, detection at 405 nm with automatic peak integration (Varian Model CDS 111 integrator). Chlorophyll *a* was quantified using purified standards prepared in the extraction solvent (Sigma Chemical Corporation).

Following extraction, the glass filters were removed from the scintillation vials and placed into petri dishes to air dry. Any extraction solvent containing displaced filtered material was dripped onto the filter and allowed to air dry. All filters were then dried at 55 °C overnight, allowed to cool for 12 h in a dessicator and weighed to the nearest 0.1 mg. Filters were then ashed in a muffle furnace at 500 °C for 6 h, cooled to room temperature, wetted with distilled water and dried again at 55 °C overnight (to correct for dehydration of clays). Ash-free dry mass, calculated as the weight of the filter and ash subtracted from the dry weight, provided an estimate of the total biomass (autotrophic, heterotrophic and detrital) of the Aufwuchs community.

Calculations and statistical analyses

Measured DO values were converted to gross primary production (GPP) by multiplying the net hourly oxygen production (gm O_2 m^{-2} h^{-1}) by the length of the photoperiod and adding to it the photoperiod respiration. Gross primary production is

defined as the total oxygen produced on a daily basis as gm O_2 m^{-2} d^{-1} (Bott *et al.*, 1985). Net community respiration was converted to total daily community respiration (CR_{24}) by multiplying the hourly respiration rates by 24 hours. Twenty-four hour community respiration is the total amount of oxygen consumed on a daily basis as gm O_2 m^{-2} d^{-1}. Ash free dry mass (AFDM), Chl*a*, GPP and CR_{24}, were used to compute four indices of community autotrophy and photosynthetic activity:

Assimilation Ratio (*AR*) The net hourly primary production (HPP) divided by the amount of Chl*a* (mg O_2 mg^{-1} Chl*a* h^{-1}). This ratio is an index of photosynthetic oxygen production per unit plant material (Bott *et al.*, 1985; Marker, 1976). Larger AR values indicate relatively more productive photosynthetic assemblages.

2) *Photosynthetic Efficiency* (*PE*) The net hourly primary production (HPP) divided by the integrated light intensity (as PAR); this ratio measures how much incident light energy is used by photosynthetic organisms, and is computed as follows (Bott *et al.*, 1985):

$$PE = \frac{HPP\ (gm\ O_2\ m^{-2}\ h^{-1})}{PAR\ (\mu E\ m^{-1}\ h^{-1})} \times \frac{500\ (gm\ Cal\ gm^{-1}\ O_2)}{0.159\ (gm\ Cal\ \mu E^{-1})} \times 100$$

Thus, for a given light intensity, more productive photosynthetic communities should have a relatively higher PE.

3) *Production/Respiration Ratio* (*P/R*) Calculated by dividing the community production (GPP) by the community respiration (CR_{24}); this ratio has been used to classify aquatic systems as autotrophic ($P/R > 1$) or heterotrophic ($P/R < 1$) (Odum, 1956; Vannote *et al.*, 1980).

4) *Trophic Index (TI)* The mass of Chl*a* divided by the AFDM (Clark *et al.*, 1979). This ratio provides an estimate of the amount of autotrophic biomass relative to the total biomass; thus, autotrophic systems should have a relatively higher TI than heterotrophic systems. Strictly speaking, the Trophic Index is actually a community descriptor and not a functional measure, as no rate function is measured.

Both parametric and non-parametric statistical procedures were used in our analyses. A Kruskal-Wallace non-parametric analysis of ranks was used to statistically analyze all of the measured parameters and calculated indices (Table 2). Respiration (CR_{24}), and log transformed AFDM and Chl*a* could be analyzed using ANOVA. These group means, shown to be significantly different in the ANOVA, were subjected to Scheffe's multiple-range test (Sokal & Rohlf, 1981) to discern group differences. The production data (GPP) could not be transformed such that the assumption of homogeneous variances could be met for ANOVA. No transformations were used with the calculated ratios, AR, PE, P/R and TI. Significant differences reflect only the Kruskal-Wallace non-parametric analysis.

The methods used to describe the periphyton community structure are presented in detail elsewhere (La Point *et al.*, 1983). Briefly: three replicate periphyton samples were collected from riffle zone rock substrates at each station. A rubber ring enclosing an area of 3,772 mm^2 was placed on the rock to be sampled and the periphyton removed with a stiff nylon brush and collected into a 500 ml glass jar. Each replicate was adjusted to a standard volume with distilled water and acid-lugols preservative was added to produce a 1–5% (V/V) concentration.

Diatom proportional counts were performed on 10–20 ml acid digested subsamples from which permanent slide mounts were prepared. Random strips were scanned until at least 300 diatom cells were identified. A second subsample was examined using an inverted microscope to identify all nondiatoms and obtain a total count of all viable diatom frustrules to convert proportional diatom counts to cells mm^{-2}. All non-diatom cells were counted during this step as well as total viable diatom frustule number.

Statistical analyses of the structural data were limited to diatom species. Mean diatom cell abundances (cells/mm^2), mean species richness (species/sample) and mean Shannon-Wiener diversity (H') for each of the three sites were compared using ANOVA and Student-Newman-Keuls (SNK) multiple-range tests (La Point *et al.*, 1983).

Results

Of the four measured variables, AFDM, Chl*a* and CR_{24} were statistically different among sites (Table 3). Gross primary production (GPP) was not statistically different among the stations, although it did follow the same trend as CR_{24}, increasing from control through impact and recovery reaches (Fig. 3). Scheffe's multiple-range test indicates that AFDM, Chl*a* and CR_{24} measures fall into two groups, the impact and recovery sites from one group with significantly greater biomasses and respiration than at the control station.

There were statistically significant differences

Table 3. Kruskal-Wallace comparisons for measured parameters and calculated indices for Prickly Pear Creek study sites.

Parameter	X^2 Value	Probability level
Ash-free Dry Mass (AFDM)	16.45	0.001
Chlorophyll *a* Biomass (Chl*a*)	16.65	0.001
Community Respiration (CR_{24})	8.66	0.05
Gross Primary Production (GPP)	4.34	ns[a]
Assimilation Ratio (AR)	11.02	0.01
Production/Respiration (P/R)	6.74	0.05
Photosynthetic Efficiency (PE)	2.88	ns
Tophic Index (TI)	0.50	ns

[a] ns = not significant.

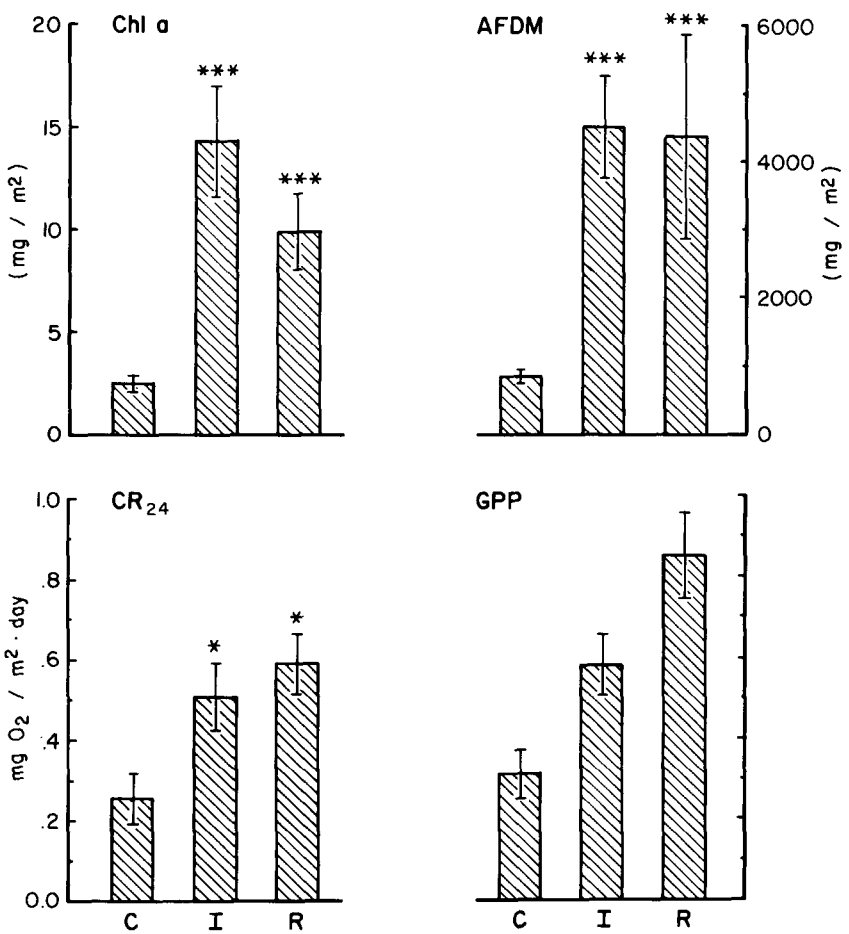

Fig. 3. Chlorophyll *a* (Chl*a*), ash-free dry mass (AFDM), community respiration (CR_{24}), and gross primary production (GPP) for substrates colonized in control (C), impact (I) and recovery (R) reaches of Prickly Pear Creek. Histograms represent means ± one standard error. Asterisks indicate means significantly different from controls (***, $P<0.001$; *, $P<0.05$).

among means for two of the four calculated indices (Table 3 and Fig. 4). Both the production respiration ratios (P/R) and the assimilation ratios (AR) were significantly greater in the control site than in the downstream sites. Production/respiration ratios (Fig. 4) markedly declined from a mean of 3.05 in the control zone to 1.13 and 1.49 in the impact and recovery zones respectively. Similarly, AR (Fig. 6) declined from 7.8 mg O_2 h^{-1} mg Chla^{-1} in the control zone to 1.8 mg O_2 h^{-1} mg Chla^{-1} in the impact zone, followed by an increase to 4.6 mg O_2 h^{-1} mg Chla^{-1} in the recovery zone.

Despite the lack of statistical significance, the pattern for PE values (Fig. 4) was identical to that of AR in that there was a decline from control to impact, 0.70 to 0.30 mg O_2 PAR^{-1}, respectively, and then an increase to 0.75 mg O_2 PAR^{-1} in the recovery zone. Differences in mean TI among the three sites were not statistically significant and varied by only 10% among sites.

Periphyton community structural parameters indicate that although diatom cell abundances increased significantly ($P \leq 0.05$) in the impact and recovery sites relative to the control sites, the diatom community richness was significantly lower in both the impact and recovery sites ($P \leq 0.05$) and the diatom diversity was significantly lower in the impact site ($P \leq 0.05$) (Fig. 5). These results indicate an over-

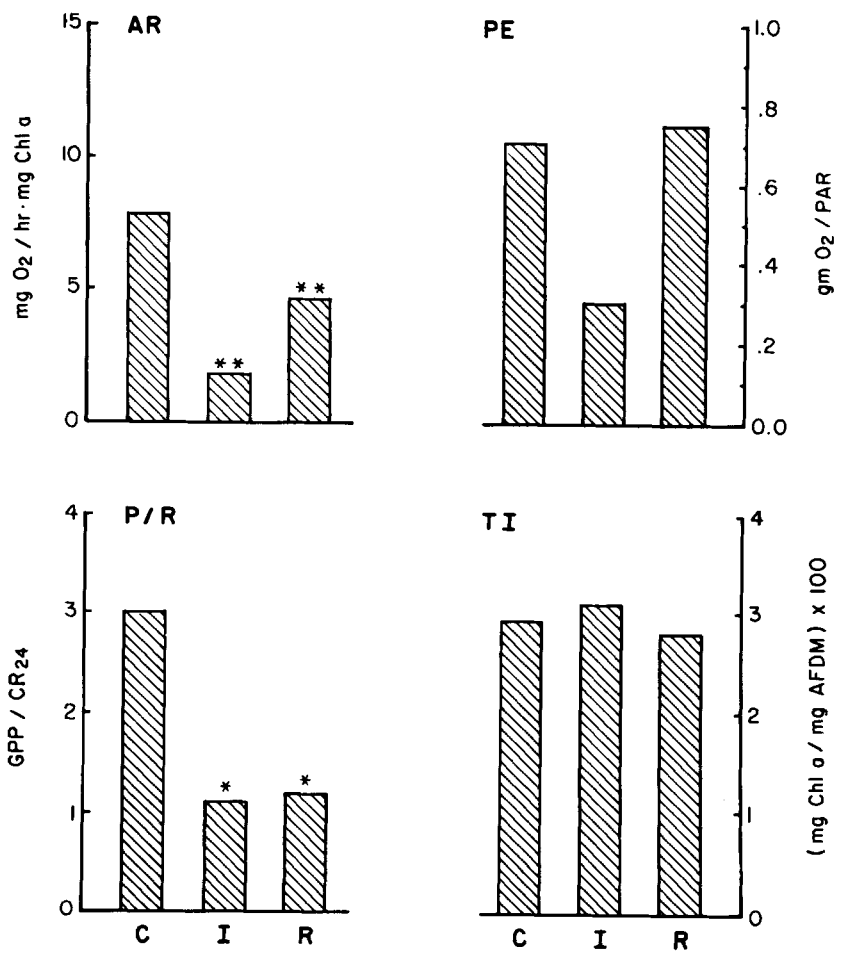

Fig. 4. Assimilation ratio (AR), production-respiration ratios (P/R), photosynthetic efficiency (PE) and trophic index (TI) values for substrates colonized in control (C), impact (I) and recovery (R) reaches of Prickly Pear Creek. Histograms represent mean values for each parameter. Asterisks indicate means significantly different from controls (**, $P < 0.01$; *, $P < 0.05$).

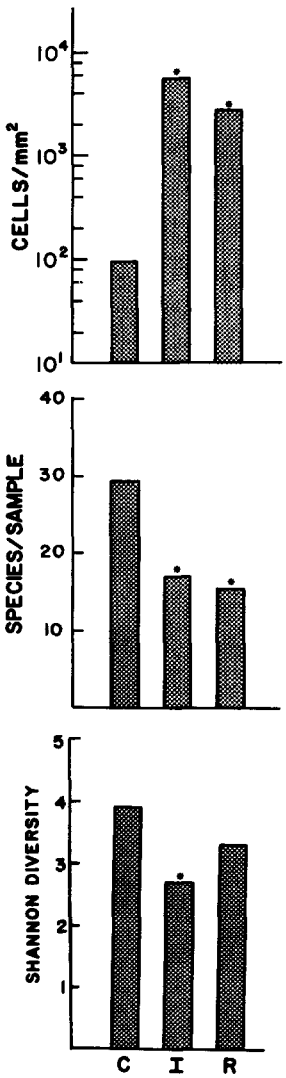

Fig. 5. Diatom cell abundances, richness and Shannon diversity for samples taken from control (C), impact (I) and recovery (R) reaches of Prickly Pear Creek. Data were obtained from the U.S. EPA survey, July 1982 (La Point et al., 1983). Histograms represent mean values from three replicate samples. Impact and recovery means were shown to be significantly different (*) by the Student-Newman-Keuls multiple-range test ($P \leq 0.05$).

all decline in community complexity in which nondiatom species predominate.

Of the 89 algal taxa identified in Prickly Pear Creek, 79 taxa were diatoms while only six were Cyanophyta (blue-green algae) and four taxa were Chlorophyta (green algae). Because very few Cyanophyta or Chlorophyta species were collected in the control zone, no statistical analyses were performed on species counts for non-diatom algae. However, relative cell abundances (number of cells of taxon/total cells counted) of non-diatom algae reveal that in the impact zone the green algae, *Ulothrix* sp. is common (6–30%) and the blue green, *Chroococcus* sp., is an abundant (61–100%) non-diatom species. Neither of these taxa were present in the control site samples; hence, their increased numbers indicated a significant change in the structure of the algal community between the control and impact sites of Prickly Pear Creak.

Discussion

Our analyses of community structure and function indicated that changes had occurred in the periphyton community of Prickly Pear Creek downstream from its confluence with Spring Creek. We were able to statistically distinguish among control, impact and recovery reaches using some parameters from both structural and functional measures.

The Community Respiration (CR_{24}), Assimilation Ratio (AR) and the Production/Respiration Ratio (P/R), all indicated statistically significant changes among study reaches. High variability in Gross Primary Production and Photosynthetic Efficiency (GPP, PE) precluded detecting statistically significant differences even though mean values appeared fairly different. The P/R ratios indicate that the Aufwuchs communities downstream from Spring Creek are less autotrophic than those above. The AR indicates less efficient primary production by the communities below Spring Creek as well. This occurs in spite of the fact that the standing crop or biomass indicators, Ash Free Dry Mass (AFDM) and Chlorophyll *a* (Chl*a*) show a significant increase from the control to the impact reaches. The Trophic Index (TI), did not indicate a significant difference. The Trophic Index, like AFDM and Chl*a*, is not actually a functional parameter but a community descriptor. Because both AFDM and Chl*a* increased in the impact and recovery reaches TI, which is simply the ratio of the two, showed no change. The TI indicates that AFDM and Chl*a* changed proportionately or nearly so. The TI would probably

be a better indicator in the case where a pollutant caused a shift from an autotrophic to a heterotrophic community such as sewage discharge.

The structural parameters, although limited to diatoms, did indicate a significant change within the impact zone. Overall, there is a decrease in community complexity. Declining species diversity and species richness and increasing cell abundances point to a loss of sensitive species with a concomitant proliferation of more resistant species (Patrick, 1978; Weber, 1981; Whitton & Say, 1975).

A comparison of structural and functional responses is not as straight-forward as are statistical sensitivity comparisons, because the two methods define different but complementary aspects of aquatic communities. Structural studies enumerate the abundance and distribution of species. Functional studies describe community level energy dynamics. Quite often, functional studies will greatly enhance the value of structural studies. In Prickly Pear Creek, for example, there was a significant change in the structure of the algal community following the Spring Creek confluence. Although this statement is definitive and statistically defensible it is not particularly informative. Is the change 'good' or 'bad'? Is there a decline in the 'biological integrity' (Weber, 1981) of the system? Functional measures can indicate the consequences of changes in community structure.

Community function measurements

Because of a lack in satisfactory standard methods, a variety of methods have been used by various investigators to study community function (Vollenweider, 1974; Bott *et al.*, 1978, Bott *et al.*, 1985). These studies have differed in primarily two ways: (1) in the method used to measure primary production, and (2) in the type of test system used (natural or artificial streams). Primary production can be measured using either dissolved oxygen or radiolabeled carbon. The types of streams studied have ranged from natural, undisturbed streams to indoor model streams with controlled lighting and flow regimes (Kimball & Levin, 1985).

Rodgers *et al.* (1979) used glass slide diatometers colonized in controlled outdoor model streams that helped decrease sample variance by controlling substrate and hydrologic variability. In addition, production was measured using carbon fixation (as ^{14}C), which has been shown to be a precise method (Vollenweider, 1974). These authors indicate that measured functional values were statistically less variable than measured structural values. Furthermore, ^{14}C assimilation (light bottle) was consistently less variable than biomass indicators (Chl*a*, AFDM and ATP).

Several studies dealing with the effects of pollutants on communities are more like the situation at Prickly Pear Creek (Clark *et al.*, 1979; Maki & Johnson, 1976; Rodgers *et al.*, 1979). Although these studies were not designed or conducted using identical methods there are some general patterns that can be drawn from the results of those studies. First, dissolved oxygen measurements tend to be less variable when measured in chambers than when measured in open water systems (Bott *et al.*, 1978; Hansmann, 1971; Busch & Fisher, 1981). Second, production measurements using ^{14}C appear to be less variable than measures made using DO techniques. However, the use of radioactive material in open water systems is rarely feasible due to regulatory constraints. Third, production and respiration values measured in model streams, using controlled lighting and flow regimes, tend to be less variable than those measured in natural streams (McIntire *et al.*, 1964; McIntire & Phinney, 1965). However, great care must be taken to ensure that the populations within model streams mimic those in the natural environment. This is especially true for streams that are controlled for extended periods of time (Patrick, 1978). Finally, artificial substrates appear to decrease variability by standardizing the size and age of the community being tested.

Although Maki & Johnson (1976) and Rodgers *et al.* (1979) stress the utility of production-respiration measures in biomonitoring, our results indicate that functional measures alone may be limited due to high variability. This difference may be due to differences in study design. Maki & Johnson (1976) tested the effects of a pulsed toxicant (the lampricide 3-trifluoromethyl-4-nitrophenol, TFM) on community metabolism, emphasizing P/R ratios. They meas-

ured an acute effect in a stream previously unexposed to the toxicant and produced a drastic effect. Our study was designed to measure production-respiration rates for periphyton communities chronically exposed to elevated heavy metal concentrations in a natural setting. In fact, as discussed above, the communities in the control and impact reaches of Prickly Pear Creek are different assemblages of species. Maki & Johnson (1976) found that an acute exposure of TFM to an established community caused large differences between functional parameters in the control (pre-exposure) and impact (post-exposure) community. In Prickly Pear Creek, however, longterm exposure has changed the community structurally; hence, functional parameters showed fewer consistent differences. We believe that functional parameters are good indicators of both chronic and acute community level stress. However, it may be difficult to discern true stress in the physiological sense and simply a difference in functional measures due to the population shifts seen in long-term studies. Obviously this emphasizes the different but complementary relationship between community functional and structural measures. When used in combination these data can describe the community level ramifications of the changes caused by contaminants (see Warren, 1971 for a discussion of this point).

Determining what is to be measured in a biomonitoring program will depend on the sensitivity of the biological parameter to changes in the chemical milieu. Changes in community structure, e.g., changes in the kinds of species and their relative abundances and distributions, have been shown to be sensitive to pollutants (La Point *et al.*, 1983; Sheehan & Winner, 1984; Giddings *et al.*, 1984). Structural measures are relatively inexpensive to monitor, given the available expertise in benthic taxonomy. Community function measures, on the other hand, require more equipment, manpower and (usually) sample numbers. Yet functional measures provide an important insight into how community metabolism responds to contaminant inputs and how changes in community structure affect community metabolism. Understanding community function will help in mitigating the toxic effects of pollutants on aquatic ecosystems and provide the framework upon which to build the relationship between community structure and community function.

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